ECOLOGICAL ECONOMICS

Research Trends

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ECOLOGICAL ECONOMICS RESEARCH TRENDS

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PREFACE

This new book presents new and important research in the field of ecological economics which is a transdisciplinary field of academic research that addresses the dynamic and spatial interdependence between human economies and natural ecosystems. Ecological economics brings together and connects different disciplines, within the natural and social sciences but especially between these broad areas. As the name suggests, the field is dominated by researchers with a background in economics and ecology. An important motivation for the emergence of ecological economics has been criticism on the assumptions and approaches of traditional (mainstream) environmental and resource economics. Ecological economics presents a more pluralistic approach to the study of environmental problems and policy solutions, characterized by systems perspectives, adequate physical and biological contexts, and a focus on long-term environmental sustainability. Ecological economics can be regarded as a version of environmental science with much emphasis on social, political, economic and behavioral issues.

Chapter 1 - The term ‘carbon footprint’ has become tremendously popular over the last few years and is now in widespread use across the media – at least in the United Kingdom. With climate change high up on the political and corporate agenda, carbon footprint calculations are in demand. Numerous approaches have been proposed to provide estimates, ranging from basic online calculators to sophisticated life-cycle-analysis or input-output based methods and tools. Despite its ubiquitous use however, there is an apparent lack of academic definitions of what exactly a ‘carbon footprint’ is meant to be. The scientific literature is surprisingly void of clarifications, despite the fact that countless studies in energy and ecological economics that could have claimed to measure a ‘carbon footprint’ have been published over decades.

This commentary explores the apparent discrepancy between public and academic use of the term ‘carbon footprint’ and suggests a scientific definition based on commonly accepted accounting principles and modelling approaches. It addresses methodological questions such as system boundaries, completeness, comprehensiveness, units, and robustness of the indicator.

Chapter 2 - In a finite world, developing a sustainable society requires both a more sustainable management of resources and a more sustainable management of human beings. Therefore deep understanding of processes underlying systems of production and consumption is needed. The question that is being asked in this chapter is whether neoclassical economics can provide solid ground to further advance the economic system and the society at large considering the known environmental limits, or approaches for future development and remedial actions should be looked for elsewhere? This chapter outlines main assumptions and limitations of the neoclassical economics. It demonstrates that the neoclassical economics that laid grounds for the current economic system is ill-suited to address the problems that the system created, e.g. environmental pollution and inequalities, and discusses how ecological economics addresses the identified deficits.

Then, institutional and individual driving forces for consumption are analysed, drawing on contributions from sociology, ecological economics and psychology. This provides the basis for discussing the change process that is needed for reaching more sustainable consumption and better quality of life. It is argued that in studying consumption it is useful to think in terms of people, places and processes. In order to understand how the change process towards sustainable consumption should be shaped a 5Es framework is suggested and discussed.

Chapter 3 **-** In recent years, there are two notable developments in the research of ecological economics. On the one hand, a tremendous amount of resources have been invested to preserve or improve ecosystem services we derive from natural resources. There is an increasing need for us to better quantify such services and assess their benefits against the investments that we have made. On the other hand, there are significant advancements in environmental modeling that can be used to facilitate the quantitative estimation of ecosystem services. In this study, the authors present an application that combines developments regarding both of these aspects.

Over the last two decades, the U.S. federal government spent billions of dollars annually on the conservation of cropland. Such expenditures were supplemented with sizable state and local government funding. However, the authors’ understanding is incomplete as to the impacts of these expenditures on the use of conservation practices and their environmental effectiveness. Expanded knowledge on these issues is greatly needed to support societal decisions as to how much more, if any, we must do to improve environmental quality to a desired level. In this article, the authors provide some insights into these issues by examining two broad questions: 1) What conservation practices are currently in place, what is their coverage, and what is the cost of these practices? 2) What have been the environmental impacts of the currently-installed conservation practices?

These two questions were examined both at the national level and, as an in-depth case study, at the state level for the state of Iowa. At the national level, information on government expenditures were gathered and previous literature was surveyed to provide an overview of what the authors know with regard to the two questions. At the state level, to address the first question, the authors collected data from various surveys, and from federal and state conservation program sources. A database for the costs and coverage of major conservation practices was developed. In order to answer the second question, the widely used Soil and Water Assessment Tool (SWAT) water quality model (Arnold et al., 1998; Arnold and Forher, 2005; Gassman et al., 2007) was employed to estimate the impacts of key conservation practices. The challenges and problems encountered in establishing accurate statewide cost estimates were discussed. The advantages and drawbacks of modeling were also identified to put our results in perspective.

Chapter 4 **-** Individuals’ contribution to electricity generation based on renewable energy sources can be channelled in two ways. The “green” market approach relies on an ‘unconditional’ contribution to renewable power while the certificate scheme represents a corresponding 'conditional' support (i.e., I can only contribute if the scheme is at place, and if so many others will also contribute). In both systems the support to renewable power is made possible through a price premium paid for these types of energy sources. In this chapter the authors draw on the economics literature on individual contributions to public goods and empirically test the overall hypothesis that the framing of renewable power support in a 'conditional' and an 'unconditional' scenario, respectively, will tend to trigger different types of moral deliberations. In the former case the deliberations concern mainly the division of efforts between individuals, while the deliberations in the latter case relate more to the characteristics of the public good in question and the perceived personal responsibility and ability to contribute to this good. This implies also that the variables determining the willingness to accept price premiums for renewable power may differ across schemes considered. The authors analyze the responses to dichotomous willingness to pay (*WTP*) questions from two different versions of a postal survey sent out to 1200 Swedish house owners. A random effects binary logit model is applied, and the estimated marginal effects support the notion that different types of factors tend to dominate choices depending on the support scheme considered. From these results follow a number of important implications for measures undertaken to increase the public's valuation of renewable power as well as the legitimacy of measures to increase renewable power production.

Chapter 5 - There is a broad recognition that sustainable land management (SLM) is crucial for ensuring an adequate, long-term supply of food, raw materials and other services provided by the natural environment to the human society. However, to date, SLM practices are the exception rather than the rule in many parts of the world. Among the causes for unsustainable land management is a general lack of understanding of the economic costs of land degradation and the benefits of sustainable land management. This paper presents a methodological framework for analyzing the benefits of sustainable land management. The framework comprises three complementary types of assessment: partial valuation, total valuation and impact analysis. The first two allow for static assessment of selected respectively all economic benefits from a certain land use. The third approach is dynamic, and allows for analyzing the costs and benefits related to changes in land use. Each approach requires the application of a number of sequential methodological steps, including (i) ecosystem function and services identification; (ii) bio-physical assessment of ecosystem services; (iii) economic valuation; and (iv) ecological-economic modeling. The framework is demonstrated by means of a simple case study in the Guadalentin catchment, SE Spain.

Chapter 6 - Since the beginning of the 1990s, analysis of the relationships between economic growth and environmental pressures has been influenced by the hypothesis known as the environmental Kuznets curve or the inverted U relationship between environmental pressure and per capita income. According to this hypothesis, once a certain income level has been achieved, economic growth leads to improvements in environmental quality. This chapter analyzes and discusses the theoretical basis of the hypothesis and the available empirical evidence. The research then analyzes the relationship between per capita GDP and various atmospheric pollutants for the Spanish case. The results show that in general, the empirical evidence for the Spanish case does not support the hypothesis. The evidence found shows that by itself, economic growth does not lead to the reduction in pollution suggested by the hypothesis, but to the opposite effect, at least if the appropriate measures to avoid this are not taken. This is especially evident in the case of greenhouse gases, for which there is a strong contrast between their actual evolution and Spain's international commitments.

Chapter 7 - It is nowadays accepted that human welfare cannot be sustained without the preservation of natural resources and without ensuring social coherence and stability. As a result, there has been growing interest among policy-makers and scientists in sustainability and sustainable development. Sustainable decision-making requires systematic methods for measuring and improving sustainability. In this work the authors present a model which uses fuzzy logic and combines basic indicators of environmental integrity and human welfare to provide a measure of sustainability. Using sensitivity analysis, the authors identify those indicators that each country should improve if it is to improve its sustainability status. Finally the authors provide a sustainability ranking of various EU countries together with the most critical indicators.

Chapter 8 - Over the last decade, Ecological Economics has provided key contributions to the human understanding of the importance and value of ecosystem services. While there has been much debate about the validity and limits of the various existing monetary valuation tools, in this chapter the authors actually wish to draw attention to the extension of the concept of valuation. The 'economistic' language of valuation has gained dominance in the policy domain, yet the literature on policy evaluation is more pluralistic, and tends to stress the importance of developing and using other quantitative and qualitative measures of value to complement monetary estimates of costs and benefits of policy interventions. Indeed 'beyond monetary valuation' lies a rich and partially unchartered territory. In this chapter the authors draw attention to the possibility to extend the notion of valuation, and its observation, beyond the individual and towards the social, cultural and institutional. The authors focus on the wider notion of property in relation to value and contrast the simple economic typology of property rights with the anthropological concept of property as social relations that govern the conduct of people with respect to the use of certain material objects. The authors argue that these recent anthropological observations, whilst mostly derived from studies of non-western groups, are (a) likely to be compatible with many observations of publics and stakeholders in developed countries and (b) can provide a framework to engage with values in human society in a more holistic way. The authors conclude that a more critical and sensitive look at enacted property relations can benefit ecological economics by complementing the insights gained through the use of more established environmental economics approaches and by potentially reaching beyond some of the limitations of these approaches to inform more contextualized and locally appropriate policy interventions.

Chapter 9 - The nature of rural development has undergone considerable change in the last 30 years. It is now recognised to include the construction of new networks, the combination of resources, and the renewed use of social, cultural and ecological capital. It often involves the reconfiguration of rural resources, many of which have previously been considered without value. Recent theories about rural development pay more attention to causes of local variations in development capacities and outcomes. Ecological economics as a discipline should attempt to equip itself with the approaches and tools to assess the complex and multi-faceted nature of rural development. This chapter discusses how various works in the field of Ecological Economics have started to build theories and approaches to do this. It goes on to review contributions from other disciplines, the Sustainable Livelihoods Approach and selected works in the fields of institutional and new institutional economics. Insight gained is used to inform the development of an approach to guide an appraisal of the rural development process and factors important in generating outcomes. The approach developed is used to assess several woodland related initiatives in Scotland. The assessment reveals the importance of factors such as culture, informal arrangements, payments in kind and networks in achieving environmental, economic and social outcomes and making the initiatives viable. It also brings to light constraints to development outcomes. It proposes that quantitative assessments of outcomes should be embedded in a broader analysis such as this in order to provide understanding and to inform policy. Also that further research is needed to test and refine tools and frameworks to guide the appraisal approach.

Chapter 10 - Creating or enlarging nature reserves to preserve key habitats and species living within those reserves is one of the important strategies to conserve biodiversity. This paper uses 0-1 programming models originating from the location science and termed SSCP (or Species Set Covering Problems), requiring representation of each and every species in the system within a minimum number of land parcels. The species under consideration in this study are the 68 mammals, reptiles and amphibians listed as threatened species in Thailand by the International Union for Conservation of Nature and Natural Reserves (IUCN), and the sites under consideration are known in Thailand as "amphoes", or small administrative districts. Since habitat requirements rarely have been introduced explicitly in reserve selection methods, this paper aims at identifying strategies to protect each threatened species while taking into account the habitat range of each particular terrestrial vertebrate of the data set. Results of the model are compared with the standard SSCP model and differences in outcomes are evaluated. Estimating the opportunity costs of converting countryside and forested areas for conservation purposes in terms of loss in economic output and incorporating them in the formulations further refines the model. Reserve networks that protect all threatened species and also that consider habitat needs are then selected at minimum opportunity costs. Results are compared with the former models to evaluate conservation policy options.

Chapter 11 - Numerous factors affect the auction price of principal forest products of the state forest enterprises in Turkey. This study tries to determine the factors affecting the price of third-class normal-sized beech timber sale by auctions. It is carried out in two rival state forest enterprises (Bartın and Yenice) of Zonguldak Regional Forest Directorate in the West Blacksea Region of Turkey. The data in this study has been obtained from a total of 149 timber auctions in the period 1998-2002. The effects of seasonal and other factors on the auction price of beech timber are investigated by variance analysis, seasonal and monthly indices, correlation, regression, and principal component analyses respectively.

The analyses indicate that beech timber prices in Yenice differ significantly over the seasons, but there are no significant differences in beech timber prices in Bartın. Moreover, the month with the highest timber price is April in both Yenice and Bartın. A correlation analysis indicates that Yenice is more dependent on Bartın than the other way around. Furthermore, according to the regression analysis, the rival prices and price mark-ups significantly explain the variation in prices. Additionally, the quantity offered during an auction significantly explains the prices in Bartın, while the log size and time between auctions significantly explains the prices in Yenice. Finally, the principal component analysis indicates that the (1) price and timing of the auctions, (2) demand level, and (3) average volume per log are the main decision variables.

Chapter 12 - The Index of Sustainable Economic Welfare (ISEW), introduced by H. Daly and J. Cobb in 1989, is an ecological economic instrument that was created in order to integrate the information embodied in GDP. Actually, GDP has been deeply criticizing for long time as an indicator of welfare: some corrections and adjustments are hence necessary in order to consider those environmental and social aspects that are relevant for human life, either ignored or wrongly treated in the official estimates of GDP. The ISEW was already calculated for several national economies but rarely for a region. The aim of this paper is to show the ISEW calculation and the results for a local economy, the Province of Pescara in Italy. This case-study is one of the first time series analyses of the ISEW (1971-2003) for a local (sub-national) territorial system. The ISEW is a tool that provides a thorough representation of the local socio-economic organization, that is important in those cases of administrative decentralization, autonomy and responsibility at the local level. For this reason, public authorities need a more and more comprehensive knowledge of the characteristics and peculiarities of the territorial system they manage. The maintenance of identity for a local system constitutes a unique resource to be emphasized, because the diversities among different sub-national areas are a prerequisite of sustainability at the national level. The results show a stagnation of the ISEW after the 1980s, compared with a constant increase of GDP during the period 1971-2003.

Chapter 13 - This article investigates the stochastic and dynamic relationship of land use in Brazilian Amazon. The authors adopt the structural VAR (SVAR) model with panel data to access for impacts of the identified exogenous sources. In this study the authors expand and refine previous works to tackle these two metodological issues. First, the authors take into account the heterogeneity among land units using the panel data analysis model [(Hsiao, 1995), (Baltagi, 1995), (Arellano, 2003)], mixing information concerning variation of individual unities with variations taking place over time. Second, in the literature of structural VAR, the contemporaneous relationship is identified on the basis of prior information supported by theoretical considerations. Notwithstanding, other distinctive appeal of this article is the employment of Directed Acyclic Graphs (DAGs) estimated by the TETRAD (Spirtes et alli, 1993) to obtain the contemporaneous causal order of the SVAR.

The article is organized as follows. Section II defines the transitional land-use model. Section III discusses identification issues to ascertain which are the true contemporaneous relationships among the land uses. The authors also introduce here the general issues regarding the DAGs model. In Section IV the authors propose a consistent methodology to estimate the model proposed in section II that contemplates the structural VAR with panel data. The description of database is done in Section V. The econometric results are presented in Section VI. Finally, the authors discuss the results and offer some concluding remarks on Section VII.

Chapter 14 - In this paper the authors propose a combination of incentives (deposit-refund system) to encourage the adoption of good environmental practices to reduce nitrate emissions due to livestock management. The main objective of this study is to describe the equilibrium conditions that make adapting good environmental practices desirable from social and private points of view. For this purpose the authors design a specific tax on a polluting input (deposit) and a subsidy for the voluntary adoption of good environmental practices (refund). In contrast to previous work (Fullerton and Wolverton, 2000), the deposit refund system is not linked to output or input but the way the input is applied. As the correct application of good environmental practices cannot be observed by the regulator, the payment of the refund does not depend on any control exercised directly by the regulator. Instead, the payment depends on the presentation of a certificate, issued by independent persons or firms certified to apply the polluting input. Those persons or firms guarantee with their reputation and future business perspectives that the polluting input has been applied in accordance with good environmental practices. In practice, the certified person can be a farmer of the region who has obtained a license to apply the polluting input.

The theoretical analysis is presented in the following section and thereafter the authors present an empirical analysis for the optimal management of livestock and cultivation activities.

Chapter 15 - Ecolabelled fish products, obtained under the sustainable fisheries certification programme, are proliferating in international food markets. This was boosted by the creation of the Marine Stewardship Council (MSC) in 1996 at the request of the WWF and the multinational company Unilever, whose function it is to accredit world fisheries sustainably managed in accordance with the directives put forward in the FAO's Code of Conduct for Responsible Fishing. The ecolabels guarantee consumers that a certain fish product comes from a fishery which conforms to regulations on sustainable fishing. And the principal factor which will determine the success or failure of ecolabelling is the acceptance of the products by the consumer.

This chapter aims to find out Spanish consumers' preferences for ecolabelled fish products. In order to do so, the authors have selected the national market's most highly-demanded fish products obtained by Spanish fleets, whose consumption has undergone a growth in trend in recent years. The results clearly show the preference of Spanish households for this type of product.

Chapter 16 - The discussion on environmental problems and economic growth is closely related to the concept of sustainable development. The best known definition is given by the World Commission on Environment and Development - the Brundtland Commission, which promoted closer links between the environment and development. They emphasized issues of social and economic sustainability. Sustainable development is now featured as a goal in dozens of national environmental policy statements, and in the opening paragraphs of Agenda 21 adopted by the Earth Summit in Rio de Janeiro in June 1992. The Johannesburg World Summit 2002 had a renewed political commitment to Agenda 21. The Brundtland report's Our Common Future defines sustainable development as "development that meets the needs of the present without compromising the ability of future generations to meet their own needs". This definition is ambiguous and raises more questions than it answers. A more precise definition would be, for example, requiring utility levels, or resource stocks, or total capital stocks including natural capital and human capital to be non-decreasing over time. Thus, sustainable paths confront standard optimal solutions as formalized in the traditional theory of economic growth. Key point for sustainable development is continuous technological improvements or productivity progress. This chapter provides measures from theoretical and empirical models. Additionally, empirical results and their interpretations are provided.

Whether pollution abatement technologies are used most efficiently is crucial in the analysis of environmental management because it influences, at least in part, the cost of alternative production and pollution abatement technologies. The role of environmental policy in encouraging or discouraging productivity growth is also well documented in the theoretical literature. As a result of this policy, two possibilities are likely. First, abatement pressures may stimulate technological innovations that reduce the actual cost of compliance below those originally estimated (e.g., Jaffe, Newell, and Stavins, 2003). Second, firms may be reluctant to innovate if they believe that regulators will respond by 'ratcheting-up' standards. In addition to the changes in environmental regulations and technology, management levels also influence ecological productivity. Therefore, whether the productivity and technological frontier expands over time is an empirical question. The principal focus of this chapter is to measure total factor productivity within a joint-production model that considers both market and environmental pollution variables, and then provide an analysis of environmental policy and productivity, especially employing an example in China.

Chapter 17 - It is usually assumed that the appraisal of the impacts experienced by present generations does not entail any difficulty. However, this is not true. Moreover, there is not a widely accepted methodology for taking these impacts into account. Some of the controversial issues are: the appropriate value for the discount rate, the choice of the units for expressing the impacts, physical or monetary units —income, consumption or investment— and the valuation of tangible and intangible goods. When approaching the problem of very long term impacts, there is also the problem of valuing the impacts experienced by future generations, through e.g., the use of an intergenerational discount rate. However, if this were the case, the present generation perspective would prevail, as if all the property rights on the resources were owned by them. Therefore, the sustainability requirement should also be incorporated into the analysis. The authors will analyze these problems in this article and show some possible solutions.

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Chapter 1

A DEFINITION OF 'CARBON FOOTPRINT'

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ABSTRACT

The term 'carbon footprint' has become tremendously popular over the last few years and is now in widespread use across the media – at least in the United Kingdom. With climate change high up on the political and corporate agenda, carbon footprint calculations are in demand. Numerous approaches have been proposed to provide estimates, ranging from basic online calculators to sophisticated life-cycle-analysis or input-output based methods and tools. Despite its ubiquitous use however, there is an apparent lack of academic definitions of what exactly a 'carbon footprint' is meant to be. The scientific literature is surprisingly void of clarifications, despite the fact that countless studies in energy and ecological economics that could have claimed to measure a 'carbon footprint' have been published over decades.

This commentary explores the apparent discrepancy between public and academic use of the term 'carbon footprint' and suggests a scientific definition based on commonly accepted accounting principles and modelling approaches. It addresses methodological questions such as system boundaries, completeness, comprehensiveness, units, and robustness of the indicator.

Keywords: carbon footprint, ecological footprint, carbon emissions, indicators, environmental accounting, input-output analysis, life-cycle analysis, hybrid analysis

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INTRODUCTION

‘Carbon footprint’ has become a widely used term and concept in the public debate on responsibility and abatement action against the threat of global climate change. It had a tremendous increase in public appearance over the last few months and years and is now a buzzword widely used across the media, the government and in the business world.

But what exactly is a ‘carbon footprint’? Despite its ubiquitous appearance there seems to be no clear definition of this term and there is still some confusion what it actually means and measures and what unit is to be used. While the term itself is rooted in the language of Ecological Footprinting (Wackernagel 1996), the common baseline is that the carbon footprint stands for a certain amount of gaseous emissions that are relevant to climate change and associated with human production or consumption activities. But this is almost where the commonality ends. There is no consensus on how to measure or quantify a carbon footprint. The spectrum of definitions ranges from direct CO_2 emissions to full life-cycle greenhouse gas emissions and not even the units of measurement are clear.

Questions that need to be asked are: Should the carbon footprint include just carbon dioxide (CO_2) emissions or other greenhouse gas emissions as well, e.g. methane? Should it be restricted to carbon-based gases or can it include substances that don’t have carbon in their molecule, e.g. N_2O, another powerful greenhouse gas? One could even go as far as asking whether the carbon footprint should be restricted to substances with a greenhouse warming potential at all. After all, there are gaseous emissions such as carbon monoxide (CO) that are based on carbon and relevant to the environment and health. What's more, CO can be converted into CO_2 through chemical processes in the atmosphere. Also, should the measure include all sources of emissions, including those that do not stem from fossil fuels, e.g. CO_2 emissions from soils?

A very central question is whether the carbon footprint needs to include indirect emissions embodied in upstream production processes or whether it is sufficient to look at just the direct, on-site emissions of the product, process or person under consideration. In other words, should the carbon footprint reflect all life-cycle impacts of goods and services used? If yes, where should the boundary be drawn and how can these impacts be quantified?

Finally, the term ‘footprint’ seems to suggest a measurement (expression) in area-based units. After all, a linguistically close relative, the ‘Ecological Footprint’ is expressed (measured) in hectares or 'global hectares' (Wackernagel et al. 2005). This question, however, has even more far-reaching implications as it goes down to the very decision whether the carbon footprint should be a mere ‘pressure’ indicator expressing (just) the amount of carbon emissions (measured e.g. in tonnes) or whether it should indicate a (mid-point) impact, quantified in tonnes of CO_2 equivalents (t CO_2-eq.) if the impact is global warming potential, or in an area-based unit if the impact is ‘land appropriation’.

Many of these questions have been discussed in the disciplines of Ecological Economics and Life-Cycle Analysis /Assessment for many years and therefore some answers are at hand. So far, however, they have not been applied to the term carbon footprint and thus a clear definition is currently missing.

This commentary addresses the questions above and attempts a clarification. We provide a brief literature overview, propose a working definition of the term 'carbon footprint' and discuss methodological implications.

A BRIEF LITERATURE REVIEW

A literature search in June 2007 for the term "carbon footprint" (i.e. where these two words stand next to each other in this order) in all scientific journals and all search fields covered by Scopus[1] and ScienceDirect[2] for the years 1960 to 2007 yielded 42 hits: 3 from the year 2005, 8 from 2006 and 31 from 2007. Most articles deal with the question of how much carbon dioxide emissions can be attributed to a certain product, company or organisation, although none of them provides an unambiguous definition of the term carbon footprint.

In most cases 'carbon footprint' is used as a generic synonym for emissions of carbon dioxide or greenhouse gases expressed in CO_2 equivalents. Some articles, however, discuss the implications of precise wording. Geoffrey Hammond writes (Hammond 2007): "…The property that is often referred to as a carbon footprint is actually a 'carbon weight' of kilograms or tonnes per person or activity." Hammond argues "…that those who favour precision in such matters should perhaps campaign for it to be called 'carbon weight', or some similar term."

Haven (2007) mentions the carbon footprint analysis of an office chair as a "life-cycle assessment which took into account materials, manufacture, transport, use and disposal at every stage of development."[3] This hints at a more comprehensive approach, rarely described in other articles. However, there is no definition or methodological description. Eckel (2007) points out that the "Assessment of a business' carbon footprint is … not just calculating energy consumption but also with increasing every scrap of data from every aspect of the business practices." Again, no clear scope of analysis is provided.

While academia has largely neglected the definition issue, consultancies, businesses, NGOs and government have moved forward themselves and provided their own definitions. In the grey literature is a plethora of descriptions, some of which are presented in Table 1.

In the UK the Carbon Trust[4] has aimed to develop a more common understanding what a carbon footprint of a product is and circulated a draft methodology for consultation (Carbon Trust 2007, see definition in Table 1). It is emphasised that only input, output and unit processes which are directly associated with the product should be included, whilst some of the indirect emissions – e.g. from workers commuting to the factory – are not factored in.

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1 Scopus (www.scopus.com) is currently the largest abstract and citation database of peer-reviewed research literature. Scopus is updated daily and covers 30 million abstracts of 15,000 peer-reviewed journals from more than 4,000 publishers ensuring a broad interdisciplinary coverage.

2 ScienceDirect (www.sciencedirect.com) contains over 25% of the world's science, technology and medicine full text and bibliographic information, including a journal collection of over 2,000 titles as well as online reference works, handbooks and book series.

3 Note that a carbon footprint of a product derived in such a way cannot just be added to the carbon footprint of an office using this chair as this would lead to double counting. Furthermore, double (or multiple) counting would occur if companies involved in the life cycle chain of the chair (manufacturing, transport, disposal) reported their full emissions (see e.g. Hammerschlag and Barbour 2003, Lenzen 2007 and Lenzen et al. 2007).

4 The Carbon Trust is a private company set up by the UK government to accelerate the transition to a low carbon economy.

Table 1. Definition of 'Carbon Footprint' from the Grey Literature

Source	Definition
BP (2007)	"The carbon footprint is the amount of carbon dioxide emitted due to your daily activities – from washing a load of laundry to driving a carload of kids to school."
British Sky Broadcasting (Sky) (Patel 2006)	The carbon footprint was calculated by "measuring the CO_2 equivalent emissions from its premises, company-owned vehicles, business travel and waste to landfill." (Patel 2006)
Carbon Trust (2007)	"… a methodology to estimate the total emission of greenhouse gases (GHG) in carbon equivalents from a product across its life cycle from the production of raw material used in its manufacture, to disposal of the finished product (excluding in-use emissions)." "… a technique for identifying and measuring the individual greenhouse gas emissions from each activity within a supply chain process step and the framework for attributing these to each output product (we [The Carbon Trust] will refer to this as the product's 'carbon footprint')." (Carbon Trust 2007, p.4)
Energetics (2007)	"… the full extent of direct and indirect CO_2 emissions caused by your business activities."
ETAP (2007)	"the 'Carbon Footprint' is a measure of the impact human activities have on the environment in terms of the amount of greenhouse gases produced, measured in tonnes of carbon dioxide."
Global Footprint Network (GFN 2007)	"The demand on biocapacity required to sequester (through photosynthesis) the carbon dioxide (CO_2) emissions from fossil fuel combustion."[5] (see also text)
Grubb & Ellis (2007)	"A carbon footprint is a measure of the amount of carbon dioxide emitted through the combustion of fossil fuels. In the case of a business organization, it is the amount of CO_2 emitted either directly or indirectly as a result of its everyday operations. It also might reflect the fossil energy represented in a product or commodity reaching market."
Paliamentary Office of Science and Technology (POST 2006)	A 'carbon footprint' is the total amount of CO_2 and other greenhouse gases, emitted over the full life cycle of a process or product. It is expressed as grams of CO_2 equivalent per kilowatt hour of generation (gCO_2eq/kWh), which accounts for the different global warming effects of other greenhouse gases.

Life-cycle thinking can be found in many other documents (references) and seem to have developed into one characteristic of carbon footprint estimates. A standardisation process has been initiated by the Carbon Trust and Defra aimed at developing a Publicly Available Specification (PAS) for LCA methodology used by the Carbon Trust to measure the embodied greenhouse gases in products (DEFRA 2007). Below, we discuss the pro's and con's of various methodologies.

The Global Footprint Network, an organisation that compiles 'National Footprint Accounts' on an annual basis (Wackernagel et al. 2005) sees the carbon footprint as a part of the Ecological Footprint. Carbon footprint is interpreted as a synonym for the 'fossil fuel footprint' or the demand on 'CO_2 area' or 'CO_2 land'. The latter one is defined as "The demand on biocapacity required to sequester (through photosynthesis) the carbon dioxide (CO_2)

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5 http://www.footprintnetwork.org/gfn_sub.php?content=glossary#co2area

emissions from fossil fuel combustion. … [It] includes the biocapacity, typically that of unharvested forests, needed to absorb that fraction of fossil CO_2 that is not absorbed by the ocean."[5] However, while individual documents have used such a land-based definition, for example the Scottish Climate Change Strategy (see Scottish Executive 2006), it has not changed the common understanding of the carbon footprint as a measure of carbon dioxide emissions or carbon dioxide equivalents in the literature.

A DEFINITION OF 'CARBON FOOTPRINT'

We propose the following definition of the term 'carbon footprint':

> "The carbon footprint is a measure of the exclusive total amount of carbon dioxide emissions that is directly and indirectly caused by an activity or is accumulated over the life stages of a product."

This includes activities of individuals, populations, governments, companies, organisations, processes, industry sectors etc. Products include goods and services. In any case, all direct (on-site, internal) and indirect emissions (off-site, external, embodied, upstream, downstream) need to be taken into account.

The definition provides some answers to the questions posed at the beginning. We include only CO_2 in the analysis, being well aware that there are other substances with greenhouse warming potential. However, many of those are either not based on carbon or are more difficult to quantify because of data availability. Methane could easily be included, but what information is gained from a partially aggregated indicator, that includes just two of a number of relevant greenhouse gases? A comprehensive greenhouse gas indicator should include all these gases and could for example be termed 'climate footprint'. In the case of 'carbon footprint' we opt for the most practical and clear solution and include only CO_2.

The definition also refrains from expressing the carbon footprint as an area-based indicator. The 'total amount' of CO_2 is physically measured in mass units (kg, t, etc) and thus no conversion to an area unit (ha, m^2, km^2, etc) takes place. The conversion into a land area would have to be based on a variety of different assumptions and increases the uncertainties and errors associated with a particular footprint estimate. For this reason accountants usually try to avoid unnecessary conversions and attempt to express any phenomenon in the most appropriate measurement unit (e.g. Keuning 1994, Stahmer 2000). Following this rationale a land-based measure does not seem appropriate and we prefer the more accurate representation in tonnes of carbon dioxide.

Whilst it is important for the concept of 'carbon footprint' to be all-encompassing and to include all possible causes that give rise to carbon emissions, it is equally important to make clear what this includes. The correct measurement of carbon footprints gains a particular importance and precariousness when it comes to carbon offsetting. It is obvious that a clear definition of scope and boundaries is essential when projects to reduce or sequester CO_2 emissions are sponsored. When accounting for indirect emissions, methodologies need to be applied that avoid under-counting as well as double-counting of emissions, therefore the word 'exclusive' in the definition.[6] Furthermore, a full life-cycle assessment of products means that all the stages of this life cycle need to be evaluated correctly (with “full” meaning “untruncated”). In the following section we discuss the methodological implications of these requirements.

METHODOLOGICAL ISSUES

The task of calculating carbon footprints can be approached methodologically from two different directions: bottom-up, based on Process Analysis (PA) or top-down, based on Environmental Input-Output (EIO) analysis. Both methodologies need to deal with the challenges outlined above and strive to capture the full life cycle impacts, i.e. inform a full Life Cycle Analysis/Assessment (LCA). Here, only a brief impression of some of their main merits and drawbacks can be provided.

Process analysis (PA) is a bottom-up method, which has been developed to understand the environmental impacts of individual products from cradle to grave. The bottom-up nature of PA-LCAs (process-based LCAs) means that they suffer from a system boundary problem – only on-site, most first-order, and some second-order impacts are considered (Lenzen 2001). If PA-LCAs are used for deriving carbon footprint estimates, a strong emphasis therefore needs to be given to the identification of appropriate system boundaries, which minimise this truncation error. PA-based LCAs run into further difficulties once carbon footprints for larger entities such as government, households or particular industrial sectors have to be established. Even though estimates can be derived by extrapolating information contained in life-cycle databases, results will get increasingly patchy as these procedures usually require the assumption that a subset of individual products are representative for a larger product grouping and the use of information from different databases, which are usually not consistent (see e.g. Tukker and Jansen 2006).

Environmental input-output (EIO) analysis provides an alternative top-down approach to carbon footprinting (see e.g. Wiedmann et al. 2006). Input-output tables are economic accounts providing a picture of all economic activities at the meso (sector) level. In combination with consistent environmental account data they can be used to establish carbon footprint estimates in a comprehensive and robust way taking into account all higher order impacts and setting the whole economic system as boundary. However, this completeness comes at the expense of detail. The suitability of environmental input-output analysis to assess micro systems such as products or processes is limited, as it assumes homogeneity of prices, outputs and their carbon emissions at the sector level. Although sectors can be disaggregated for further analysis, bringing it closer to a micro system, this possibility is limited, at least on a larger scale. A big advantage of input-output based approaches, however, is a much smaller requirement of time and manpower once the model is in place.

The best option for a detailed, yet comprehensive and robust analysis is to combine the strength of both methods by using a hybrid approach (Bullard et al. 1978, Suh et al. 2004, Heijungs and Suh 2006), where the PA and input-output methodologies are integrated. Such an approach allows to preserve the detail and accuracy of bottom-up approaches in lower

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[6] Compare with the discussion of 'shared reponsibility' as outlined by Lenzen et al. (2007).

order stages, while higher-order requirements are covered by the input-output part of the model. Such a Hybrid-EIO-LCA method, embedding process systems inside input-output tables, is the current state-of-the art in ecological economic modelling (Heijungs and Suh 2002, Heijungs et al. 2006, Heijungs and Suh 2006). The literature is just emerging and few practitioners so far have acquired the skills to carry out such a hybrid assessment. However, rapid progress and much improved models can be expected over the next few years.

The method of choice will often depend on the purpose of the enquiry and the availability of data and resources. It can be said that environmental input-output analysis is superior for the establishment of carbon footprints in macro and meso systems. In this context a carbon footprint of industrial sectors, individual businesses, larger product groups, households, government, the average citizen or an average member of a particular socio-economic group can easily be performed by input-output analysis (e.g. Foran et al. 2005, SEI et al. 2006, Wiedmann et al. 2007). Process analysis has clear advantages for looking at micro systems: a particular process, an individual product or a relatively small group of individual products.

PRACTICAL EXAMPLES

To date carbon footprints have been established for countries and sub-national regions (SEI and WWF 2007), institutions such as schools (GAP et al. 2006), products (Carbon Trust 2006), businesses and investment funds (Trucost 2006).

In this section we present two practical examples of a carbon footprint analysis that adhere to the definition suggested above. Both analyses were undertaken by researchers of the Stockholm Environment Institute at the University of York, employing an input-output based approach.

The 'UK Schools Carbon Footprint Scoping Study' (GAP et al. 2006) estimates that all schools in the United Kingdom had a carbon footprint of 9.2 million tonnes of carbon dioxide in 2001, equating to 1.3% of total UK emissions. Only around 26% of this total carbon footprint can be attributed to on-site emissions from the heating of premises, whereas the other three quarters are from indirect emission sources, such as electricity (22%), school transport (14%), other transport (6%), chemicals (5%), furniture (5%), paper (4%), other manufactured products (14%), mining and quarrying (2%) and other products and services (3%).

The second example is a calculation of the carbon footprint of UK households, taking into account direct and indirect emissions occurring on UK territory due to consumption activities of UK residents as well as the (indirect) emissions that are embodied in imports to the UK. The results, presented in the 'Counting Consumption' report (SEI et al. 2006), suggest that the carbon footprint of an average UK household was 20.7 tonnes of CO_2 in 2001. A breakdown of this total is presented in Figure 1.

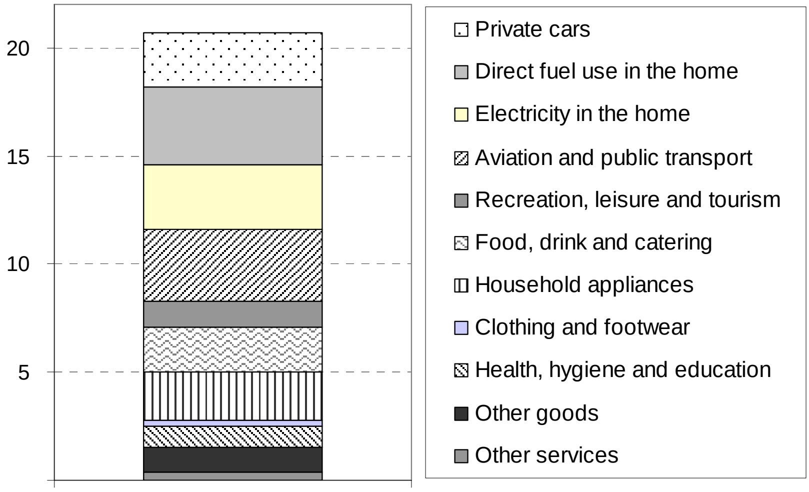

Figure 1: Carbon dioxide emissions associated with UK household consumption in 2001 (t CO_2 per household) (SEI et al. 2006)

Direct emissions occur through heating and car use. Indirect emissions are the emissions that occur during the generation of electricity and the production of goods and services (whether they are produced in the UK or in other countries). They make up 70 per cent of the almost 21 tonnes of CO_2 per household. Transport (private cars, aviation and public transport) account for 28% of total emissions. Electricity use in the home and use of fuels for space and water heating in the home account for almost one third of the emissions.

These findings have also been published by the UK Department for the Environment, Food and Rural Affairs (DEFRA) in the 'The Environment in your Pocket' publication (DEFRA 2006).

CONCLUSIONS

A review of scientific literature, publications and statements from the public and private sector as well as general media suggests that the term 'carbon footprint' has become widely established in the public domain albeit without being clearly defined in the scientific community. In this commentary we suggest a definition of the term 'carbon footprint' and hope to stimulate an academic debate about the concept and process of carbon footprint assessments.

We argue that it is important for a 'carbon footprint' to include all direct as well as indirect CO_2 emissions, that a mass unit of measurement should be used, and that other greenhouse gases should not be included (or otherwise the indicator should be termed 'climate footprint'). We discuss the appropriateness of two major methodologies, process analysis and input-output analysis, finding that the latter one is able to provide comprehensive and robust carbon footprint assessments of production and consumption activities at the meso level. As an appropriate solution for the assessment of micro-systems such as individual products or services we suggest a Hybrid-EIO-LCA approach, where life-cycle assessments are combined with input-output analysis. In this approach, on-site, first- and second-order process data on environmental impacts is collected for the product or service system under study, while higher-order requirements are covered by input-output analysis.

Whatever method is used to calculate carbon footprints it is important to avoid double-counting along supply chains or life cycles. This is because there are significant implications on the practices of carbon trading and carbon offsetting (see e.g. Hammerschlag and Barbour 2003, Lenzen 2007, Lenzen et al. 2007).

REFERENCES

BP (2007) "What is a Carbon Footprint?", http://www.bp.com/liveassets/bp_internet/globalbp/STAGING/global_assets/downloads/A/ABP_ADV_what_on_earth_is_a_carbon_footprint.pdf

Bullard, C.W., Penner, P.S. and Pilati, D.A. (1978) "Net energy analysis: Handbook for combining process and input-output analysis". *Resources and Energy* 1(3): 267-313. http://dx.doi.org/10.1016/0165-0572(78)90008-7.

Carbon Trust (2006) "Carbon footprints in the supply chain: the next step for business". Report Number CTC616, November 2006, The Carbon Trust, London, UK. http://www.carbontrust.co.uk..

Carbon Trust (2007) "Carbon Footprint Measurement Methodology, Version 1.1". 27 February 2007, The Carbon Trust, London, UK. http://www.carbontrust.co.uk.

DEFRA (2006) "The Environment in Your Pocket". Department for Environment, Food and Rural Affairs, London. http://www.defra.gov.uk/environment/statistics/eiyp/index.htm.

DEFRA (2007) "Step forward on reducing climate change impacts from products." *DEFRA press release, 30 May 2007*, from http://www.defra.gov.uk/news/2007/070530a.htm.

Eckel, A. (2007) *Gravure* 21(2): 35-36.

Energetics (2007) "The Reality of Carbon Neutrality", London, UK http://www.energetics.com.au/file?node_id=21228

ETAP (2007) "The Carbon Trust Helps UK Businesses Reduce their Environmental Impact", Press Release, http://www.ec.europa.eu/environment/etap/pdfs/jan07_carbon_trust_initiative.pdf

Foran, B., Lenzen, M. and Dey, C. (2005) "Balancing Act: A triple bottom line analysis of the 135 sectors of the Australian economy". CSIRO Resource Futures and The University of Sydney, Canberra, ACT, Australia. http://www.cse.csiro.au/research/balancingact

GAP, SEI and Eco-Logica (2006) "UK Schools Carbon Footprint Scoping Study". Report by Global Action Plan, Stockholm Environment Institute and Eco-Logica Ltd for the Sustainable Development Commission, London, March 2006. http://www.sd-commission.org.uk/publications.php?id=389.

GFN (2007) "Ecological Footprint Glossary". Global Footprint Network, Oakland, CA, USA. Accessed July 2007 from http://www.footprintnetwork.org/gfn_sub.php?content=glossary.

Grubb & Ellis (2007) "Meeting the Carbon Challenge: The Role of Commercial Real Estate Owners, Users & Managers". April 2007, Grubb & Ellis Company, Chicago, USA. http://www.grubb-ellis.com/research/Whitepapers/CarbonWhitefnl.pdf.

Hammerschlag, R. and Barbour, W. (2003) "Life-Cycle Assessment and Indirect Emission Reductions: Issues Associated with Ownership and Trading". Institute for Lifecycle Environmental Assessment (ILEA), Seattle, Washington, USA, May 2003. http://www.ilea.org/downloads/LCAEmissReductions.pdf.

Hammond, G. (2007) "Time to give due weight to the 'carbon footprint' issue". *Nature* 445(7125): 256. http://dx.doi.org/10.1038/445256b.

Haven, J. (2007) *Environment Business* 129: 27.

Heijungs, R. and Suh, S. (2002) "The computational structure of life cycle assessment". Kluwer Academic Publishers, Dordrecht, The Netherlands.

Heijungs, R. and Suh, S. (2006) "Reformulation of matrix-based LCI: from product balance to process balance". *Journal of Cleaner Production* 14(1): 47-51. http://dx.doi.org/10.1016/j.jclepro.2005.05.022.

Heijungs, R., de Koning, A., Suh, S. and Huppes, G. (2006). "Toward an Information Tool for Integrated Product Policy: Requirements for Data and Computation". *Journal of Industrial Ecology* 10(3): 147-158. http://dx.doi.org/10.1162/jiec.2006.10.3.147.

Keuning, S.J. (1994) "The SAM and Beyond: Open SESAME!", *Economic Systems Research* 6(1): 21-50.

Lenzen, M. (2001) "Errors in Conventional and Input-Output-based Life-Cycle Inventories". *Journal of Industrial Ecology* 4(4): 127-148. http://dx.doi.org/10.1162/10881980052541981.

Lenzen, M. (2007) "Double-counting in Life-Cycle Assessment". *Journal of Industrial Ecology*: submitted.

Lenzen, M., Murray, J., Sack, F. and Wiedmann, T. (2007) "Shared producer and consumer responsibility - Theory and practice". *Ecological Economics* 61(1): 27-42. http://dx.doi.org/10.1016/j.ecolecon.2006.05.018.

POST (2006) "Carbon footprint of electricity generation". POSTnote 268, October 2006, Parliamentary Office of Science and Technology, London, UK. http://www.parliament.uk/documents/upload/postpn268.pdf.

Patel, J. (2006) "Green sky thinking". *Environment Business* 122: 32.

Scottish Executive (2006) "Changing Our Ways: Scotland's Climate Change Programme", March, 2006. Scottish Executive, Edinburgh, UK. http://www.scotland.gov.uk/Publications/2006/03/30091039/0

SEI and WWF (2007) "The Carbon Footprints of Local Authorities in England". Stockholm Environment Institute and WWF-UK, forthcoming.

SEI, WWF and CURE (2006) "Counting Consumption - CO_2 emissions, material flows and Ecological Footprint of the UK by region and devolved country". Published by WWF-UK, Godalming, Surrey, UK, 2006. http://www.ecologicalbudget.org.uk.

Stahmer, C. (2000) "The magic triangle of input-output", Paper presented at the *XIII International Conference on Input-Output Techniques*, 21-25 August, Macerata, Italy.

Suh, S., Lenzen, M., Treloar, G.J., Hondo, H., Horvath, A., Huppes, G., Jolliet, O., Klann, U., Krewitt, W., Moriguchi, Y., Munksgaard, J. and Norris, G. (2004) "System boundary selection in life-cycle inventories using hybrid approaches". *Environmental Science & Technology* 38(3): 657-664.

Trucost (2006) "Carbon Counts: The Trucost Carbon Footprint Ranking of UK Investment Funds". June 2006, Trucost, London, UK. http://www.trucost.com//carbon_news.html.

Tukker, A. and Jansen, B. (2006) "Environmental impacts of products: A detailed review of studies". *Journal of Industrial Ecology* 10(3): 159-182. http://dx.doi.org/10.1162/jiec.2006.10.3.159.

Wackernagel, M. and Rees, W.E. (1996) "Our Ecological Footprint - Reducing Human Impact on the Earth". New Society Publishers Gabriola Island, B.C., Canada.

Wackernagel, M., Monfreda, C., Moran, D., Wermer, P., Goldfinger, S., Deumling, D. and Murray, M. (2005) "National Footprint and Biocapacity Accounts 2005: The underlying calculation method". May 2005, Global Footprint Network, Oakland, CA, USA. http://www.footprintnetwork.org.

Wiedmann, T., Barrett, J. and Lenzen, M. (2007) "Companies on the Scale - Comparing and Benchmarking the Footprints of Businesses". *International Ecological Footprint Conference*, May 8-10, 2007, Cardiff, UK. http://www.brass.cf.ac.uk/uploads/Wiedmann_et_al_P36.pdf.

Wiedmann, T., Minx, J., Barrett, J. and Wackernagel, M. (2006) "Allocating ecological footprints to final consumption categories with input-output analysis". *Ecological Economics* 56(1): 28-48. http://dx.doi.org/10.1016/j.ecolecon.2005.05.012.

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Chapter 2

CONSUMPTION AND ECOLOGICAL ECONOMICS: TOWARDS SUSTAINABILITY

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1. ABSTRACT

In a finite world, developing a sustainable society requires both a more sustainable management of resources and a more sustainable management of human beings. Therefore deep understanding of processes underlying systems of production and consumption is needed. The question that is being asked in this chapter is whether neoclassical economics can provide solid ground to further advance the economic system and the society at large considering the known environmental limits, or approaches for future development and remedial actions should be looked for elsewhere? This chapter outlines main assumptions and limitations of the neoclassical economics. It demonstrates that the neoclassical economics that laid grounds for the current economic system is ill-suited to address the problems that the system created, e.g. environmental pollution and inequalities, and discusses how ecological economics addresses the identified deficits.

Then, institutional and individual driving forces for consumption are analysed, drawing on contributions from sociology, ecological economics and psychology. This provides the basis for discussing the change process that is needed for reaching more sustainable consumption and better quality of life. It is argued that in studying consumption it is useful to think in terms of people, places and processes. In order to understand how the change process towards sustainable consumption should be shaped a 5Es framework[1] is suggested and discussed.

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[1] It is an extension of the 4Es framework developed in SCR (2006a). *Shifting opinions and changing behaviours*. London, Sustainable Consumption Roundtable**:** 76.

2. INTRODUCTION

The current economic model is built on ideas of neoclassical economics and its theories of consumption. Consumption is seen as extremely important component in gross domestic product (GDP) – the main traditional indicator of economic growth, which beyond eliminating poverty, ultimately increases well-being and happiness. Indeed, the expansion of global consumption expenditures allowed considerable advances in human development resulting in substantial improvements in health care, communication, and education. Besides economic sphere, the neoclassical economics has also dominated policy making across the globe and led to significant improvements in the level of technological and industrial development.

Until recently, the neoclassical economics was well suited to provide guidance on the development of the economies. However, lately, a range of side effects of the "well-functioning" market system has been brought to light, including escalating environmental problems, health deterioration and increasing social disparities between the rich and the poor. Thus, together with societal progress, industrial revolution and economic growth have generated externalities that put under threat the very goal of economic development – well-being and the very basis of our society – biophysical system.

The outcome is not surprising, since the neoclassical economics evolved in situation when environmental problems were not yet known or as acute as they are nowadays. The question that is being asked in this chapter is whether neoclassical economics can provide a solid approach to further advance the economic system and the society at large considering the known environmental limits, or approaches for future development and remedial actions should be looked for elsewhere? Eco-efficiency approaches undertaken to reduce the environmental impacts of the systems of production and consumption have not been able to cope with mounting environmental and social consequences of increasing consumption levels. New knowledge is needed in order to address consumption-related problems, since it is clear that the fundamental assumptions of the neoclassical economic theory provide little insight into intricate processes that shape consumer behaviour and consumption patterns and levels. The neoclassical economics is unable to cope with the complexity of societal consumption-related problems.

Ecological economics, on the other hand, seems to be well suited for the challenge, since it is capable to provide better insights into the driving forces of the current systems of consumption and production, drawing on contribution from many disciplines, including sociology, psychology and environmental studies. It therefore has a higher potential to discuss and envision the required system-level changes toward sustainable society. It is better suited for addressing problems associated with the scale of the economy and aggregate and per capita consumption of resources.

The goal of this chapter is to briefly examine the main assumptions and limitations of the neoclassical economic theory of consumption, to outline the most important viewpoints of the ecological economics on the complexity of consumption and to bring into light some of the knowledge from economic, sociological and psychological studies that can explain institutional and individual drivers and reasons for consumer behaviour. This knowledge allows better understanding of the consumption complexity and provides the basis for discussing the change process that is needed for reaching more sustainable consumption and better quality of life. It is argued that in studying consumption it is useful to think in terms of people, places and processes. In order to understand how the change process towards sustainable consumption could be shaped a 5Es framework is suggested and discussed. Future contributions of the ecological economics that are needed to further the discourse on sustainable consumption are outlined in conclusions.

Traditional Economic Perspective on Consumption

Neoclassical economics refers to a general approach in economics that deals with the allocation of scarce resources by focusing on the determination of prices, outputs, and income distributions in markets through supply and demand. Neoclassical economics is conventionally dated from contributions of William Stanley Jevons' "Theory of Political Economy" (1871), Carl Menger's "Principles of Economics" (1871), and Leon Walras's "Elements of Pure Economics" (1874 – 1877).

It rests on some basic assumptions that simplify the economic analysis. *First* assumption is that people are rational actors who have rational preferences. *Second* assumption is that individuals maximise their utility and companies maximise their profits. *Third* assumption is that people act independently on the basis of full and relevant information. These assumptions of neoclassical economics lay grounds for a wide range of theories about various areas of economic activity. For example, profit maximisation lies behind the neoclassical theory of the firm, while the utility maximisation is the source for the neoclassical theory of consumption.

The Neoclassical Economic Theory of Consumption

The neoclassical economic theory of consumption dominated scientific debate from mid-19th century and prevailed well into the 20th century. Following the assumptions of the neoclassical economics, the neoclassical economic theory of consumption is also based on a number of simplifications and assumptions about consumer behaviour and consumer interaction with the surrounding world (see more modern authors: Friedman 1957; Douglas and Isherwood 1980; Harris 1997; van den Bergh and Ferrer-i-Carboneli 2000). It is useful to consider how these assumptions influence the economic analysis and understanding of consumption-related processes and decision-making by private and business actors.

Generally speaking, neoclassical economics takes the insatiability of consumer desires and the sovereignty of consumer choices as definite features of consumption without going into their underlying motivations. The fundamental assumption is that consumer is a rational decision-maker with well-defined aspirations and rational preferences for goods and services. Tastes and preferences are perceived as being exogenous to the economic system and the goal of economics is to optimally satisfy these preferences (Silberberg 1978). Since tastes and preferences usually do not change fast, in the short term this assumption makes sense and assists the traditional economic analysis. Since preferences are fixed, they are not influenced by changes in other people's behaviour or other factors and therefore can be aggregated in the analysis (Norton, Costanza et al. 1998). The position of many traditional economists is that if preferences can be influenced, then there is a danger that they may be influenced by specific societal powers, regimes or private interests. This line of thought thus accepts that preferences can be changed at individual level, but no external forces should influence the decisions so that consumer sovereignty could remain untouched (Norton, Costanza et al. 1998).

According to neoclassical economics, the goal of individuals is to maximise their utility and the goal of companies is to maximise their profits. The traditional microeconomic consumption theory describes how rational consumers allocate their budget among different goods in order to maximise private benefit (utility), since the desires are insatiable and consumption is determined and measured in terms of quantities of products and services consumed. Utility maximisation thus consists of a process of balancing consumption among different alternatives available on the market. This simplification leads to that consumer choices are easily predictable on the basis of income and price. Thus policy makers and businesses can modify consumers' demand by influencing prices and income levels. This understanding leads to proliferation of financial mechanisms not only in economic sphere, but also in environmental economics and in environmental policy (van den Bergh and Ferrer-i-Carbonell 2000).

In the neoclassical economics incomes and prices are treated as one of the main limiting factors and consumer's choice is essentially an exercise of finding the optimal condition where the utility/price ratios of the chosen goods are equal (Lancaster 1971; Michael and Becker 1973) (Toumanoff and Farrokh 1994). This model implies that since preferences are taken as given, only changes in the budget constraint (income or prices) affect consumers demand. Thus in this simplistic model, the main prerequisite for consumption and the main limiting factors, assuming adequate supply of goods and services at market prices, are the availability of income and time to consume or rather its allocation between different product and service alternatives (Becker 1965). The level of income depends on the working time, qualification and how well it is applied in producing value-added. Generally, higher incomes are possible along with technological improvements, which provide higher productivity and lower production costs and final prices of goods. Since consumption is directly linked to the available budget, consumers are interested in increasing personal incomes, which is possible by raising productivity, i.e. producing more with fewer production factors (labour, capital, technology or resources). This may lead to over-supply and the resulting price drops and rebound effects, which in turn can promote consumption and result in higher environmental impacts. Other important factors are personal tastes, but these often fall outside the realm of economics and most of the times traditional economists restrict themselves to discussing the role of prices and incomes in determining consumption choices.

In the neoclassical economics, consumers maximise private utility in the world of perfect - full and relevant - information and market competition. They consume goods without considering social, ethical or environmental issues, e.g. culture, institutional frameworks, social interactions. They decide how to allocate available resources in time by considering their current and future incomes as well as the interest rates (van den Bergh and Ferrer-i-Carbonell 2000). Therefore, economic analysis can study the impact of changing prices and incomes on purchasing behaviour and consequently develop policies that change consumer behaviour into the desired direction by changing prices or incomes, or both, as is the case with environmental tax policies.

Limitations and Use of the Neoclassical Economic Theory of Consumption

A change in neoclassical economics took place in the mid-1930s, with the publications of Joan Robinson's "The Economics of Imperfect Competition" (1933) and Edward H. Chamberlin's "The Theory of Monopolistic Competition" (1933), which introduced models of imperfect competition, new theories, such as Herbert Simon's theory of bounded rationality, and new tools, such as the marginal revenue curve.

Since then the main premises of the neoclassical economics have been critically scrutinised. Neoclassical economics has been criticised for having a normative bias: it is argued that it does not focus on explaining real life economies, but instead on describing an "utopia", in which Pareto optimality is achieved. One of the major deficits of the neoclassical economics is that it largely ignores the problems of scale both in time and space assuming infinite substitution possibilities among resources and unlimited technological change. Although it is important to study the individual level, especially in microeconomic perspective, the primary focus on individuals in the economy may make the analysis of broader and long term issues difficult. It may prevent posing the question regarding the desirability and stability of the economic system on a planet with finite resources.

The assumption of the exogenous nature of consumer preferences has been broadly questioned, since it cannot provide correct guidance for understanding behaviour and for decision making over long term. Preferences of people do change, subject to influences from education, advertising, social groups and cultural developments. Therefore, the definition of how to optimally satisfy evolving preferences and the bundle of material and non-material satisfiers should be changing accordingly (Norton, Costanza et al. 1998).

The assumption of the perfect knowledge for market actors has also been heavily criticised both at the individual and company levels. Taking into account the concept of bounded rationality with lack of information and cognitive limitations, it is clear that consumers cannot be efficient in their choices and that neoclassical economics failed to provide sufficient explanation of consumption processes. Consumer behaviour has been found to be far more complicated than just rational response to price signals being influenced by different internal and external drivers induced by human psychology, social norms and institutional settings. Since then, the neoclassical economic theory has been criticised by many scholars for its over-simplification of reality, particularly on the assumptions of a rational and sovereign consumers with limitless consumption needs (see: Scitovsky 1976; Daly and Cobb 1989; Daly 1996; Bianchi 1998; Samuelson and Nordhaus 2004).

When it comes to the assumption of companies maximising their profits, it has been criticised because this ideal may come at the expense of social issues. The neoclassical economics addresses this problem with concepts of private versus social utility.

Finally, the neoclassical economics does not hold true in its assumption that increasing levels of material ownership lead to increasing satisfaction with life, as, for example, millions of poor people in India consider themselves happy, while the opposite is true for some rich people in the USA (WVS 2006).

Despite the existing criticism of the neoclassical economic theory of consumption, policy makers heavily rely on the traditional neoclassical economics. Their authority is heavily based on their capacity to increase economic growth, and their measures to sustain it are sometimes quite controversial. For example, in order to keep GDP growing, policies and economic instruments are used that under-price natural resources or provide perverse subsidies (EEA 2006). More positive outcomes have policies built on the assumption that lack of appropriate information and improper prices are the main barriers to more sustainable behaviour of consumers. Based of these assumptions, the emerging policy instruments focus on reducing market failures by providing information to consumers and by adjusting the prices to internalise environmental and social costs. Consumer sovereignty is often used as an argument to justify the deficit of consumption oriented policies even though the notion per se is clearly questionable, since consumers are not independent actors and are heavily influenced by available information, marketing and advertising and become locked-in by provided products and existing infrastructure.

Being simplistic, i.e. ignorant to the exogenous factors, such as society, culture and institutions, the neoclassical economic theory is nevertheless well-suited for modelling the effects of technological and labour productivity improvements. It is also convenient for environmental policy interventions using economic instruments, such as e.g. road fees, carbon taxes or any other general pollution surcharge taxes (Kletzan, Köppl et al. 2002).

3. Ecological Economics Perspective on Consumption

An important driving force for the emergence of ecological economics has been criticism on the assumptions and tools of the neoclassical economics. Ecological economics presents a more pluralistic approach to the study of environmental and economic systems and integrates tools and methods of environmental, social, political, economic and behavioural studies. As its founding fathers can be regarded economists N. Georgescu-Roegen, K.E. Boulding and H.E. Daly, ecologists C.S. Holling, H.T. Odum and R. Costanza, and physicist R.U. Ayres.

3.1 Ecological Economics as "The Management of Knowledge about the House"

Ecological economics is the management of knowledge about the house (earth)[2] (Costanza 1996). It is not guided by a single theory or a framework, but is based on pluralistic views of researchers and academicians and the shared commitment to explore possibilities for devising new knowledge in a multidisciplinary environment. Ecological economics is a trans-disciplinary effort that aims at bridging together the natural and social sciences (Costanza 1996). The goal of ecological economics is to provide a deeper understanding of the complexity and linkages between ecological and economic systems. As a consequence, multiple tools from all contributing disciplines are applied to the questions raised within ecological economics paradigm. The methodological goal is to develop tools and approaches that effectively deal with the issues of sustainability from the ecological economic perspective. The obtained knowledge is used to develop policies for a more sustainable future, including environmental sustainability and fair distribution of resources.

[2] oikos= "house"; logy= "study or knowledge"; nomics= "management"

> "[Ecological economics] is problem-focused rather than concerned with abstract modeling, and, in contrast to conventional neo-classicism, ecological economics shifts the focus from micro to macro and relevant time frames from the very short term to deep time. Ecological economics complements the relational and synergistic realities of ecology. It is, therefore, a holistic rather than a reductionist endeavor and gives due weight to process, change and flux, rather than stasis. Such an economics also incorporates an ethical and visionary dimension—necessary because grounding economic thought within a broader and prior context requires strictures of "ought" to govern contextual relationships." (Hay 2002): 233.

Ecological economics aims at efficient allocation of resources (Pareto optimal), at fair and just distribution of income or wealth and at defining the physical scale of the economy relative to the ecosystem so that adverse effects are not generated faster than positive effects so that environmental sustainability is maintained (Daly 2003). The goals of ecological economics are thus:

- Economically efficient allocation
- Ecologically sustainable scale
- Socially fair distribution

The main premise of the ecological economics is the assertion that economy and nature are interrelated and complementary, and that human economic system depends on the nature and the environment. In contrast to this understanding, the neoclassical economics assumes that the environment is a subset of the human economy and undermines the contribution of the natural capital to the creation of wealth. It claims that infinite economic growth is possible as long as it is possible to substitute natural capital for manmade capital and to allocate resources efficiently. Ecological economics challenges this normative approach that is taken by the neoclassical economics towards the use of natural resources arguing that it undermines the value of the natural capital by perceiving it as being interchangeable with human capital. In the view of ecological economics human capital is instead dependent upon natural capital and cannot substitute it fully. This means that if natural capital is in short supply, then according to the neoclassical economics it can be substituted by the man-made capital, while according to the ecological economics it becomes a limiting factor.

Regarding the optimum scale of the economy, the ecological economics clearly distinguishes between economic growth and economic development. According to Daly, growth is a "*…quantitative increase in the scale of the physical dimensions of the economy-,"* while development is *" the qualitative improvement in the structure, design and composition of physical stocks and flows, that result from greater knowledge, both of technique and of purpose*" (Daly 1987). Therefore, if our goal is sustainable *development* and not *growth*, traditional policies building on the premises of the neoclassical economics will not be able to secure and sustain it.

The ecological economics takes into consideration interests of future generations, as well as the value basis that is much broader than traditional economic approaches. The neoclassical economics ignores the distribution (equity) problem of allocating resources, both inter- and intra-generational, which is addressed in the ecological economics. "*The market does not distinguish an ecologically sustainable scale of matter-energy throughput from an* *unsustainable scale, just as it does not distinguish between ethically just and unjust distributions of income. Sustainability, like justice, is a value not achievable by purely individualistic market processes.*" (Daly 1986). Related to the latter is the example of another distributional problem - the bias of allocating resources to private consumption and away from collective goods (Dodds 1997), imbalance of which increases as private incomes rise.

Finally, ecological economics reconciles the macro- and micro-scales of consumption. On the one hand, it is concerned with studying global phenomena with the entire world, countries and sectors being the unit of analysis. On the other hand, ecological economics is concerned and goes deep into the functions at the household level and into the individual processes that shape purchasing behaviour and lifestyle choices of people.

In relation to consumption, the goal of ecological economics is to provide a trans- and multidisciplinary understanding of the complex processes and driving forces shaping consumption beyond those proposed by the neoclassical economics and suggest ways for how more sustainable consumption patterns and levels can be reached and more sustainable societies created.

3.2 Ecological Economics' Stance on the Complexity of Consumption

History of consumption and consumerism has its roots in a number of economic, social, cultural and geo-political transformations that evolved throughout history. Therefore, consumption has been discussed in a number of different scientific disciplines, such as economics, sociology, psychology, behavioural science, etc. and each discipline understands consumption from a different perspective. As Fine expresses it, "…for economists, consumption is used to produce utility; for sociologists, it is a means of stratification; for anthropologists – a matter of ritual and symbol; for psychologists – the means to satisfy or express physiological and emotional needs; and for business, it is a way of making money" (Fine 1997). Ecological economics incorporates and builds on the knowledge and methods used and acquired in different disciplines and offers a much richer understanding of the processes underlying consumer decision making.

According to ecological economics, people preferences evolve continuously, subject to changes in lifestyles, fashion, economic situation, technological progress, developments in the peer social group, influence of media and other cultures (van den Bergh and Ferrer-i-Carbonell 2000). The main question is therefore how preferences change, and the neoclassical economics cannot provide comprehensive answers to this question.

According to the neoclassical economics, the main goal of people is to maximise utility. Sociological studies however offer an entire range of explanations as to why people consume. Some of explanations include satisfaction of needs, influences of social or group behaviour, habit and routine behaviour (van den Bergh, Ferrer-i-Carbonell et al. 1998), and many other.

Another difference between ecological economics and neoclassical economics is the understanding of the relation between economic growth (GNP) and well-being. The dominant theory of the neoclassical economics supports the "the more the better" notion (Samuelson 1947), while ecological economics considers that "small is beautiful" (Schumacher 1973). Following this notion, the neoclassical economics sees a high coupling between the economic growth and well-being, while according to ecological economics studies the gap between GNP and happiness widens as we move toward higher levels of consumption.

This gap to some extent can be explained by how the perception of "well-being", defined purely as the amount of material assets, is changing with increasing income. Both Scitovsky (1992) and Easterlin (1973) argued that since satisfaction with life is culturally determined, the increase in income does not automatically lead to improved perceived well-being as views on wealth and poverty adjust over time (hedonic adaptation) and are dependent on the comparison with social peers. Easterlin (2003) demonstrates that subjective well-being correlates well with the level of education, health and marital status and not very well with income. *"People make decisions assuming that more income, comfort, and positional goods will make them happier, failing to recognize that hedonic adaptation and social comparison will come into play, raise their aspirations to about the same extent as their actual gains, and leave them feeling no happier than before. As a result, most individuals spend a disproportionate amount of their lives working to make money, and sacrifice family life and health, domains in which aspirations remain fairly constant as actual circumstances change, and where the attainment of one's goals has a more lasting impact on happiness. Hence, a reallocation of time in favor of family life and health would, on average, increase individual happiness"(Easterlin 2003).* Thus, increasing per capita consumption does not necessarily lead to improving the level of well-being, once a certain level of material prosperity has been reached, but may instead bring about harmful effects on the environment and reduce quality of life of individuals. Instead of focusing on economic growth, ecological economics concentrates on economic development, quality of life and on identifying means of increasing human well-being that do not lead to detrimental environmental and social/distributional effects.

Besides income, prices and time identified and studied by the neoclassical economics as the main limiting factors of consumption, ecological economics defines other types of constraints to consumption, including social norms, policies and standards, family routines and group pressure. Social norms evolve over time due to collective behaviour of the entire population, intertwined with institutional and individual developments (Azar 2004). It is believed that the evolution of social norms can be affected by introducing and sharing visions of preferred states of the world (Costanza 2000). In addition to social norms, consumption is shaped by socio-psychological drivers (Røpke 1999a) and other factors, such as demography, institutions and culture (Gatersleben and Vlek 1998). Thus, the forces shaping consumption decisions can be divided into two levels: individual needs and wants of people and collective societal structures that stem from, but also shape, individual wants and needs.

The modern consumer theories perceive individuals as active members of the market and as one of the main actors for creating utility. The main notion is that goods are simply inputs to the consumption process, and consumers are creating utility by applying individual skills. The end result is a great variety of ways for consumers to produce utility, which makes every day decision-making process a complex task.

4. Understanding Consumption Complexity

Our understanding of the forces shaping consumption patterns and levels is growing, especially within realms of individual disciplines. Ecological economics builds on the trans-disciplinary contributions in order to provide a richer, deeper understanding of consumption processes, so that future policies and other types of interventions would be more effective and would ultimately lead towards more sustainable consumption patterns and levels. As was mentioned above, consumption is driven by a number of factors, including economic forces and technological development, political settings and environmental issues, as well as sociological contexts and psychological determinants (Figure 1). These will be discussed in detail in this section.

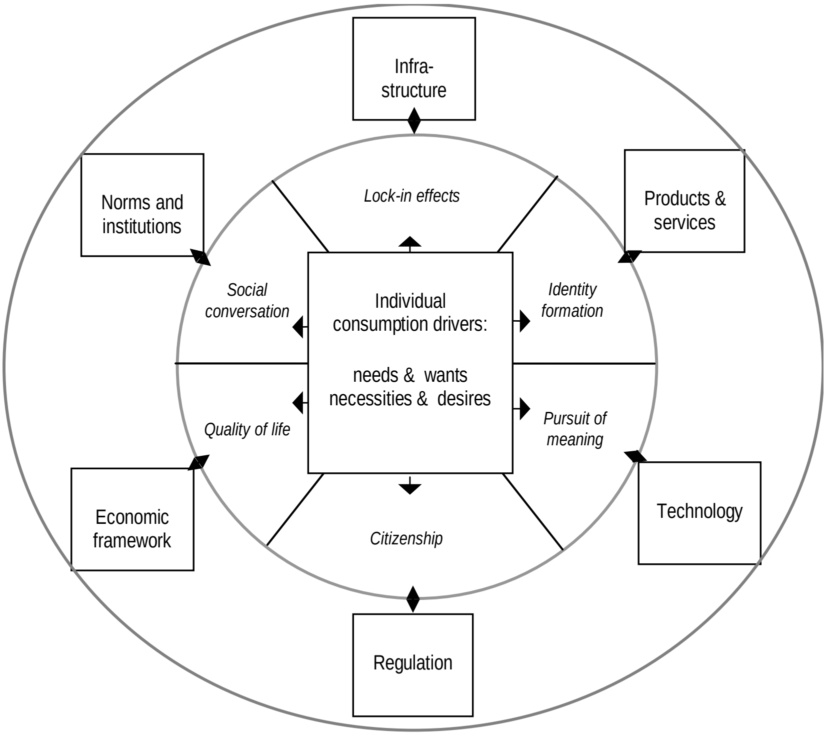


Figure 1. Factors affecting consumption.

4.1 Institutional Forces

Several institutional forces shape consumption patterns and levels. They include economic, technological and regulatory institutions, but also normative and cognitive institutions. The latter two types of institutions will be addressed in section 4.2, since their influence can be better understood from the individual level.

4.1.1 Economic Forces

Contemporary consumption levels and patterns are shaped by the entire history of industrial development. The industrial revolution of the 18th-19th centuries in developed countries solved the largest problem of the society at the time – the underproduction. Technological progress and increasing productivity stimulated by competition led to increased production volumes of goods, which needed to find their customers. Therefore, together with reduced product prices, strategies for stimulating consumption – advertising and marketing - were instigated and continue to develop until today. At the same time, there was and still is a clear tendency towards increasing incomes, leading to the growing purchasing power of individuals, which, being guarded by sovereignty principle, also leads to increasing consumption (Galbraith 1958). Since then the *perpetuum mobile* of production and consumption cycle has been gathering momentum. The increase in production volumes is associated with economic growth and the amount and volume of products with happiness. This perception is supported by prevailing economic and political institutions.

According to the neoclassical economics, the main prerequisite for consumption, assuming adequate supply and access to goods and services, is the availability of money (income) and leisure time. Other important influencing factors that affect consumption are prices and personal tastes. With its set of assumptions, the neoclassical theory is convenient for analysing how price elasticity, incomes and product substitution influence consumer behaviour. Income elasticity measures the responsiveness of the demanded product quantity to income change. If spending on a good does not grow proportionally to income, the good can be considered a necessity. On the other hand, if an increase in income induces more than proportional increase in expenditure, the good can be considered a luxury (Pearce 2000).

This market mechanism usually helps to move a good from the realm of luxury to the realm of affordable common good, which inevitably increases consumption levels of products. Practice shows that wealthy individuals are more likely to consume luxury items and are typically early adopters of a new technology absorbing initially high costs of innovation. As soon as the market for the early adopters (usually wealthier consumers) saturates, the manufacturers may choose to lower their profit margins or produce similar, but simpler and cheaper products in order to reach mass consumer (EC IST 2001). As technological improvements are made, the price of new technologies drops and demand increases allowing the suppliers to benefit from the economies of scale. As the production volumes increase, so does the competition, which further presses down the prices making products more affordable and available to masses.

An affordable good may even evolve into a necessity, which then influences consumption of other goods. This is especially true when institutional settings and infrastructures surrounding the product change in a way that promotes consumer dependency on the good. For example, in the USA cars were luxury goods until 1920s. However, as income grew, cars became an affordable good and later, facilitated by governmental strategies for infrastructure development and town planning, evolved into an absolute necessity. Increasing availability of cars increased consumer mobility, allowed time saving and access to more shopping places. In reality, however, time saving from car use is a relative benefit. With the increasing number of cars, time benefits usually shrink as roads become more clogged with traffic.

Looking at the demand side (individuals), the level of income depends on personal skills, amount of time for work and how well the skills are applied in producing value-added (i.e. labour productivity). Generally, higher incomes are possible along with the increasing workers' qualification and technological improvements, which provide higher labour and resource productivity and in turn lower manufacturing costs and final prices of products and services.

Since the levels of consumption are directly linked to the available budget, consumers are interested in increasing personal incomes, which is possible by raising productivity, i.e. producing more using less of the production factors (labour, capital, technology or resources) to maintain high level of output. The latter often leads to over-supply, which results in price drops that in turn contributes to higher levels of consumption.

Increasing income may also have another contribution to consumption in the form of savings if there is a surplus between income and expenditure. In this case, the excess of income is turned into savings, which eventually end up as investments, which further contribute to the economic growth and/or increase of productivity. Besides income, competition of financial institutions offering attractive credits to consumers is another important factor stimulating consumption. Moreover, there is an immense pressure from producers through the advertising media, which stimulates consumers to buy a new product before the old one is worn out or when it becomes unfashionable.

Along with increasing personal incomes, productivity growth should (in theory) reduce work time and provide more opportunities for leisure. The reduction of working hours does take place (e.g. in most of the countries in Western Europe), but at a rate that is far below productivity growth, and in some countries, e.g. Japan and USA, working hours tend to increase. One explanation can be found in the existing institutional structures and economic settings between the employers and employees. For example, working long hours has become a norm for many salaried workers in the USA and Japan, because this signals their loyalty to the employer and gives prospects for salary increase in the future based on seniority. In the “hours-invariant” forms of employment, an employee working overtime does not cost the employer extra, while the employee perceives it as a chance for the carrier advancement (Schor 1991). Furthermore, the combination of the existing financial settings in terms of the availability of credits combined with the pressure of advertising facilitate the creation of the “work-and-spend” culture (Schor 1999).

The relationship between the income and consumption (and finally – the environmental impact) is not always clear and is case-dependent, where leisure time and consumer choices are the key elements in determining environmental impacts. Unfortunately, labour has a tendency to become more expensive relative to other production factors, thus less-material commodities, such as services, become worse-off alternatives in comparison to products, so more of income is spent on material intensive products rather than less material services. One explanation is the changes of relative prices and product substitution. With the increasing labour costs, industrially produced products have lower prices than products and services produced with high labour input. Opportunity costs for buying a new or replacing an old product are lower than the costs of repair work, personal care, etc. This results in the development of self-service economy (Gershuny 1978). The self-service activities, such as dishwashing, laundry, cleaning and small repairs, are cheaper if performed by the households themselves rather than outsourced to the market.

At the time of industrial revolution and well into the beginning of the 20th century, increasing production volumes was a legitimate strategy for providing products and services since resources seemed to be abundant and assimilating capacity of the planet limitless (Princen 1999). Nowadays, however, understanding the limitations in terms of availability of resources and assimilating capacity, the economic model based on neoclassical premises seems to be somewhat outdated.

4.1.2 Technological Forces

Industrial revolution was one of the most important drivers for reaching current standards of living, lifestyles and consumption patterns. Industrialisation took place during several significant political and economic transformations in the North, such as colonisation, which secured the access to cheap natural resources from the South. Competition for the lowest price and reduction of manufacturing costs was accompanied by various structural changes on the macro-economic level, especially pricing of natural resources. On the demand side, increasing urbanisation and growing incomes facilitated increasing consumption. A giant leap in production output and consumption levels was possible due to the increase in work productivity through a series of innovations in production systems, such as division of labour and technological modernisation. These developments led to that one of the main characteristics of the current production system is mass production. In order to support the constant flow of resources and their effective processing, the production system emphasises process optimisation. To keep the demand for resource flow, the relatively short-life products are produced and sold on the market. To keep the speed of resource flows and market supply, new products with incremental improvements are sold and justified with arguments of progress in technology or safety, changes in fashion and others.

A number of technological forces gave an impetus to consumption, especially the invention of the general-purpose technologies, such as electricity, internal combustion engine, and communication technology (Grübler 1998; Røpke 1999a; Grübler, Nakicenovic et al. 2002). These technologies at first generated stand-alone products and applications, but later evolved into product systems, infrastructures, social practices, institutions and cultures. For example, the arrival of the car required creating road network, traffic police, road administrations, driving schools, etc. The arrival of television induced the creation of broadcasting systems, studios, advertisement and film industry.

Technological progress created a variety of household appliances that became more and more affordable with the increasing income. Most of the appliances were designed to save time and ease everyday household work. However, mechanisation of the household created empty time slots, but filled them with more and more diverse work. For example, while time was saved with the mechanical laundry, technological progress and increasing income allowed buying more diverse clothes, which require more time for separate washing, much more diverse washing programmes, various washing powders and drying techniques. Changes in socially accepted norms also play their role – the levels of cleanness half a century ago are no longer acceptable, so eventually people are “forced” to spend more time on laundry.

Information and communication technologies also gave a new impetus to consumerism. Internet became a powerful marketing channel, improved consumer access to information, created a perfect frictionless market with abundant information and close-to-zero transaction costs, which expanded markets in terms of time and space. The “one-click-shopping” e-commerce has dramatically changed the way we find, chose and buy products and services. These developments have a great environmental potential through optimisation of supply chains (Romm, Rosenfield et al. 1999). At the same time, increased availability of information about new products and services and lower prices of goods stemming from optimisations on the market has resulted in higher consumption and negative environmental impacts.

Although, the goal of mass production is to produce cheaper products with sufficient quality and to contribute to wealth creation, constraints and limitations imposed by natural resource availability, energy supplies and the absorbing capacity of nature have led to the obvious problem of making, using, and disposing of the increasing number of products. In addition to these environmental limitations, social limitations are also evident, as well as discrepancies between the increasing productive capacity and the decreasing absorptive market capacity leading to trade frictions. What is also interesting is that mass consumption patterns increase both the environmental burden and social costs of satisfying the same set of needs, as illustrated in an indicative 22 year study on the correlation between household consumption habits and associated environmental impacts (Garcia 1999).

Realisation of environmental impacts associated with production processes and products led to the development of more efficient technologies and more eco-efficient products. Products are becoming smaller, they use less energy and other consumables during their lifecycle, they may contain less toxic materials and overall have fewer environmental impacts. On the other hand, the number of products and the number of consumers is growing, negating eco-efficiency improvements in many areas. In addition to the direct effects of shopping and ownership, there are also indirect or so-called rebound effects that undermine intended results of technological improvements by creating additional needs and conditions that require people to consume more. For example, car was a product that was supposed to help people with mobility and to save time. Together with these outcomes, mobility also led to increased distances travelled because now people can live far away from places of work, from shopping malls and from their friends and family and can take their car to reach these places (Røpke 1999b). This example demonstrates that both products and especially infrastructures may lock consumers into specific consumption patterns (Otnes 1988; Sanne 2002). In a way, since infrastructural systems are so large and embedded into many aspects of everyday life, at a certain point of time they become a barrier to change even if consumer is interested in shifting to a different mode of consumption (Otnes 1988).

4.1.3 Political Forces

Since policy-makers use premises of the neoclassical economics and results of the neoclassical analysis as guidance, they support the goal of continuous economic growth and translate it into the growth that is largely based on material- and energy-intensive production and consumption. In order to keep GDP growth, many policies are developed that under-price natural resources relative to social costs. This takes places in the form of governmental subsidies, due to poorly defined property rights and due to market failure to incorporate the (negative) externalities linked to the use of natural resources (Arrow, Dasgupta et al. 2004).

When it comes to specific measures that affect consumption patterns of individuals, the role of policy makers has been to protect consumer sovereignty and to set standards and monitor features of products that may affect consumer health. Large efforts have been undertaken in many countries to help ensure that the consumer information on the market is correct and not misleading. Consumer agencies deal with issues of advertising and contract terms, consumer information and education, domestic finances, product safety, product quality, and in some cases also with product environmental impacts. However, the environmental and sustainability impacts of consumption have not yet been widely addressed in the work of consumer agencies (Mont and Dalhammar 2005).

Together with protecting consumer rights and interests and securing conditions for continuous economic growth, policy makers also have to make sure that business climate is favourable for companies in a specific market. This means that sometimes policies are devised that rather protect vested interests of various actors even though they are counterproductive to the good of the society at large or in the long term. This is especially visible in cases when policies and interventions of one country negatively affect another country or entire regions, or when decision-makers stay blind and deaf to business practices that take place elsewhere. For example, businesses are in continuous search for cheap labour and resources, which leads to the shift of production to developing countries where standards for working conditions are poor and salaries are minimal (Schor 2005). There are many cases when companies use the lacking legislation in developing countries to export waste and post-consumer products for some sort of recycling or even simply for final disposal.

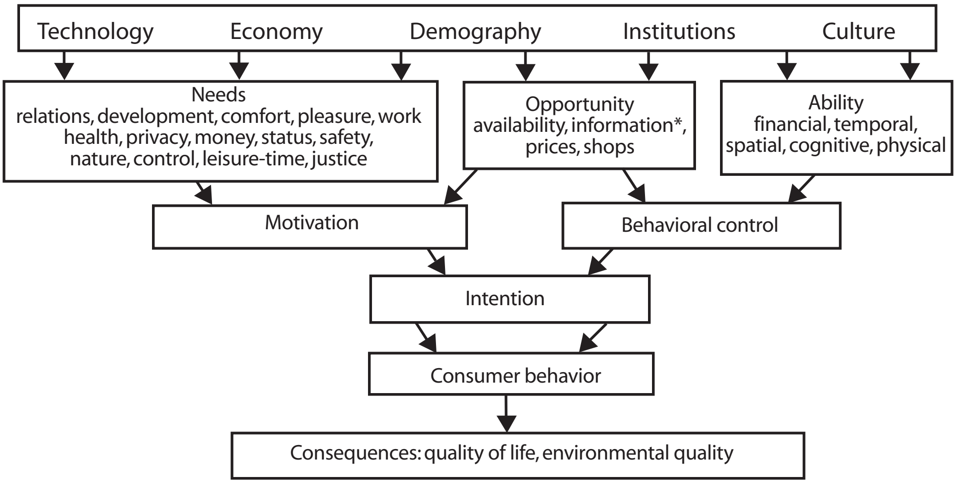
And finally at least in some cases, businesses and governments deliberately stimulate material- and energy-intensive consumption. One of examples is Americanisation of North East Asian consumption patterns, which results in creation of more energy intensive and environmentally and socially unsustainable lifestyles (Kasa 2003). In this example, the Americanisation leads to introduction of larger vehicles, less developed public transportation, reliance on processed food, high energy use, and consumption of more consumer durables and high volume of beef, which has much higher energy-intensity than traditional North East Asian fish-based diet (Durning 1992). Another example is the US response to the European development programme to help small banana growers in Grenada (Schor 2005). In these cases, instead of moderating production volumes of certain goods as response to reduced demand or improved environmental standards, US producers use political power to impose their own, not always sustainable, products or production methods to other countries. Often, international financial organisations, e.g. IMF, support such measures and directly contribute to unsustainable growth (Kasa 2003).

4.2 Individual Forces

Along with institutional driving forces of consumption outlined above, micro-level drivers of consumer decision-making processes are important for understanding current consumption patterns and levels. In order to understand individual driving forces, contributions from sociologists, psychologists and behavioural scientists are helpful and will be considered below. These disciplines offer much richer and more complex explanations of human behaviour, which can be useful in understanding how more sustainable consumption practices can be facilitated in the future.

4.2.1 Consumption as Fulfilment of Needs and Availability of Opportunities and Abilities

According to recent consumption theories, a large part of factors determining consumption are based on socio-psychological drivers (Røpke 1999a) and other external macro factors, such as demography, institutions and culture (Gatersleben and Vlek 1998). One of the models that brings together institutional and individual factors in explaining consumer behaviour is the so-called Needs-Opportunities-Abilities (NOA) model (Figure 22).



Source: (Gatersleben and Vlek 1998; Gatersleben 2001)

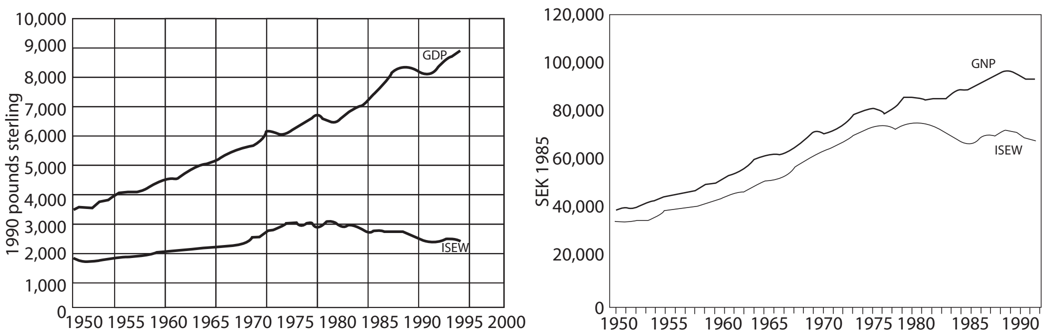
Figure 2. The NOA model of consumer behaviour.

According to the NOA model, motivations are determined by physiological and emotional needs, such as nutrition, safety, comfort, social positioning and interaction with society, status, etc. Behavioural control factors, on the other hand, are those that limit consumer motivations. The most common control factors are financial, temporal, informational, cognitive, or physical. Opportunities influence both motivational and behavioural control factors and are important in terms of availability or lack of information, time, money, locations and access. It is also argued that needs, opportunities and abilities are in turn influenced by macro factors, such as technologies, economic systems, demographics, institutional structures, social relationships, customs and cultures (Gatersleben and Vlek 1998; Gatersleben 2001).

A group of theories that try explaining consumer decision-making process comes from the fields of sociology and psychology, which focus on non-economic interpretations of consumerism and consumer behaviour. These theories make a distinction between the needs and wants of private consumers and to a large extent explain consumption patterns as a process of balancing needs and wants. The needs are usually viewed as pertinent to biological or bodily necessities, while wants are not biologically determined, but are acquired by learning. Once they are attained and their ability to give satisfaction has been learned, wants become habitual and at a certain level can be perceived as needs. Both needs and wants are to certain degree influenced by both psychological factors acquired over individual’s lifetime and by social drivers induced by social relationships, norms and culture.

4.2.2 Consumption in Pursuit of Happiness, Well-Being and Quality of Life

Throughout history, material goods and ownership has been associated with well-being, happiness and higher quality of life. However, socio-economic studies demonstrate that the link between material welfare and happiness is highly non-linear. With the constant progress and rise of welfare, there is a point, at which a luxury item becomes a necessity, e.g., tap water or lawn mowers, but the welfare effect of even greater consumption seems dubious, and, in many cases, even negative. Social indicators in many Western countries tend to show that social welfare does not continue to grow at the same rate as the economy (GDP) expands, and in many developed countries endless growth of material wealth and consumption has not made people much happier. According to Max-Neef (1995), every society has a period when economic growth brings about an improvement in the quality of life only until a certain point, beyond which further growth brings diminishing improvements. The World Value Survey demonstrates that up until $13,000 of annual income per person (in 1995 purchasing power parity) income and happiness tend to track well (WVS 2006), but after this level they start delinking. This welfare theory suggests that the per capita GDP, used as a traditional measure of welfare, is not an adequate indicator of happiness. One of the alternatives suggested instead is the Index of Sustainable Economic Welfare (ISEW). The index is intended to include different external factors that determine sustainable economic welfare (Daly and Cobb 1989). It adjusts personal consumer expenditure by taking into consideration a variety of social and environmental factors. A significant difference between ISEW and GDP can be illustrated by the two examples from the UK and Sweden (Figure 33).



Source: (Jackson and Marks 1999b) Source: (Jackson and Stymne 1996)

Figure 3. The Index of Sustainable Economic Welfare (ISEW) and GDP in the UK and Sweden, 1950–1996

These studies confirm the notion suggested already in 1954 by Maslow, that self-realisation and social acceptance are as important as the basic needs of food and shelter (Maslow 1954). In *The Psychology of Happiness*, Michael Argyle (1987) concludes that personal happiness is determined by the level of satisfaction with social life (marriage, family and friends), work and leisure. People themselves list the most desirable and satisfying activities to be: education, art, music, religion, basic scientific research, athletics, and social interactions (Meadows, Randers et al. 1972). A growing number of scholars suggest that material wealth fail to provide happiness. Rather the opposite - social life and leisure suffer from one's time and energy directed to obtaining more money and material goods (Elgin 1981; Wachtel 1983; Dominguez and Robin 1993; Durning 1995). Indeed, many consumers in modern industrialised counties feel trapped in a work-and-spend cycle trying to compensate excessive stress and widening social and cultural vacuum through increasing consumerism (Schor 1999).

4.2.3 Consumption in Pursuit of Symbolic Value and Social Conversation

People attach different meanings to the process of consumption and see consumption behaviour as part of identity construction. They purchase goods and services for their qualities and functions, as well as for their symbolic or identity value (Bauman 1990). Consumers create themselves and are created by products, services, and experiences. Four different types of meanings that people attach to products can be distinguished: utilitarian meaning (perceived usefulness of a product in its ability to perform functional tasks), hedonic meaning (specific feelings the products evoke or facilitate), sacred products that are very important to people, and social meanings (products and services are seen as "media for interpersonal communication" and for statements about people's positions and statuses in social groups) (Engel, Blackwell et al. 1995). Some authors show that for many consumers today, the symbolic value of products has become even more important than the physical aspects of goods (Leiss 1983), and traditional marketing strategies further cultivate this trend. Other authors indicate that the value of goods and services is not inherent, but depends on the activities they are used for (Lunt and Livingstone 1992). For example, one of the reasons for purchases is to increase people's "personal availability"—to free/obtain more time to be engaged in other activities, which are more pleasant, and which have the potential to increase the social status of people, such as organising a party, recreational activities, and so forth (Henry 2002; Williams 2002). On the other hand, the personal availability reason, if true, may have direct implications for associated environmental impact that arises from alternative tasks or activities people become involved in when some time or money is freed.

Stratification and social conversation are important parts of people's identities and our lifestyles. Through social conversation people define and create their status in society and differentiate themselves from others. Achieving a certain status in a social group stimulates consumption of the so-called "status goods" leading to conspicuous consumption (Veblen 1902). Therefore, at the heart of the sociological view are studies on the role played by goods in marking the distinction between different social groups and classes and strengthening one's identity within the group. Following this line of research, Leibenstein (1950) suggested that since the desire of people to consume certain goods is rooted in the desire to be accepted by a social group, people can be trapped by the desire to adopt to the most accepted or prestigious way of living. This mechanism implies that if the prestigious way of living is unsustainable, it might be difficult to change it, as non-members will always struggle for being accepted into the prestigious circle. The contrary is also true: if it is possible to make prestigious lifestyle more sustainable, then it will be easier to solicit more followers into it.

According to Røpke's (1999a) interpretation of Douglas and Isherwood's (1980) theory, goods are used both as means of interacting with society and the world at large, as well as for making a personal differentiation in society. An interesting conflict has been pointed out by Baudrillard (1998): while people need to buy the same items because they want to belong to a certain stereotypic group ("assisted" by heavy marketing), they also want to buy goods that allow them to feel different and unique. Individuals have a strong desire to differentiate themselves from a social group, to create and confirm to a unique style and in this way manifest one's identity. These desires create demand for value-added goods and services, such as unique handcrafted or local products. A general trend towards individualisation is also seen as reinforcing consumption demand. Individualisation implies that people are freed from social and traditional bonds, meaning that their identities are no longer defined by a community or traditional roles, but rather by the increasing number of owned goods, which serve as messages about their identity (Halkier 1998).

Serving the growth task, prevailing economic and political institutions make people believe that the pursuit of self-interest in the quest for higher material prosperity is the expected behaviour (Kilbourne, Beckmann et al. 2001) or even a patriotic duty (Princen 1999). Thus there is a trend of formalisation and institutionalisation, where people are shaped by the society and fashion and are pressed into conformity of the accepted social norms and lifestyles. This could be exemplified by consumption of certain cuisines or the use of different dress codes to differentiate special occasions according to shared understanding in society or to conform to accepted rules and norms. These desires increase the demand and supply of goods from all over the world, which creates something unique for the modern world – the global culture and the global consumer class.

5. Responding to Complexity: Towards Sustainable Consumption

In the previous sections we have outlined the main consumption related problems and considered how the assumptions of the neoclassical economics limit the possibility of the adequate response to them. To summarise, the neoclassical economics considers that individuals' preferences are exogenous and that people demonstrate consistent behaviour in different situations. This means that consumer behaviour is easily predictable and entirely depends on income and prices. In terms of policy implications this means that consumer behaviour can be easily modified by market based mechanisms that affect prices and incomes. And surely, economic policy aimed at changing consumption patterns has a long history. Products that have negative health effects or large environmental impacts are often banned or their use is restricted, e.g. drugs and the use of chemicals in certain types of products. Economic instruments that internalise external costs, primarily taxes, are widely used to affect consumer behaviour, e.g. tax on tobacco and gasoline. Environmental policy based on the neoclassical economics also largely believes in provision of incentives, but does not accept that it is possible to change people's preferences (Norton, Costanza et al. 1998).

We have also described how the ecological economics attempts to address the limitations of the neoclassical economics and how, with contributions from sociological and psychological studies, it provides a more comprehensive understanding of consumer behaviour. In the following sections, the ideas from the modern consumption theories are used to discuss the needed changes at three levels: people, places and processes. The process of change follows the 5Es approach[3] that includes five elements: enabling, engaging, exemplifying, encouraging and envisioning, which was elaborated from the 4Es strategy of the Sustainable Consumption Roundtable in the UK.

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[3] Developed from the 4E approach of the (SCR 2006)

5.1 People

In modern consumption theories, which are largely built on the fields of psychology and sociology, consumer behaviour is seen as much more complicated than simply rational response to price signals or to the availability of technical solutions. Society is also seen not only as a market system, but as an intricate network of actors and institutions. Social institutions, public groups and individual behaviours mutually reinforce each other and shape the development of society and people. Beside price and income, consumer choice is influenced by functional, conditional, social, emotional and epistemic values of products. This means that these values need to be incorporated and alternatives suggested in shaping more sustainable behaviours and lifestyles.

Alternative views are already available from ecological economics, sociological and psychological research that can provide a starting point for identifying relevant environmental policies towards more sustainable lifestyles. For example, the desire to belong to a certain social class is nowadays to a large extent satisfied by the level of material possessions. The same desire can instead be fulfilled through less environmentally intensive services, such as through access to education, cultural events or exclusive sport activities. Research also confirms that in the last 40 years consumer expenditure can be allocated to an attempt to satisfy psychological and social aspirations, rather than material ones (Jackson and Marks 1999a). On the other hand, studies demonstrate that many people engage in conspicuous consumption in order to substitute for the lost sense of community (Hacker 1967). Traditional communities and neighbours are being substituted by virtual societies and groups of consumers sharing the same brand name. All this demonstrates that people even in materialistic societies are still social beings looking for the sense of belongingness, for communication, acceptance, confirmation and appreciation. Thus, alternative systems for supporting sustainable lifestyles could rely at least to the same extent on social processes and less-material services, as they do currently rely on materialistic values and provision of products to “satisfy every need”.

However, a distinction should be made between a consumer as an individual consumer and as citizen-consumer. While citizen-consumer is willing to act for the public benefit, individual consumer typically favours private benefits. Some studies demonstrate that it is still a small percentage of population who have adequate knowledge and are sufficiently engaged in environmental issues so that they are willing to change their everyday practices towards more sustainable ones (Reisch and Røpke 2005).

When developing visions of more sustainable society, consideration should be given to the fact that as individuals we all have and play different roles depending on the context we are in. We are parents, colleagues, consumers, children and play different other social roles that are prescribed, expected or assumed by us, but which all affect our behaviour and influence our choices. In order to understand better how to influence the choices of people in different roles, collective and shared processes and contexts where behaviour takes place need to be considered (Georg 1999). Following that premise, it is also suggested that strategies and policies need to be developed that affect consumption behaviour not only at individual, but most importantly at collective level (Jackson 2005).

5.2 Places

As noted above, people assume different roles following the context in which they operate. Depending on the context various strategies can be used to communicate with people in different roles. For example, chances of reaching citizen-consumers are perhaps larger at work, where people often think in collective terms or about the public good. School children have proven to be advocates of changes in parents' behaviour after they have learned at school about, e.g. the importance and practical side of waste separation. As Røpke argues, different places have very different levels of embedding in socio-technical systems and a diversity of routines is associated with the variety of places (Røpke 2005). Therefore, approaches that are useful for addressing people at home, may not be efficient when addressing people at work and visa versa.

Household constitutes the primary social unit in the majority of countries and therefore it is important to understand the dynamics of individual behaviour in household in relation to sustainable consumption. The household is a place where consumers spend most of their time and this is where the large part of behaviour is shaped. Recent studies of environmental impacts have already taken the household as the primary unit of analysis and highlighted the large differences between households in their consumption patterns and associated environmental impacts (EEA 2005). However there seems to be a need to better understand environmental implications of behaviour of various members of the household, also depending on the roles people assume in different stages of their lives.

One of the places that have been overlooked in the political and economic context is the community – a place where we live and communicate with other people. Large potential exists at community level to develop alternative consumption models and social enterprises (Manzini and Jegou 2006). It is demonstrated that communities have the potential to respond to people's deepest desires for communication, social contact and belongingness. In addition to the social benefits, alternative community-based non-commoditised goods and cooperative activities might be more environmentally sustainable. However, it is also noted that such community initiatives are to a large extent marginalised or not taken into consideration in policy development (Leyshon, Lee et al. 2003), which prevents these potential 'islands of sustainability' from scaling-up.

Finally, it is important to look at places because they represent and reflect the entire diversity of cultures, norms and religions that shape consumer behaviour and lifestyles. Such contextual factors need to be considered when policies and strategies for sustainable lifestyles are proposed. For example, while promoting the use of bicycles in Scandinavia by men and women could be the most natural strategy to pursue, the same approach may not be acceptable in a different cultural context, e.g. for women in some Muslim countries. A different approach would need to be found to reduce transportation related environmental impacts in the second case. Thus, sustainable consumption strategies should be linked to specific places that are associated with certain cultures, norms and routines.

5.3 Processes

Changing people's behaviour is a daunting task since it both influences and is embedded into social and institutional contexts and is guided by individuals themselves, as well as by their immediate and distant surroundings. It has been suggested that in order to initiate the change process people need to be enabled to make more sustainable consumption choices through provision of more sustainable products, infrastructures and services. People also need to be encouraged by various means, not least by marketing and advertising, but also by economic instruments to make more sustainable choices. They also need to be engaged in on-going or planned efforts towards more sustainable consumption patterns and levels through community actions or participation in activities of various societies, NGOs or social enterprises and clubs. And finally people need to see examples from other people and societal actors, who already live more sustainably and clearly see what kinds of positive impacts the changed behaviour and choices lead to. The process of change discussed above is based on the 4Es approach developed by the Sustainable Consumption Roundtable in the UK that includes: enabling, engaging, exemplifying, encouraging (SCR 2006a), but is extended here to include the 5th important element – envisioning of more sustainable futures and creating a shared understanding in the society of preferred states of the world.

5.3.1 Enable

Public policy at all levels plays a key role in shaping and enabling consumer decisions by developing the regulatory, economic and institutional frameworks within which all actors operate. Industry develops business models in response to the set rules of the game and consumers respond to the supply of the market and to the regulatory settings that apply to them, e.g. ecolabelling or taxes. Public policy influences markets through permits and licenses, through demands on disclosure of environmental and consumer information, through taxation and fees. Choices of consumers are very much driven or subconsciously made because of the existing and embedded regulatory and economic frameworks.

In addition to regulatory and economic frameworks, another area that can enable changes in consumption patterns is infrastructure. People often find that their choices and behaviours are restricted by the existing infrastructure and even if they are willing to change their behaviour towards more sustainable choices, infrastructure and the choice of products and services may simply not support this change. Therefore one of the most important contributions of the government and public policy is to ensure that the development of infrastructure supports more sustainable consumption choices and lifestyles of people, that they are not locked-in in unsustainable behaviours, but instead are enabled and assisted by available options.

Mobility is the area where existing infrastructure plays the most important role in enabling certain consumer behaviour. Availability and efficiency of a well-functioning public transportation system in some countries is reflected in consumer choices for not owning a car or for becoming a member of a car sharing cooperative to satisfy the need for travel on weekends. In other countries, e.g. USA, infrastructure developed for private car use deprives customers from the option of using public transportation system and locks them into exclusively using cars. Urban sprawl also contributes to increasing car dependency. Since small food shops in city centres are disappearing and are replaced by large one-stop-shop commercial malls situated at city outskirts, shopping for food becomes a weekly or bi-weekly activity, for which people nowadays need cars. Developing infrastructure and services that would allow satisfaction of people needs, e.g. food shopping, without the need to use a car would drastically reduce environmental impacts, improve social climate and contribute to increased quality of life (in small shops people do communicate with each other, which they do not tend to do in the efficient layouts of commercial superstores).

Changing the rules of the game or setting sustainability goals at policy level would also affect consumer behaviour and patterns and clearly demonstrate policy priorities to citizens. Some authors therefore argue that social policy should include the goal of increasing quality of life, and not just attainment of economic prosperity (Layard 2005). One example comes from the kingdom of Bhutan that has already set the policy goal to be “gross national happiness” (Bond 2003). Setting this kind of goals will require investments into creating and developing opportunities for people to satisfy their needs through engagement of the most suitable means (not only material means, but by involving built, human, social or natural capitals), which will ultimately lead to the best fulfilment of human needs and provide the highest quality of life (Costanza, Fisher et al. 2007).

Policy can also play a role in enabling more sustainable consumption patterns and levels by shaping social norms. As was mentioned elsewhere, social norms do evolve all the time subject to collective behaviour, policies and institutional developments. Since the prevailing belief in the society is “the more the better”, as consequence we see policies that support increasing GDP at any price and it is definitely the price of the environment that has been paid so far or the GDP increase. If instead, the goal of social policies was increasing the quality of life, policies would perhaps focus more on education, health and time availability for recreational, family and community activities (Easterlin 2003).

5.3.2 Encourage

One way to involve people into changing their everyday practices is to encourage their thinking about long-term and global implications of their decisions. This means to connect producers and consumers: to provide consumers with knowledge about where and in what conditions products are manufactured, what kind of environmental and social impacts are associated with production, transportation and product disposal (Mont and Bleischwitz 2007). The problem with providing more information to consumers is of course the cognitive and time limitations of people. However, in some instances evidence is available on people actually taking the time and engaging in more conscious decision-making process especially if the consequences of their decisions are visualised, either by pictures of trees being cut or by information about child labour being involved in producing goods. As Gintis (2000) argues, in various contexts people act differently, but in social groups people have clear ideas about justice and fairness, cooperation and punishment of inadequate behaviours. Thus, public awareness raising campaigns that inform consumers about negative consequences of certain products, activities or consumption patterns can create a feeling of social responsibility not only for their own actions, but for the actions of others. In this way, people may engage in the change process and participate in the evolution of new social norms. Changing consumer preferences and behaviours could be done so that they would not feel “*... deprived and unhappy ...” but “... enlightened and happy after being educated into the joys …*” (Norton, Costanza et al. 1998): 203.

The often heard suggestion that western lifestyles should be redefined into more sustainable alternatives can be supplemented with promoting the importance of social relations and spare time, instead of being concerned with reducing their material and energy intensity alone. In this case, shifting to more sustainable options would not only reduce environmental impacts and thus improve the quality of life in the long run, but would also improve the level of satisfaction with life in the short term. Indeed, evaluation of community-based behaviour change projects in the UK report improvements not only in the environmental performance, but in the social sphere, including social cohesion, building new institutions and neighbourliness, bringing people together who share similar values and goals (SCR 2006b). Examples of such projects that encourage people participation in changing their lifestyles range from "green gym", "green consumer clubs" and reduction of carbon dependency to a wide range of educational and skill-buildings activities, as well as in-house audits and discussion groups where personal change towards more sustainable lifestyles is measured and discussed. Some of these projects also reach out to individuals of different incomes and with different physical abilities. They, for example, involve people with disabilities and teach them in an interactive manner how to calculate their annual car mileage and convert it to carbon emissions.

Whichever approach is taken, encouraging people should also be connected to increasing the perceived value to people – either through saving money or time, increasing their social value or the level of comfort or providing them with other types of gratification. One example that demonstrates how people can be encouraged to more willingly participate in waste separation activities is to allow individuals, who separate organic waste, use the final product – compost – for their home and garden use or to use the biogas from composting as fuel for cars (Lundberg 2007).

5.3.3 Engage

Engaging people in the shift towards more sustainable consumption should be based on the deep understanding of human nature and that well-being always involves both doing and being. Being able to engage in activities that bring happiness and meaning to life is a freedom, a matter of interaction and availability of choices. Interactive approaches, e.g. dialogues and discussions, help people understand the issues at stake, evaluate alternatives, discuss the bottlenecks that they encounter on the way and find solutions together. Michaelis and Jackson (2003) argue that "*there does seem to be more potential for a shift in consumption patterns if people are engaged in a community dialogue than if they simply reflect on their own lives*".

Several social movements provide examples of how people become engaged in changing and recreating their lifestyles in an interactive process and for various reasons. Some people seek social contact, while others re-evaluate values and priorities and search for more fulfilling and healthy lives. Whatever the reason for change is, examples are available and provide a good starting point for more concerted effort. Those people who seek social contact, become members of *co-housing communities* or *eco-villages*. In both cases, people share common facilities, usually engage in common activities, such as gardening or hobbies, they exchange services with each other and report much higher level of satisfaction with life, e.g. (Mulder, Costanza et al. 2006). Eco-villages often follow similar principles, but the main underlying idea is the environmental sustainability of their lifestyles. Therefore it is typical that people build houses according to environmental criteria and organise car sharing cooperatives in eco-villages. It is demonstrated that the footprint of these settlements is typically lower than the nearby communities of similar size (Widén 1998). People also engage in exchange of services in a more formalised manner. Some communities where it is practiced develop *Local Exchange Trading Schemes* (LETS) - local community-based networks, where people earn LETS credits by providing a service and can buy services of other people with these LETS credits, e.g. (Briceno and Stagl 2004).

Another example is lifestyles based on *self-sufficiency* (Princen 1997; Krongkaew 2003; Princen 2003; Princen 2005). Despite its name, the idea becomes appealing to people because it does not necessarily imply self-restriction and Spartan lifestyle, but rather calls for identifying other than materialistic values in life and enjoying life to the fullest, by taking the time and finding pleasure in common activities and communication. The slow living movement, simple living and slow food movement (Segal 2003), (Center for a New American Dream 2001) exemplify this concept. Besides these, sufficiency solutions also include common practices of renting out cottages to several people throughout a year, hotel sharing programmes or car sharing cooperatives organised by people or provided by commercial organisations.

For these grass-root initiatives to scale-up and get institutionalised, they need support of local authorities and local communities that would help engage people in similar activities. Even support and encouragement from public policy recognising the importance of such contributions to changing lifestyles towards more sustainable alternatives is important.

5.3.4 Exemplify

Providing information about examples of more sustainable consumption and sustainable lifestyles is one of the most important contributions to the change process. Examples can come from individual and community levels, as well as from governmental level, since government is a large consumer and can therefore lead by example. Examples are important because they present ongoing activities that combine all the steps in the change process: envision, enable, encourage and engage. They provide ideas and learning opportunity to other people and stimulate others to follow.

The amount of examples of small and large initiatives towards sustainable lifestyles and sustainable consumption is growing. There are examples where people get educated about impacts of their activities and are taught new patterns of behaviour that bring about higher quality of life with less environmental impact. For example, Norton, Costanza et al. (1998) give an example of sustainable consumption where people who enjoy high impact lawns in an arid environment are educated about the benefits and practicalities of low-impact gardening. The authors promote this idea because people can enjoy both their gardens and the fact that they do not pose threat to the local habitat. There are also examples of people changing their consumption routines once they have learned about eco-labels and as eco-labelled products become available in shops and are supported by marketing campaign (Sammer and Wüstenhagen 2006). Another example is about how commuting to work can be reduced. An on-going project develops a web-based service that helps people in the Netherlands to reduce home-work travel by switching jobs with someone who has a similar job, but closer to their home (Maase and Dekker 2007). There are also examples of people who support local food systems and prefer paying premium price to local farmers at farmers' markets instead of shopping in large retail chains (Nilsson and Hansson 2006).

Examples are numerous and could be found in all the main domains of household consumption: food, mobility, housing, tourism and waste management. Although their role is extremely important – to exemplify and encourage others for action, it is vital that support to these initiatives is offered and they become mainstream activities, rather than stand-alone initiatives.

5.3.5 Envision

Envisioning is an important exercise if more sustainable futures are to become a reality one day. Envisioning should aim at creating shared pictures of preferred states of the world and embedding them into the very societal fabric, including institutional, regulatory and economic frameworks. The challenge really is to design a vision that is both desirable for people from different societies with different level of economic development and that is environmentally sustainable in the short and long run. Without knowing where we want to go it is impossible to get there. Plus, the challenge is to balance our desires with what we have at hand – all the resources and natural capital that the earth can offer us, and make sure that we do not undermine the very basis of our society – the natural ecosystems. The goals of the shared vision, according to Meadows, (1996) should be crystal clear, while path should be as flexible and diversified as possible. The vision itself should be evolving and therefore the process of creating shared visions is at least as important as the visions themselves (Costanza 2000). "*Probably the most challenging task facing humanity today is the creation of a shared vision of a sustainable and desirable society, one that can provide permanent prosperity within the biophysical constraints of the real world in a way that is fair and equitable to all of humanity, to other species, and to future generations. This vision does not now exist, although the seeds are there*" (Costanza 2000). It is therefore paramount to start building up on these seeds at all levels: individual, community, regions and nations, and bring these visions into a coherent picture of a more sustainable future. The premises and ideas of ecological economics in this work are of great value and importance.

6. CONCLUSION

In a finite world, developing a sustainable society requires deep understanding of processes underlying systems of production and consumption. Until recently, the neoclassical economics has provided solid basis for building economic system and securing economic growth. However, growing knowledge about externalities associated with the current market systems and the need to understand the roots of these problems called attention to the main assumptions, on which the neoclassical economics rests. It became clear that the nature of the problems we face is systemic, since problems are found at the interface of economy and biophysical systems supporting our society. The ecological economics seems to be better equipped for addressing complex and interlinked problems of the real world than the neoclassical economics. It has trans-disciplinary basis since in order to move towards more sustainable societies, a systems perspective is needed that cannot be provided by a single discipline. It is also better suited for identifying strategies towards more sustainable consumption and production, since the largest challenge seems to lie not in the technological development and in the more eco-efficient management of resources, but in much more subtle processes that shape people preferences, perceptions and consumer behaviour. It is therefore as important to have a more sustainable management of resources, as it is to better understand and manage ourselves (Dodds 1997).

The contributions from sociology and psychology provide invaluable insights into the nature of human beings and into the nature and dynamics of social processes. In this chapter we discuss the main institutional forces driving consumption, including regulatory settings, economic and technological forces. Although they explain quite well the historical development, they provide little advice on how future economic systems may look like, since future society needs to rely on a quite different natural system – with strained natural resources and much reduced assimilating capacity. In these new circumstances, ecological economics will need to help shape and develop institutions and attitudes that recognise biophysical limits. It is therefore important to understand why resources and environmental services are important to human well-being and how economic institutions may be structured to make the best use of them without jeopardising the long-term sustainability of the planet. This will require deeper understanding of how perception of well-being is formed and consumer preferences are constructed. Reaching long-term sustainability will also perhaps require re-evaluation of our desires and especially the ways of meeting these desires. Therefore, more attention should be allocated to developing better understanding about the human behaviour in the economic system and beyond, to identifying the fundamental human needs and the social construction of preferences, to understanding individuals as part of the community and of the specific cultural context. All this knowledge will help understand how people behaviour could be influences in the direction of more sustainable choices, not limiting their role to being just consumers, but recognising their multiple roles and preferences in different contexts and acknowledging their desire and right for fulfilling and happy lives.

REFERENCES

Argyle, M. (1987). *The Psychology of Happiness*. New York, Methuen and Company.

Arrow, K., P. Dasgupta and L. Goulder (2004). "Are We Consuming Too Much?" *Journal of Economic Perspectives* 18(3): 147-173.

Azar, O. H. (2004). "What sustains social norms and how they evolve? The case of tipping." *Journal of Economic Behavior & Organization* 54(1): 49-64.

Baudrillard, J. (1998). *The consumer society: myths and structures*. London, Sage Publications.

Bauman, Z. (1990). *Thinking Sociologically*. Oxford, Blackwell.

Becker, G., S. (1965). "A Theory of the Allocation of Time." *Economic Journal* 75(299): 493-517.

Bianchi, M. (1998). *Active Consumer: Novelty & Surprise in Consumer Choice*. Florence, KY, USA, Routledge.

Bond, M. (2003). "The pursuit of happiness." *New Scientist*(180): 440–443.

Briceno, T. and S. Stagl (2004). Examining the Social Aspect of Sustainable Consumption. International workshop "Driving forces of and barriers to sustainable consumption", University of Leeds, UK.

Center for a New American Dream (2001). Homepage of the Center for a New American Dream.

Costanza, R. (1996). "Ecological Economics: Reintegrating the Study of Humans and Nature." *Ecological Applications* 6(4): 978-990.

Costanza, R. (2000). "Visions of alternative (unpredictable) futures and their use in policy analysis." *Conservation Ecology* 4(1): 5.

Costanza, R., B. Fisher, S. Ali, C. Beer, L. Bond, R. Boumans, N. L. Danigelis, J. Dickinson, C. Elliott, J. Farley, D. E. Gayer, L. M. Glenn, T. Hudspeth, D. Mahoney, L. McCahill, B. McIntosh, B. Reed, S. A. T. Rizvi, D. M. Rizzo, T. Simpatico and R. Snapp (2007). "Quality of life: An approach integrating opportunities, human needs, and subjective well-being." *Ecological Economics* 61(2-3): 267-276.

Daly, H. (1996). *Beyond Growth: The Economics of Sustainable Development*. Boston.

Daly, H. (2003). *Ecological Economics: The Concept of Scale and Its Relation to Allocation, Distribution, and Uneconomic Growth*, University of Maryland**:** 26.

Daly, H. and J. Cobb (1989). *For the Common Good - Redirecting the Economy Towards Community, the Environment and Sustainable Development*. London, Green Print.

Daly, H. E. (1986). "Thermodynamic and economic concepts as related to resource-use policies: comment." *Land Economics* 62(3): 319-322.

Daly, H. E. (1987). "The economic growth debate: What some economists have learned but many have not." *Journal of Environmental Economics and Management* 14(4): 323-336.

Dodds, S. (1997). "Towards a 'science of sustainability': Improving the way ecological economics understands human well-being." *Ecological Economics* 23(2): 95-111.

Dominguez, J. and V. Robin (1993). *Your Money or Your Life: Transforming Your Relationship with Money and Achieving Financial Independence*. New York, Penguin Books.

Douglas, M. and B. Isherwood (1980). *The World of Goods. Towards an Anthropology of Consumption*. London, Penguin Books.

Durning, A. (1992). *How Much is Enough? The Consumer Society and the Future of the Earth*. New York, W. W. Norton & Company.

Durning, A. T. (1995). Are we happy yet? Ecopsychology: Restoring the Earth, Healing the Mind. T. Roszak, M. E. Gomes and A. D. Kanner. San Francisco, Sierra Club Books**:** 68-76.

Easterlin, R. (2003). "Explaining happiness." *Proceedings of the National Academy of Sciences*(100): 11176–11183.

Easterlin, R. A. (1973). "Does Money Buy Happiness?" *Public Interest*(3): 3-10.

EC IST (2001). *State of the Art Survey - Consumption Theories (Achieving Sustainability by using Substitutive Information Society Technology (ASSIST)*. Brussels, European Commission**:** 104.

EEA (2005). *Household consumption and the environment*. Copenhagen, European Environmental Agency**:** 72.

EEA (2006). *Using the market for cost-effective environmental policy. Market-based instruments in Europe*. Copenhagen, European Environmental Agency**:** 48.

Elgin, D. (1981). *Voluntary Simplicity: Toward a Way of Life that is Outwardly Simple, Inwardly Rich*. New York, William Morrow & Co.

Engel, J. F., R. D. Blackwell and P. W. Miniard (1995). *Consumer Behavior*. Forth Worth, Dryden Press.

Fine, B. (1997). "Playing the Consumption Game." *Consumption, Markets and Culture* 1(1): 7-30.

Friedman, M. (1957). On the methodology of positive economics. Essays in Positive Economics. Chicago, University of Chicago Press.

Galbraith, J. K. (1958). *The Affluent Society*. Boston, Houghton Mifflin.

Garcia, E. (1999). Household Consumption, Status, and Sustainability. "Nature, Society and History". Long Term Dynamics of Social Metabolism, "Kursalon im Stadtpark", Vienna, Austria.

Gatersleben, B. (2001). "Household Consumption and quality of life: the acceptability of sustainable consumption patterns and consumer policy strategies." *International Journal of Environment and Pollution* 15(2): 200-216.

Gatersleben, B. and C. Vlek (1998). Household Consumption, Quality of Life and Environmental Impacts: A Psychological Perspective and Empirical Study. Green Households? T. S. Uiterkamp. London, Earthscan**:** 141-183.

Georg, S. (1999). "The social shaping of household consumption." *Ecological Economics* 28(3): 455-466.

Gershuny, J. (1978). *After Industrial Society?: The Emerging Self-Service Economy*. London, UK, Macmillan.

Gintis, H. (2000). "Beyond Homo economicus: evidence from experimental economics." *Ecological Economics* 35(3): 311-322.

Grübler, A. (1998). *Technology and Global Change*. Cambridge, Cambridge University Press.

Grübler, A., N. Nakicenovic and D. W. Nordhaus, Eds. (2002). *Technological Change and the Environment*. Resources for the Future. Washington, DC, USA, International Institute for Applied Systems Analysis, Laxenburg, Austria.

Hacker, A. (1967). "A defence (or at least an explanation) of American materialism." *Sales Management*(March): 31-33.

Halkier, B. (1998). Miljøhensyn i Forbrug. Erfaringer og Forhandlinger i Ambivalente Hverdagsliv [Environmental consideration in consumption. Experiences and negotiations in ambivalent everyday lives]. Institut for Miljø, Teknologi og Samfund. Roskilde, Roskilde Universitets.

Harris, J., M. (1997). Consumption and the Environment. The Consumer Society. Series: Frontier Issues in Economic Thought. D. Kiron. Washington, D.C.; Cavelo, California, Island Press. 2**:** 350.

Hay, P. (2002). *Main Currents in Western Environmental Thought*. Bloomington, IN., Indiana University Press.

Henry, P. (2002). "Systematic variation in purchase orientations across social classes." *Journal of Consumer Marketing* 19(5): 424-438.

Jackson, T. (2005). *Motivating sustainable consumption*. Surrey, Centre for Environmental Strategy, University of Surrey**:** 170.

Jackson, T. and N. Marks (1999a). "Consumption, Sustainable Welfare and Human Needs - with Reference to UK Expenditure Patterns between 1954 and 1994." *Ecological Economics* 28(3): 421-441.

Jackson, T. and N. Marks (1999b). "Consumption, sustainable welfare, and human needs - with reference to UK expenditures patterns between 1954 and 1994." *Ecological Economics* 28(3): 421-441.

Jackson, T. and S. Stymne (1996). *Sustainable Economic Welfare in Sweden—a pilot index 1950–1992*. Stockholm, Stockholm Environment Institute**:** 56.

Kasa, S. (2003). "US Trade Policy Power and Sustainable Consumption: Beef and Cars in North East Asia." *Journal of Consumer Policy* 26(1): 75-100.

Kilbourne, W. E., S. C. Beckmann and A. Lewis (2001). "A multinational examination of the role of the dominant social paradigm in environmental attitudes of university students." *Environment & Behavior* 33(2): 209-229.

Kletzan, D., A. Köppl, K. Kratena and M. Wüger (2002). Economic Modelling Sustainable Consumption Patterns for Mobility and Heating. Life-Approaches to Sustainable Consumption -- Workshop Proceedings, Laxenburg, Austria, International Institute for Applied Systems Analysis.

Krongkaew, M. (2003). The Philosophy of Sufficiency Economy. Feature.

Lancaster, K. (1971). *Consumer Demand: A New Approach (Study in Economics: No. 5)*. New York, Columbia University Press.

Layard, R. (2005). *Happiness: lessons from a new science*. London, Allen Lane.

Leibenstein, H. (1950). "Bandwagon, Snob, and Veblen Effects in the Theory of Consumers' Demand." *Quarterly Journal of Economics* 44: 183-207.

Leiss, W. (1983). "The Icons of Marketplace." *Theory, Culture & Society* 1(3): 10-21.

Leyshon, A., R. Lee and C. Williams, Eds. (2003). *Alternative Economic Spaces*. London.

Lundberg, G. (2007). *Practical communication on waste - Ideas and conclusions from the RECO project regarding awareness raising and communication to change behaviour*. Stockholm, KSL**:** 21.

Lunt, P. K. and S. M. Livingstone (1992). *Mass Consumption and Personal Identity. Everyday Economic Experience*. Buckingham, Open University Press.

Maase, S. J. F. M. and J. W. A. Dekker (2007). Work closer to Home. A web-based service to reduce home-work travel. SCORE 2, Paris.

Manzini, E. and F. Jegou (2006). Creative communities and sustainable lifestyles. SCORE, Wuppertal.

Maslow, A. (1954). *Motivation and Personality*. New York, Harper and Row.

Max-Neef, M. (1995). "Economic growth and quality of life: a threshold hypothesis." *Ecological Economics* 15(2): 115-118.

Meadows, D. (1996). Envisioning a sustainable world. Getting down to earth: practical applications of ecologial economics. R. Costanza, O. Segura and J. Martinez-Alier. Washington, D.C., Island Press**:** 117-126.

Meadows, D., J. Randers and W. W. Behrens III (1972). *The Limits to Growth: A Report for the Club of Rome's Project on the Predicament of Mankind*. New York, Universe Books.

Michael, R. T. and G. S. Becker (1973). "On the new Theory of Consumer Behaviour." *Swedish Journal of Economics* 75: 378-396.

Michaelis, L. and T. Jackson (2003). *Policies for Sustainable Consumption*. London, Sustainable Development Commission.

Mont, O. and R. Bleischwitz (2007). "Sustainable consumption and resource management in the light of life cycle thinking." *European Environment* 17(1): 59-76.

Mont, O. and C. Dalhammar (2005). "Sustainable consumption: at the cross-road of environmental and consumer policies." *International Journal of Sustainable Development* 8(4): 258-279.

Mulder, K., R. Costanza and J. Erickson (2006). "The contribution of built, human, social and natural capital to quality of life in intentional and unintentional communities." *Ecological Economics* 59(1): 13-24.

Nilsson, H. and L. Hansson (2006). *The Social and Economic aspects of Farmers' Markets – from both producers and consumers perspectives. The cases of Malmö and Halmstad Farmers' Markets*. Lund, IIIEE at Lund University**:** 59.

Norton, B., R. Costanza and R. C. Bishop (1998). "The evolution of preferences - Why `sovereign' preferences may not lead to sustainable policies and what to do about it." *Ecological Economics* 24(2-3): 193-211.

Otnes, P. (1988). Housing Consumption: Collective Systems Service. The Sociology of Consumption. P. Otnes, Humanities Press Int.**:** 119-138.

Pearce, D. W. (2000). *Public Policy and Natural Resource Management: A Framework for Integrating Concepts and Methodologies for Policy Evaluation. Report to DGXI, European Commission, Brussels*. London, University College London, commisioned by the European Commission**:** 38.

Princen, T. (1997). "Toward a Theory of Restraint." *Population and Environment* 18(3): 233-254.

Princen, T. (1999). "Consumption and environment: some conceptual issues." *Ecological Economics* 31(3): 347-363.

Princen, T. (2003). "Principles for sustainability: from cooperation and efficiency to sufficiency." *Global Environmental Politics* 3(1): 33-50.

Princen, T. (2005). *The Logic of Sufficiency*. Cambridge, MIT Press.

Reisch, L. and I. Røpke, Eds. (2005). *The Ecological Economics of Consumption*, Edward Elgar Publishing Ltd.

Romm, J., A. Rosenfield and H. S. (1999). *The Internet Economy and Global Warming*. Washington, DC, The Center for Energy and Climate Solutions/The Global Environment & Technology Foundation.

Røpke, I. (1999a). "Analysis: the dynamics of willingness to consume." *Ecological Economics* 28: 399-420.

Røpke, I. (1999b). "The Dynamics of Willingness to Consume." *Ecological Economics* 28: 399-420.

Røpke, I. (2005). Work-related consumption drivers and consumption at work. The Ecological Economics of Consumption. L. Reisch and I. Røpke, Edward Elgar Publishing Ltd**:** 60-78.

Sammer, K. and R. Wüstenhagen (2006). "The influence of eco-labelling on consumer behaviour - results of a discrete choice analysis for washing machines." *Business Strategy and the Environment* 15(3): 185-199.

Samuelson, P. (1947). *Foundations of Economic Analysis*. Cambridge, MA, Harvard University Press.

Samuelson, P. and W. Nordhaus (2004). *Economics*, McGraw-Hill/Irwin.

Sanne, C. (2002). "Willing consumers or locked-in?" *Ecological Economics* 42(1-2): 273-287.

Schor, J. (1999). *The overspent American: why we want what we don't need*. New York, N.Y., HarperPerennial.

Schor, J. B. (1991). *The Overworked American. The Unexpected Decline of Leisure*, Basic Books, USA.

Schor, J. B. (2005). "Prices and quantities: Unsustainable consumption and the global economy." *Ecological Economics* 55(3): 309-320.

Schumacher, E. F. (1973). *Small is Beautiful. Economics as if People Mattered*. London, Blond and Briggs.

Scitovsky, T. (1976). *The Joyless Economy: An Inquiry into Human Satisfaction and Consumer Dissatisfaction*. Oxford., Oxford University Press.

Scitovsky, T. (1992). *The Joyless Economy: The Psychology of Human Satisfaction*, Oxford University Books.

SCR (2006a). *Shifting opinions and changing behaviours*. London, Sustainable Consumption Roundtable**:** 76.

SCR (2006b). *Sustainable Consumption Roundtable Briefing: Communities of Interest – and Action?* London, Sustainable Consumption Roundtable**:** 53.

Segal, J. M. (2003). *Graceful Simplicity. Toward A Philosophy & Politics Of Simple Living*. New York, Henry Holt.

Silberberg, E. (1978). *The Structure of Economics: A Mathematical Analysis*. New York, McGraw-Hill.

Toumanoff, P. and N. Farrokh (1994). *A mathematical approach to economic analysis*, West Publishing Company, st. Paul, MN, USA.

van den Bergh, J. and A. Ferrer-i-Carboneli (2000). Economic Theories of Sustainable Consumption: Empirical and Policy Implications. ESEE Conference 2000, Vienna, Department of Spatial Economics Free University.

van den Bergh, J. C. J. M. and A. Ferrer-i-Carbonell (2000). *Economic Theories of Sustainable Consumption: Empirical and Policy Implications*. Amsterdam, SEO, Foundation for Economic Research of the University of Amsterdam**:** 35.

van den Bergh, J. C. J. M., A. Ferrer-i-Carbonell and G. Munda (1998). Models of Individual Behaviour and Implications for Environmental Policy. Amsterdam, Free University**:** 18.

Veblen, T. (1902). *The Theory of the Leisure Class: An Economic Study of Institutions*. New York, Macmillan.

Wachtel, P. L. (1983). *The Poverty of Affluence: A Psychological Portrait of the American Way of Life*. New York, Free Press / Macmillan, Inc.

Widén, S. (1998). *Bygga och bo i Gebers: varsam ombyggnad för resurshushållning och gemenskap*. Karlskrona, Boverket.

Williams, T. (2002). "Social class influences on purchase evaluation criteria." *Journal of Consumer Marketing* 19(3): 249-276.

WVS (2006). Homepage, World Values Survey.

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Chapter 3

A RECENT TREND IN ECOLOGICAL ECONOMIC RESEARCH: QUANTIFYING THE BENEFITS AND COSTS OF IMPROVING ECOSYSTEM SERVICES

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ABSTRACT

In recent years, there are two notable developments in the research of ecological economics. On the one hand, a tremendous amount of resources have been invested to preserve or improve ecosystem services we derive from natural resources. There is an increasing need for us to better quantify such services and assess their benefits against the investments that we have made. On the other hand, there are significant advancements in environmental modeling that can be used to facilitate the quantitative estimation of ecosystem services. In this study, we present an application that combines developments regarding both of these aspects.

Over the last two decades, the U.S. federal government spent billions of dollars annually on the conservation of cropland. Such expenditures were supplemented with sizable state and local government funding. However, our understanding is incomplete as

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to the impacts of these expenditures on the use of conservation practices and their environmental effectiveness. Expanded knowledge on these issues is greatly needed to support societal decisions as to how much more, if any, we must do to improve environmental quality to a desired level. In this article, we provide some insights into these issues by examining two broad questions: 1) What conservation practices are currently in place, what is their coverage, and what is the cost of these practices? 2) What have been the environmental impacts of the currently-installed conservation practices?

These two questions were examined both at the national level and, as an in-depth case study, at the state level for the state of Iowa. At the national level, information on government expenditures were gathered and previous literature was surveyed to provide an overview of what we know with regard to the two questions. At the state level, to address the first question, we collected data from various surveys, and from federal and state conservation program sources. A database for the costs and coverage of major conservation practices was developed. In order to answer the second question, the widely used Soil and Water Assessment Tool (SWAT) water quality model (Arnold et al., 1998; Arnold and Forher, 2005; Gassman et al., 2007) was employed to estimate the impacts of key conservation practices. The challenges and problems encountered in establishing accurate statewide cost estimates were discussed. The advantages and drawbacks of modeling were also identified to put our results in perspective.

1. INTRODUCTION: A RECENT TREND IN ECOLOGICAL ECONOMICS RESEARCH

The importance of ecosystem services has long been recognized. In Constanza et al. (1997), the value for the entire biosphere is estimated to be over trillions of dollars per year. However, in many places, the integrity of the ecosystem is damaged due to human activities. For example, the soil and water quality in some intensive agricultural areas is so severely degraded that the survival of some wildlife species and the sustainability of the whole ecosystem are in danger. In recognition of such risk, large amounts of resources have been invested yearly in an attempt to try to conserve ecosystems. Nonetheless, given the demand for funding by competing needs like poverty relief and education provision, expenditures on conservation should not be taken for granted. In particular, hard questions are being asked as to the effects of spending on conservation.

Before we can obtain an estimate on the benefits of conservation investment, we need to identify what behavior or land use changes the expenditures have induced. It is not easy to just tally the investment on conservation. At the national level, conservation funding through government programs is well documented and we can easily obtain a number. However, expenditure data at the local and state level are much harder to collect. Moreover, there are investments that did not involve explicit government outlays. For example, in the US, farmers have to adopt conservation measures on highly erodible land under the conservation compliance requirement without any explicit compensation. Some farmers also adopt environmentally friendly practices out of their own conservation ethics or concerns about the long term productivity of their land.

The relationship between land use changes and ecosystem is very complex and thus can be hard to identify. This has been recognized by both ecological and environmental economists (Deacon et al. 1998, and Ropke 2005). Simple relationships based on field experiments and observations can be used. In fact, economists have in general utilized some simplified representations of pollution processes. Earlier studies on the economics of pollution control essentially used a conceptual model of fixed, exogenous pollution delivery coefficients (e.g., Montgomery, 1972; Ribaudo, 1986 and 1989). However, the recent trend is to use more sophisticated environmental models, which have become very useful tools for evaluating alternative conservation management scenarios. There are a variety of models available, each serving its own purposes. For example, there are the Environmental Policy Integrated Climate (EPIC) model (Williams, 1990; Gassman et al., 2005), the Agricultural Policy EXtender (APEX) model (Williams et al., 2006; Williams and Izaurralde, 2006), and SWAT. The EPIC model is an edge-of-field model, i.e., it predicts environmental impacts such as carbon sequestration and nutrient losses at the edge of a field. The APEX model is essentially a multi-field version of EPIC that was developed in the late 1990s to address environmental problems associated with livestock and other agricultural production systems. By contrast, the SWAT model, used in our analysis, simulates in-stream environmental impacts such as nutrient loadings in rivers and lakes. These models have continuously evolved since their initial development and have been applied to a wide range of field, regional, and national studies both in the U.S. and in other countries.

In this paper, we present a study that combines the two notable developments in ecological economics research discussed above. On the one hand, a tremendous amount of resources have been invested to preserve or improve ecosystem services we derive from natural resources. There is an increasing need for us to better quantify such services and assess their benefits against the investments that we have made. On the other hand, there are significant advancements in environmental modeling that can be used to facilitate the quantitative estimation of ecosystem services. More specifically, we will first provide an overview about the conservation investment and its environmental impacts in the U.S. and then we will provide an in-depth case study for the state of Iowa. The challenges and opportunities in compiling the investment and examining environmental impacts will be discussed.

2. An Overview of Conservation Investment and its Impacts in the U.S.

Over the last two decades, conservation on cropland to improve water quality and provide other environmental benefits has been of growing interest. Federal government expenditures on conservation and environmental programs have been 80% higher under the current (2002) farm bill than under the previous bill. As the expiration date for the current bill draws near, it is apparent that the total expenditures and priorities of conservation programs will again be at the heart of legislative debates. The likelihood of tight fiscal budgets over the coming years suggests that competition for federal funding of conservation programs will be at least as intense as in the past if not more so. Hard questions concerning the impacts of these programs on water quality and the environment will need to be answered if such funding is to be maintained or increased. However, there are currently no easy or clear answers to these questions. In this section, we provide an overview of the conservation expenditures in the U.S. and their consequent environmental effects.

2.1. Conservation Investment Under Federal Programs

Federal contributions are a major source of funding for cropland conservation in the U.S. with recent funding levels at over $4 billion annually. However, not all federal programs provide explicit financial incentives to farmers. For example, through Conservation Technical Assistance (CTA), land owners can obtain support from government agencies for the adoption and maintenance of conservation practices without getting direct monetary subsidies. Also, conservation practices are required on highly erodible land in exchange for government commodity payments. In this subsection we first discuss major conservation programs that directly provide payments to farmers. Then, we examine other conservation investments that may or may not require federal outlays.

Starting in the late 1930s, the first conservation programs, including CTA which remains as one of the major component of conservation programs today, were created in response to drought and the dust bowl. This was followed by a period of slow change in conservation until the 1985 farm bill which emphasized erosion and wetland conservation through the creation of the Conservation Reserve Program (CRP) and conservation compliance provisions. The former is a voluntary program that provides rental payments to land owners for keeping their land out of agricultural production. The latter essentially add soil and wetland conservation as a condition for receipt of a wide array of farm program payments. In the 1990 Farm Bill, the Wetlands Reserve Program (WRP) was created with a greater emphasis on water quality. Through the WRP, land owners receive technical and financial support from the federal government for their wetland restoration efforts.

The 1996 farm bill brought the end to several programs and the beginning of a voluntary cost-sharing program, the Environmental Quality Incentives Program (EQIP). This bill also authorized the Wildlife Habitat Incentives Program (WHIP), a voluntary program that provides technical and financial assistance to create high quality wildlife habitats. The 2002 farm bill resulted in a greater role of conservation on working-land, i.e., land in active agricultural production with nearly two-thirds of additional funds going to working land conservation and the creation of the Conservation Security Program (CSP). The CSP, also a voluntary program, provides financial and technical assistance to promote the conservation and improvement of soil, water, air, and other conservation purposes on private working lands. Funding for working land programs was projected to increase to nearly half of overall conservation spending by 2007. This increase in working land conservation funding is apparent in Table 1 by the significant growth in EQIP and the emergence of CSP around this time.

Table 1. Expenditures of Major Conservation Programs (in Million Dollars)

	1997[a]	1998[a]	1999[a]	2000[a]	2001[b]	2002[b]	2003[b]	2004[b]	2005[b]
CTA	**529**	**542**	**548**	**568**	**673**	**696**	**712**	**742**	**720**
Land retirement programs									
CRP	1,710	1,758	1,485	1,511	1,655	1,785	1,789	1,850	1,942
WRP	99	229	144	162	182	263	285	285	275
Subtotal	**1,809**	**1,987**	**1,630**	**1,672**	**1,837**	**2,048**	**2,074**	**2,135**	**2,217**
Working land programs									
EQIP	200	200	174	177	163	313	691	903	1,017
CSP	0	0	0	0	0	0	0	40	202
WHIP	0	30	20	0	13	15	29	38	47
Subtotal	**200**	**230**	**194**	**177**	**176**	**328**	**720**	**981**	**1,266**
Total	2,633	2,788	2,456	2,482	2,723	3,072	3,506	3,858	4,203

[a]Source: Heimlich et al. (2000 a).
[b]Source: Claassen and Ribaudo (2006).

Although information on conservation program funding is readily available, Table 1 does not include cost of conservation compliance provisions because they do not directly involve government appropriations. Even though explicit costs of these provisions are not available, its implications for producers are likely to be significant with over $10 billion of commodity subsidy payments subject to conservation compliance. Thus, efforts have been made to examine the implicit costs to producers and society. Hyberg et al. (1997) estimated that the national cost of conservation compliance was $7.21 per acre, of which $3.78 per acre was incurred by producers and the rest by the federal government. Other national estimates based on EQIP rates showed that producers were willing to accept $18.92, $20.36, and $14.58 per acre to adopt three major conservation practices: conservation cropping, conservation tillage, and crop residue use techniques, respectively (Claassen et al., 2004). This implies that the total cost can be high to adopt these and other practices on the 146 million acres of highly erodible land subject to conservation compliance. However, costs of conservation compliance can vary significantly from field to field with little costs necessary in some cases and significant capital investments or land to be taken out of production in other cases (Hyberg et al., 1997). This suggests the need for using non-aggregated costs and was a key motivation in our use of more spatially detailed cost estimates.

2.2. Conservation Investment Under State and Local Programs

States play critical roles in determining the eligibility of practices and cost-share rates and allocating funds they receive through federal programs such as EQIP. They also directly contribute funding to state conservation programs; however, expenditure levels, environmental concerns targeted, and method of targeting vary across states. Along with financial assistance through cost share programs, states provide low-interest loans. For example, the major state conservation cost share program in Iowa, the Iowa Financial

Incentives Program has an annual funding of approximately $5.5 million (IDALS, 2005). In Idaho, the Resource Conservation Rangeland Development Program recently financed a total of $2.8 million in low-interest loans to fund projects addressing the conservation of water and soil resources (Takasugi, 2006). States also invest in conservation through administering state regulation; however, the costs of such regulations are hard to quantify.

Joint ventures between federal agencies, state and local governments, and private organizations have also contributed to conservation. A prominent example is the control of nonpoint source pollution, which remains as one of the leading environmental concerns in the country. Federal agencies and states have worked closely together to come up with innovative solutions to this problem. There are the Clean Lakes Program, where states receive funding from the Environmental Protection Agency (EPA) for the restoration and protection of lakes and other projects, and the National Estuary Program, where the EPA assists states in implementing programs of estuary conservation (Ribaudo et al., 1999). The Small Watershed Program, a federal program, provides technical and financial assistance to states, local governments, and others to voluntarily plan and install watershed-based projects on private lands (USDA/ERS, 2002). Through the efforts of Ducks Unlimited, the Isaak Walton League, and other organizations, it was estimated that over 18 million wetland acres have been protected, restored, and enhanced since the late 1980’s (Heimlich et al., 2000 b). However, with projects like these receiving contributions from multiple sources, determining the specific contributions and the resulting environmental benefits attributable to each contributor is particularly difficult.

2.3. Overall Adoption of Conservation Practices

The use of conservation practices under federal payment programs is easy to obtain. However, it is more difficult to obtain data on conservation practice use as a result of CTA, conservation compliance, and state and local incentives, as well as for other reasons such as conservation ethics of individual farmers. Over 30 million acres, or about 10% of the nation’s cropland, is kept out of active agricultural production under the CRP, by far the largest conservation program in the country. About 3 millions acres of CRP land is devoted to conservation practices such as filter strips, grassed waterways, contour grass strips, riparian buffers, and other buffer practices. Considerable acreage is also enrolled in other land retirement programs. There is about one and half million acres enrolled in the WRP and approximately 90 million acres subject to Swampbuster provisions, which reduce the incentives to convert wetlands to croplands by denying eligibility for farm program benefits on all acres operated by a grower who either converts a wetland or plants on a converted wetland

Conservation practices applied to working cropland are also important. However, an estimate on the extent of the use of such practices is hard to obtain primarily because information from program data is usually inadequate. For example, there were about a half million acres of no till enrolled in EQIP from 1997-2001 (USDA/NRCS, 2007). However, about 50 million cropland acres were managed with no till annually between 1998 and 2002, based on data reported by the Conservation Technology Information Center (CTIC) (CTIC, 2007). The extensive use of conservation tillage beyond the cost share program, EQIP, may be largely due to conservation compliance provisions because 51 percent of the conservation systems approved to fulfill the compliance provisions use only conservation cropping sequences, conservation tillage, crop residue use, or some combination of these practices (Claassen et al., 2004). It is very likely that farmers' conservation ethics also has some impacts on the wide use of conservation tillage. However, it is hard to estimate the extent of such impacts without proper data. The National Resources Inventory (NRI) contains data on the usage of several practices, including terraces and grassed waterways for the whole nation (USDA/NRCS, 2007a; Nusser and Goebel, 1997). However, the NRI is limited by the fact that the most recently complete national version was collected in 1997 as further discussed in Gassman et al. (2006).

2.4. Estimates of Environmental Benefits at the National Level

Most conservation programs are intended for specific environmental concerns. Soil erosion was the targeted environmental concern in initial CRP enrollments; however, wildlife and other environmental problems have also become important indicators. Conservation compliance has targeted soil erosion on highly erodible land while wetland conservation is the focus of WRP and Swampbuster as they respectively focus on restoration and enhancement of wetlands converted to cropland and conservation of existing wetlands. For crop land conservation, EQIP funds focus primarily on soil and land conservation and water quality. In contrast with other major conservation programs that mainly focus on environmental concerns, the CSP is intended to provide a source of income to producers in addition to improving environmental quality and natural resource conditions in agricultural landscapes. The CTA is not directly targeted towards any single environmental concern as it provides direct technical assistance related to multiple conservation programs.

Many studies have considered the impact of conservation effort on the environment. According to USDA (2006), conservation compliance provisions and other federal conservation programs overall have reduced erosion on national cropland by 1.3 billion tons annually and have also reduced the loss of wetland, leading to a net gain of 260,000 acres from 1997 to 2003. Environmental benefits have also been quantified for some specific conservation programs. An average annual erosion reduction of 294.6 million tons from 1982 to 1997 was estimated to be directly attributable to conservation compliance (Claassen et al., 2004). Other estimates suggest that conservation provisions have kept 1.5 to 3.3 million acres of wetland and 5.6 to 10.9 million acres of highly erodible land from being converted to agricultural production (Claassen et al., 2000).

(FAPRI-UMC, 2007) report several CRP benefits including decreased water erosion (2.13 tons per acre), reductions in nitrogen and phosphorus loss (7.73 and 1.67 pounds per acre, respectively), and increased total organic carbon (0.67 tons per acre). Nationally, they found that total water erosion decreased by 71 million tons, nitrogen loss decreased by 259 million pounds, phosphorus loss decreased by 56 million pounds, and total organic carbon increased by 23 million tons. CRP was also found to be strongly associated with larger populations of grassland birds; however, the relationship varied by ecological region (FSA, 2006). An erosion reduction of 8.6 tons per acre per year was attributed to EQIP (USDA, 2003). It was also estimated that producers who adopted nutrient management standard 590, a conservation practice eligible for incentive payments under EQIP, reduced application of nitrogen, phosphorus, and potash by 25, 5, and 13 pounds per acre, respectively (USDA, 2003). With the majority of program payments going towards pre-existing practices the conservation benefits may be limited from the CSP.

In general, the answers provided by studies like those mentioned above are not satisfactory. They are either too aggregate or too simplistic—depending on crude practice and impact relationships. To gain a better picture about the impacts of conservation investments, the USDA launched the Conservation Effects Assessment Project (CEAP) in 2003 to quantify the environmental benefits of conservation practices (USDA/NRCS, 2007b). The CEAP watershed assessment consists of both a national assessment and studies being conducted on thirty seven watersheds: fourteen to examine water and soil quality and water conservation, ten to examine specific resource concerns, and thirteen to examine optimal scheduling and location of conservation efforts. The watershed studies will provide detailed insights for selected practices and regions, but will provide only limited insight regarding the impacts of conservation practices at the national level.

The impacts of current conservation tillage and other practice usage were compared to hypothetical usage scenarios for the entire nation using EPIC, as part of the CEAP initiative (Potter et al., 2006). They found that current tillage practices were estimated to have reduced sediment, total nitrogen, and total phosphorus loss by 32, 7, and 13 percent respectively, relative to conventional tillage used on all cropland. Comparison of current usage of contour farming, stripcropping, and terracing relative to a scenario with none of these practices suggested that national average sediment, total nitrogen, and total phosphorus loss were reduced by 54, 16, and 28 percent, respectively. However, practice effectiveness varied by region; for example, the greatest potential for sediment reductions was estimated in regions having the highest sediment loss.

3. An in-Depth Study for the State of Iowa

It is clear that more research is necessary to better assess the benefits and costs of investments in improving and preserving ecosystem services, based on the discussion in the previous section. To facilitate such research we need cost data that are more spatially detailed and environmental modeling capacities that account for greater spatial heterogeneities. Recognizing these two needs and taking advantage of the recent developments in the modeling of ecosystem services, we conducted an in-depth study for the state of Iowa centering on two broad questions: 1) What conservation practices are currently in place, what is their coverage, and what is the cost of these practices? 2) What have been the environmental impacts of the currently-installed conservation practices? In this section, we detail the research process that we used to shed light on the above questions.

3.1. The Costs and Usage of Conservation Practices in Iowa.

To address the first question, we developed county-level average cost estimates for major conservation practices. In developing cost estimates for the practices, we primarily used data reported by conservation programs. To determine the reliability of our estimates, we also contacted conservation agencies. The offices contacted include county USDA Natural

Resources Conservation Service (NRCS) and Farm Service Agency (FSA) offices. However, county offices were not the main source of the cost data and were of limited help in creating average cost estimates, though they were very valuable in testing the reasonableness of costs reported by other sources. In fact, when we did obtain cost estimates from county conservationists they were generally based on either past conservation projects, often done under the major conservation programs, or current rates listed by the conservation programs. Therefore, we believe that utilizing cost information gathered from the major conservation programs leads to greater accuracy in our estimates, and is the method we used. The general research process is as follows:

- Collect and analyze cost data from major conservation programs;
- Determine outliers that may reflect incorrect costs;
- Contact conservationists to determine the reasonableness of outliers;
- Fill in for counties with missing average costs;
- Collect usage data;
- Calculate total state costs.

While we believe our methodology provided reasonable cost estimates, it is important to note that they were only estimates of county averages, and individual cost numbers could vary. Also, many of the practices examined differ structurally across sites, so a relatively lower cost in one county does not necessarily indicate better cost management. For example, with filter strips, the lower-cost cool-season grass seed may be selected; however, some studies suggest that certain warm-season grasses are more effective than cool-season grasses in nitrogen, phosphorous, and sediment removal (Lee et al., 1999).

3.1.1 Collecting Data and Calculating Costs

Cost data was collected from several major conservation programs, including two USDA programs, the CRP and EQIP, and a state program, the Iowa Department of Agriculture and Land Stewardship (IDALS) Iowa Financial Incentive Program (IFIP). A comparison of these programs is provided in Table 2. IFIP is a voluntary state program that provides cost share and incentive payments. It covers many of the practices listed in this report, including contour farming, grade stabilization structures, grassed waterways, no till, terraces, and water and sediment control basins. To qualify for assistance under the program a person must own at least 10 acres and produce $2,500 of an agricultural commodity. Under the program, each county is given an annual allotment based partially on its share of Iowa's most erosive cropland soils; counties are allowed to set their own priorities for application and practice selection.

CRP is a voluntary federal program that provides cost-share and financial incentives as well as annual rental payments for the retirement of cropland and the installation of conservation practices on the retired land. It also covers most of the practices listed in this report, including contour buffer strips (contour grass strips), filter strips, grassed waterways, and riparian buffers. Up to 50% of the cost to establish an approved conservation practice can be paid in cost share payments by the Commodity Credit Corporation. Certain practices are also eligible for an incentive payment of 40% of the practice installation cost. To qualify for assistance under the program, generally, a person must have owned or operated the land for at least 12 months prior to close of the CRP sign-up period. Land must also be capable of being cropped and have been planted to an agricultural commodity for several years in the recent past; however, marginal pastureland may also be eligible. Applications are prioritized using the Environmental Benefits Index. Continuous CRP and general CRP are two ways in which land can be enrolled in CRP (USDA/FSA, 2007).

Table 2. Sources for practice costs

	IFIP	CRP	EQIP
Description	State Conservation Program	Federal Conservation Program	Federal Conservation Program
Program Function	Cost-Sharing and Incentive Payments	Cost-Sharing, Maintenance and Other Incentive Payments, and Rental Payments	Cost-Sharing and Incentive Payments
Practices Used	Contour Farming GSSs Grassed Waterways No till Terraces WSCBs	General CRP Continuous CRP Contour Buffer Strips Filter Strips Grassed Waterways Riparian Buffers Wetland Restoration	Contour Buffer Strips Contour Farming GSSs Grassed Waterways No till Nutrient Management Terraces WSCBs
Years covered	1997 to 2006	CRP Contracts as of December 2005	1997 to 2006

*WSCBs = Water and Sediment Control Basins
**GSSs = Grade Stabilization Structures

EQIP is a voluntary federal program that provides cost share and incentive payments. All of the practices listed in this report are covered under EQIP. To qualify for assistance under the program a person must have eligible land on which they produce livestock or crops. Under the program, both state and local conservation practice priorities are set. Applications are prioritized for funding using a state or locally developed ranking worksheet that generally considers various factors, including cost-effectiveness, resources to be treated, meeting national EQIP priorities, and compliance with environmental regulations.

The costs were directly reported for IFIP, but the CRP and EQIP costs had to be calculated using cost share payments and average cost share rates. The IFIP data set contained contract-level data reported over the period 1997 to 2006. To calculate the average cost of a practice for a given county, the data was first aggregated within the county and over time; the average cost of the practice in the county was then calculated as the aggregated actual cost divided by the aggregated amount installed of the practice in the county.

The CRP data set contained county level data for CRP contracts installed as of December 2005. CRP costs had to be indirectly determined since actual costs were not listed. To determine the average cost of a CRP practice, we assumed an average cost share rate of 50% for all practices based on expert opinion provided by the USDA. FSA documentation (USDA/FSA, 2007) further supports the use of this rate by stating, "FSA provides cost share assistance to participants who establish approved cover on eligible cropland. The cost share assistance can be an amount not more than 50% of the participants' costs in establishing approved practices."

For EQIP, we collected and used two datasets. One dataset contained county level data aggregated for the period 1997 to 2005. This set included information on the cumulative practice cost share payment, average cost share rate, and the total amount installed. To determine a practice's average cost, we first divided the cumulative cost share payment by the average cost share rate to determine cumulative cost. Average cost was then calculated by dividing cumulative cost by the total amount installed. The other dataset was compiled from online EQIP county reports for 2004 and 2006; these reports included rates of certain practices, which were used as average cost estimates for these practices under EQIP (e.g., nutrient management and contour farming).

Once costs were calculated for these programs, we had to determine, for a given practice and county, which program should be used for establishing our average cost estimate. For most practices, IFIP data was first used because it generally had more counties with cost estimates, we believed the reported costs were more reliable, and we were able to check the data by directing any questions to the IFIP program administrators. Generally, in cases in which IFIP data was not available, we then used EQIP or CRP cost data, depending on the program's coverage of the specific practice.

3.1.2 Testing Reasonableness of Average Cost

We next considered whether the calculated average costs from these programs seemed reasonable. To determine reasonableness, outliers were determined through mapping and querying. Determination of outliers was subjective; if there was a large break between a specific observation and the other observations it was considered an outlier to be followed up on. For some practices, relevant literature was also available and was used in determining reasonableness. Next, county conservation offices were contacted regarding outliers. In speaking with the county conservationists, we attempted to determine their estimate of the practice's average cost in the county and their opinion as to the reasonableness of the average cost we derived from the conservation program data. If a cost number was decided to be unrealistic, we asked the conservationist for suggestions of an acceptable estimate.

3.1.3 Addressing Missing Costs

Because we collected information from a number of conservation programs, we were able to rely heavily upon the program costs reported for most practices. However, even when costs could be determined for most counties using conservation program data, there were usually a few counties that did not have a cost reported. Also, a few practices did not have a significant number of counties with cost information or proxies (incentive paid) listed. There were several reasons why costs were not listed for these counties. These reasons ranged from little to no use of the given practice in the county to cost share or incentives not being offered since the practice was already widely adopted in the county. A practice that was widely used throughout most of the state might not be used in a small number of counties because each county sets its own priorities for practices. Because counties are allowed to prioritize practices, there would be little reason to emphasize a practice that didn't fit the county's specific environmental conditions, even if it was effective in most other counties. An example of this is the "Prairie Pothole" region of Iowa, where the landscape is relatiely flat and thus there is little reason to install terraces even though they are used in the majority of Iowa's counties. However, this region has instead emphasized other practices such as conservation tillage and grassed waterways as a way to reduce erosion.

Regardless of the reason, when cost estimates were missing, the first course of action taken to fill them in was contacting state or county conservationists. While contacting conservationists was very useful in determining reasonableness of outliers, it was more limited in filling in missing costs. This is because missing cost information was usually due to the fact that the practice was generally not used in the county. This in turn meant that there were few to no prior examples on which the county conservationist could base his/her estimate. At this point, the method for filling in missing values varied significantly depending on the circumstances for which the data were missing. For practices that had costs reported throughout most of the state, we used an average of surrounding county costs to fill in missing values. This seemed reasonable because counties in close proximity may have more similar characteristics that could impact the cost of the practice. For no till and contour farming, costs were not listed in enough counties that average cost of surrounding counties could be effectively used. Thus, we used estimates from prior research for no till and an average estimate for the entire state based on incentive payments for contour farming. At this point, county-level average cost estimates had been established for all counties. The cost estimates by practice and by county are provided in Appendix A.

For structural practices, which include grassed waterways, terraces, water and sediment control basins, grade stabilization structures, filter strips, contour buffer strips, riparian buffers, and wetland restoration, it is important to note that the average costs are reported as the cost of construction and do not include land rental costs. They are also reported as one-time costs and have not been amortized over the life of the structures. Finally, the unit that average cost is calculated over is the amount or area of the practice installed/implemented and not the area impacted. Thus, for grassed waterways an average cost of $2,000 per acre is the average cost of installing one acre of grassed waterway and not the cost per acre impacted by the grassed waterway.

3.1.4 Determining Usage Level

To determine the amount of the practice on the ground, we also relied on several sources, which are described in Table 3. The first data set used was from the NRI. This covered practice usage for contour farming, filter strips, grassed waterways, and terraces. One difficulty with using the NRI dataset is that NRI points do not report the amount of a given practice in the area represented by the point. Instead, they report if a given practice is located within the area. Since some practices, such as grassed waterways and terraces, only account for part of the land area associated with an NRI point, the amount of the practice on the ground could not be directly determined. Instead, the amount of grassed waterways was indirectly calculated by assuming the practice takes up 2% of the land area in which they are reported; this follows the conversion rate used in Secchi et al. (2007). The second data set was from CTIC and was used to determine usage of no till. The maps in appendix A based on CTIC data can be assumed to reflect the amount of no till on the ground. Some conservation programs (IFIP, CRP, and EQIP) also had information on practice usage for no till and other practices. However, this information was limited to cases in which the practice was implemented under the given conservation programs.

Table 3. Sources of Practice Usage.

	Surveys		Conservation Programs
	NRI	CTIC	IFIP, CRP, and EQIP
Description	USDA Survey	Reported findings from the USDA's Crop Residue Management Survey	State and federal conservation programs
Program Function	Report survey findings	Report survey findings	See Table 2
Coverage	Contour Farming Filter Strips Grassed Waterways Terraces Erodibility Measures	No till	See Table 2
Years	1997	2004	1997-2006

3.1.5 Calculating Total Costs

Statewide estimates of average costs are presented in Table 4. Based on these average costs, total costs were developed for several practices currently in place and are reported in Table 5. Practice usage is based on levels reported in the 1997 NRI and 2004 CTIC, and practice life span is based on those used in Secchi et al. (2007). When NRI data were used, the amount of practice used was considered to be the entire land area corresponding to the NRI point reporting practice usage. The exception to this is terraces and grassed waterways for which conversion rates were used to estimate practice area from the total land area of an NRI point.

The cost estimate of contour stripcropping was determined by multiplying an assumed $15 average cost per acre (the flat rate incentive payment paid under IFIP) by NRI contour stripcropping acres. For contour farming, our statewide average cost estimate of $6 per acre was multiplied by NRI contour farming acres to determine cost. The NRI grassed waterway acres in each county were multiplied by 0.02, based on the conversion rate suggested in Secchi et al. (2007), to determine grassed waterway acres. Statewide cost was then calculated by multiplying grassed waterway acres in each county by our county level average cost estimates and then summing across counties. The conversion rate, 166.67 feet of terrace per acre, used in Secchi et al. (2007) for their low-cost estimate, was used to convert NRI terrace acres to terrace footage. Calculated feet of terrace in each county were then multiplied by our county level average cost estimates and summed across counties. The costs of no till and mulch-till are based on average costs of $20 and $10 per acre, for respective practices, and usage levels reported in CTIC. Total cost is calculated as average cost multiplied by practice usage. CRP cost is determined by multiplying acres under CRP for each county, reported in CTIC, by our county level average annual rental payment estimates and summing across counties. Since the first two practices in the table, terraces and grassed waterways, are structural practices, their annual costs were derived as the costs calculated above divided by practice life span. Practice life span was considered to be 25 years for terraces and 10 years for grassed waterways, following practice life spans used in Secchi et al. (2007). This yields a statewide cumulative annual cost of $434,685,154 for the seven practices listed ($37,194,949 for terraces and grassed waterways and $397,490,205 for the other five practices listed).

Table 4. Estimates of State Average Costs

Practice	Statewide Average Cost
Grassed Waterway	$2,127/acre
Terrace	$3.57/ft
Water & Sediment Control Basin	$3,989/structure
Grade Stabilization Structure	$15,018/structure
Filter Strip	$116.83/acre
Contour Buffer Strip	$78/acre
Riparian Forest Buffer	$486/acre
Wetland Restoration	$245/acre
Nutrient Management	$4.09/acre
Contour Farming*	$6/acre
No till	$17.94/acre
Continuous CRP**	$142/acre
General CRP**	$97/acre

* Estimated cost of $6/acre only applies to locations where contour farming is practical.
** Values reported are average annual rental payment per acre

Table 5. Total Statewide Costs for Selected Practices Currently in Place

Practice	Cost	
Terraces	$692,147,676	$37,194,949
Grassed Waterways	$95,090,424	
Contour Farming	$30,889,200	$397,490,205
Contour Stripcropping	$3,552,000	
No till	$104,308,740	
Much-Till	$82,861,900	
CRP	$175,878,365	

3.2. The Environmental Impacts of Conservation in Iowa.

The SWAT model was used in this study to estimate changes in water quality due to changes in conservation practices. SWAT is a hydrologic and water quality model developed by the USDA's Agricultural Research Service. It is a long-term continuous watershed scale simulation model that operates on a daily time step and is designed to assess the impact of different management practices on water, sediment, and agricultural chemical yields. The model is physically based, computationally efficient, and capable of simulating a high level of spatial detail. Major model components include weather, hydrology, soil temperature, crop growth, nutrients, pesticides, and land management. In SWAT, a watershed is divided into multiple subwatersheds, which are further subdivided into unique soil/land use characteristics called hydrologic response units (HRUs). The water balance of each HRU is represented by four storage volumes: snow, soil profile, shallow aquifer, and deep aquifer. Flow generation, sediment yield, and pollutant loadings are summed across all HRUs in a subwatershed, and the resulting loads are then routed through channels, ponds, and/or reservoirs to the watershed outlet. Description of some of the processes involved in the SWAT model is provided in Appendix B.

3.2.1 SWAT Model Setup

Watershed delineation of a study region is the first step in a SWAT application. The SWAT simulations were configured for 13 major watersheds in Iowa that range in size from 1,974 km^2 to 36,358 km^2 (Table B.1) and together cover 87% of the state (Figure 1). These watersheds were selected because they were completely or mostly located in Iowa, and they represented the majority of Iowa land area for the SWAT scenarios. Delineation of each watershed into smaller spatial units required for the SWAT simulations consists of two steps: (1) subdividing each major watershed into smaller units based on U.S. Geological Survey (USGS) 8-digit Hydrologic Cataloging Unit (HCU) watersheds (Seaber et al., 1987) or smaller 10-digit watersheds (as described in IDNR, 2001), and (2) further subdividing the subwatersheds into HRUs. Larger 8-digit subwatersheds were used for the Des Moines and Iowa River Watersheds (Figure 1), which were the two largest watersheds included in the analysis. The smaller 10-digit subwatersheds were used for those watersheds that consist of 1 to 3 8-digit watersheds (Figure 1), to avoid potential distortions in predicted pollutant indicators when only a small number of subwatersheds are used in a SWAT application, as discussed by Jha et al. (2004).

Historical precipitation, maximum temperature, and minimum temperature data obtained from the Iowa Environmental Mesonet (IEM, 2007) were used for the SWAT simulations. Key input data required for the SWAT simulations included land use, soil, management, and climate data. A key data source for the land use, soil, and management information was the 1997 NRI, which contains soil type, landscape features, cropping histories, conservation practices and other data for roughly 800,000 U.S. nonfederal land “points” including over 23,000 in Iowa. Each point represents an area that is assumed to consist of homogeneous land use, soil, and other characteristics, which generally range from a few hundred to several thousand hectares in size. Table 6 provides the characteristics of each watershed.

Clusters of NRI point areas formed the HRUs for each of the SWAT simulations. All of the points within a given category were clustered together within each 8-digit watershed for the Des Moines and Iowa River Watershed simulations, except for the cultivated cropland. For the cultivated cropland, the NRI points were first aggregated into different crop rotation land use clusters within each 8-digit watershed, based on the NRI cropping histories. These crop rotation aggregations were then subdivided based on permutations of rotations; e.g., corn-soybean versus soybean-corn. The tillage implements simulated for the different levels of tillage (conventional, reduced, mulch, and no till) incorporated in the analysis were obtained from the USDA 1990-95 Cropping Practices Survey. The soil layer data required for the SWAT simulations are input from a soil database that contains soil properties consistent with those described by Baumer et al. (1994), with the additional enhancement of ID codes that allow direct linkage to NRI points.

Table 6. Characteristics of the 13 Study Watersheds

Watershed	# of Subwatersheds	Drainage Area mi^2	Drainage Area km^2	Key Land Uses (% of watershed) Cropland	Grassland*	Forest	Urban
Boyer	5	1,089	2,820	68	26	4	2
Des Moines	9	14,477	37,496	71	16	6	7
Floyd	5	917	2,376	84	13	0	3
Iowa	9	12,663	32,796	77	12	4	8
Little Sioux	10	3,553	9,203	86	13	1	0
Maquoketa	10	1,864	4,827	56	32	10	3
Monona	5	947	2,452	78	19	2	1
Nishnabotna	11	2,980	7,718	84	15	1	0
Nodaway	7	792	2,051	52	41	5	3
Skunk	12	4,342	11,246	69	25	5	1
Turkey	9	1,699	4,400	56	25	16	3
Upper Iowa	7	992	2,569	51	26	19	3
Wapsipinicon	11	2,542	6,582	77	19	3	1

*Includes CRP and Pasture.

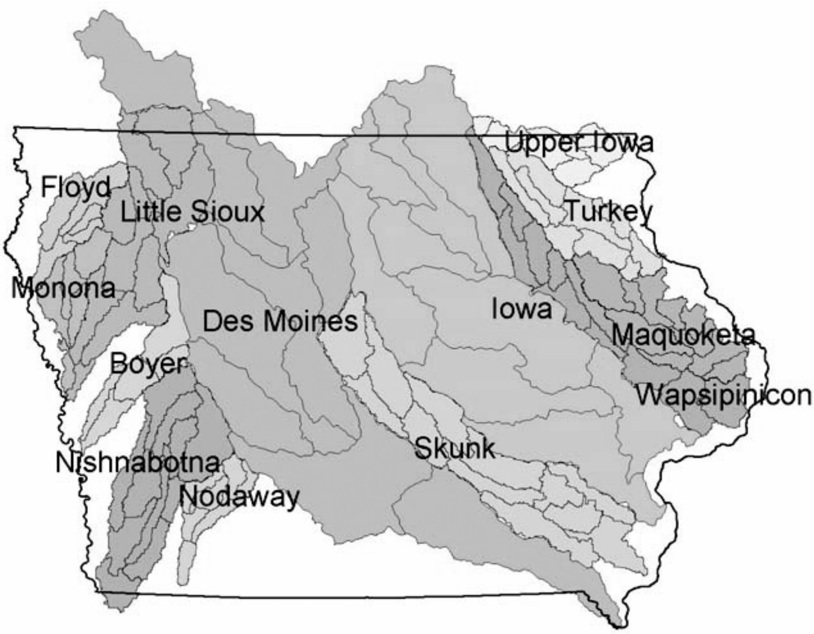


Figure 1. The Study Area and Watershed Delineations

A more complex procedure was developed to construct the HRUs for the SWAT baseline simulations for the other 11 watersheds, because the NRI data is not spatially referenced at the 10-digit watershed level. To overcome this limitation, Iowa Soil Properties And Interpretations Database (ISPAID) soil data and 2002 Iowa Department of Natural Resources

(IDNR) land use data were used to help determine which HRUs should be placed in each 10-digit subwatershed. The initial step in the procedure consisted of attempting to match soil IDs shown in the ISPAID soil map for a 10-digit watershed to soil IDs listed in the NRI for points located within the respective 8-digit watershed that the 10-digit watershed was located in. A positive match indicated that the NRI point could be located in that 10-digit watershed. The 2002 IDNR land use data was then used to help further verify which 10-digit watershed an NRI point was most likely to be located in, based on whether the land use was cropland, CRP, forest, urban, and so forth.

The effect of conservation practices is accounted for by adjusting the "support practice (P) factor," which is one of the factors used in the original USLE equation (Wischmeier and Smith, 1978) and also in the MUSLE equation that is used in SWAT. The P factors used for contouring and terraces are based on values reported by Wischmeier and Smith (1978) as a function of slope range (Table 7). The choice of a P factor value of 0.4 for grassed waterways is based on the methodology used by Gassman et al. (2006) for simulating the impact of grassed waterways in the Mineral Creek Watershed in eastern Iowa. The effect of grassed waterways was further accounted for in SWAT by adjusting the Manning's N values for the affected HRUs.

To estimate the water quality changes, it is necessary to calibrate SWAT to existing baseline data on the watersheds and to accurately represent the current land use, land management, and weather conditions of the region using data obtained from several sources. A calibration and validation exercise was performed with SWAT2005 for all 13 Iowa watersheds. Results are provided in Appendix B. The SWAT calibration results were generally quite strong, especially for streamflow because of the abundance of measured streamflow data. The water quality components were also calibrated but with lower confidence because of a lack of sufficient measured data.

Table 7. Original P-Factor Values for Contouring, Terraces, and Grassed Waterways

Slope ranges	Contouring[a]	Terraces[a,b]	Grassed Waterways[c]
1 to 2	0.6	0.12	0.4
3 to 5	0.5	0.1	0.4
6 to 8	0.5	0.1	0.4
9 to 12	0.6	0.12	0.4
13 to 16	0.7	0.14	0.4
17 to 20	0.8	0.16	0.4
21 to 25	0.9	0.18	0.4

[a]Source: Wischmeier and Smith (1978).
[b]Based on expected sediment yield for terraces with graded channels and outlets.
[c]Source: Gassman et al. (2003).

3.2.2 Impacts of Existing Conservation Practices

We undertook a hypothetical simulation experiment of removing all existing conservation practices from the landscape using the calibrated SWAT model. The resulting water quality values were then compared with the corresponding values of the current baseline results. The difference between these two simulations provides an estimate of the water quality benefits that the current existing conservation practices yield. The key water quality indicators evaluated in this study were nitrogen and phosphorus. Table 8 shows the baseline average annual values of the water quality parameters, averaged over a 20-year period from 1986 to 2005. Total N is total nitrogen, which includes nitrate and organic nitrogen (Org N). Similarly, total phosphorus (Total P) includes organic phosphorus (Org P) and mineral (soluble) phosphorus (Min P).

Table 8. Baseline Average Annual Values (Metric Tons) of Water Quality Parameters, Averaged over a 20-Year Period (1986-2005)

	Flow (mm)	Nitrate	Org N	Min P	Org P	Total N	Total P
Boyer	167	2,699	1,409	418	640	4,108	1,059
Des Moines	177	60,406	3,057	1,299	885	63,463	2,184
Floyd	115	3,625	962	195	174	4,587	368
Iowa	276	65,639	5,122	1,330	1,287	70,761	2,616
Little Sioux	167	14,699	1,395	220	434	16,094	654
Maquoketa	251	8,516	1,965	326	101	10,482	427
Monona	119	2,143	599	50	72	2,741	121
Nishnabotna	231	8,848	3,307	1,303	1,597	12,155	2,900
Nodaway	213	1,984	614	221	313	2,598	534
Skunk	230	13,609	2,166	1,324	1,200	15,775	2,524
Turkey	261	6,354	2,803	476	652	9,157	1,127
Upper Iowa	253	2,335	482	45	135	2,817	180
Wapsipinicon	284	14,253	1,166	275	268	15,419	543

Table 9. Percentage Reduction in Average Annual Baseline Values of Flow and Nutrient Loadings Due to Existing Conservation Practices in Iowa Watersheds

	Flow	Nitrate	Org N	Min P	Org P	Total N	Total P
Boyer	-8	23	56	45	50	38	48
Des Moines	-8	14	39	29	37	15	33
Floyd	-4	19	42	41	44	25	42
Iowa	-5	10	41	0	40	13	25
Little Sioux	-7	20	52	40	50	24	47
Maquoketa	-1	8	42	37	42	17	39
Monona	-3	15	49	54	61	26	58
Nishnabotna	-3	21	52	45	47	33	46
Nodaway	-2	28	56	49	51	37	50
Skunk	-6	14	46	40	44	21	42
Turkey	-5	5	38	30	37	18	34
Upper Iowa	-3	7	48	38	47	18	45
Wapsipinicon	-8	6	46	34	45	11	40

Removal of existing conservation practices includes removal of all CRP land, conservation tillage, terraces, contouring, strip cropping, and grassed waterways. CRP lands were converted into corn-soybean cropland, conservation tillage was switched to conventional tillage, and other conservation practices were just removed. These conservation practices are basically designed to prevent erosion. Reducing sediment transport helps reduce sediment-bound organic compounds of nitrogen and phosphorus. Additionally, taking land out of production removes all fertilizer application and hence reduces nutrients overall. The size and environmental conditions of each watershed also affect the predicted outcomes. Table 9 shows the percentage change in flow and nutrient loadings when all conservation practices, listed above, are removed from the existing baseline. Streamflow was estimated to increase in all watersheds, indicating that such conservation practices allow faster movement of water. Nitrate loadings were estimated to decrease significantly, especially for western watersheds. Nitrate reductions are greater than 10% for nine watersheds. The estimated reductions for Org N, Org P, and Min P were much greater.

4. CONCLUSION

In this study, we examined an important trend in ecological economics research — quantifying the benefits and costs of improving ecosystem services. We found that while some effort has been devoted to this subject, significant research is still required at the national level in the U.S. We then reported the result of a detailed study for the state of Iowa. Our results clearly revealed that large resources have been invested in conserving ecosystems in Iowa and these investments have generated significant ecological benefits. To our knowledge, our study was the first watershed-based analysis that attempted to investigate both the costs and benefits of conservation expenditures for a whole state. However, the need for such research has begun to be widely recognized and the research trend is clear as shown by recent national initiatives like the CEAP project. In time, the devotion to this research need will provide better answers as to the costs and benefits of improving services from our ecosystems.

In our research project, we felt that what we could do was greatly limited by data and modeling capacity that are currently available. To better facilitate cost-benefit assessment of conservation investment, improvements are necessary in two respects. The first is consistent and systematic data on the cost and use of conservation practices. We found that intermittent data collection and the lack of a central data source made cost data difficult to acquire/interpret. There are several things that we decided not to model because we did not think we had sufficient data. For example, we did not consider changes in the timing of fertilizer application since data on the cost of such changes were not available. We also found that the computation of full costs was problematic because of the lack of information on the opportunity cost for farmers' time and risk attitudes. To estimate costs in these dimensions, we need farm-level data that are combined with field-level data, including farmers characteristics, farm conditions, and physical environment of the farm.

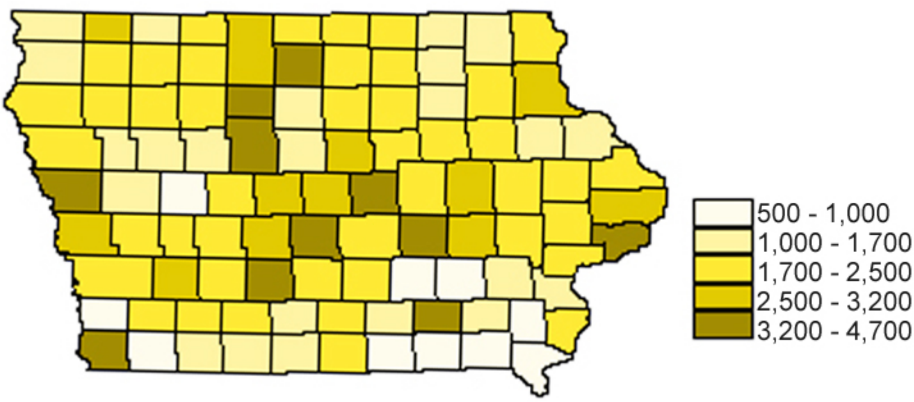
The second aspect of cost-benefit assessment that needs further development is modeling capacity that more realistically represents enviromental processes. The SWAT water quality model we used in our study is currently one of the state-of-the-art models and was shown to be able to reasonably replicate actual hydrological processes in the watersheds included in our study. However, SWAT can not directly simulate the effects of riparian buffers and has further limitations in being able to simulate the impacts of constructed wetlands and other targeted conservation practices (Gassman et al., 2007). Given this, we did not consider wetlands or riparian buffers in our analysis of environmental impacts. However, constructed wetlands are very effective in controlling nitrogen runoff, especially in landscapes that are managed with subsurface tile drains. This, coupled with the fact that we did not incorporate more detailed analysis of nutrient input management (e.g., multiple variations in application timing, modes, and/or application rates), likely results in our findings being biasd towards under-reporting of conservation practice nitrogen reduction impacts. Moreover, good monitoring data are necessary for the appropriate calibration and validation of environmental models. In our project, we obtained very good calibration and validation results for the SWAT model with regard to stream flow. However, the water quality components of the model (sediment, nitrogen, and phosphorous) were calibrated with lower confidence levels due to the lack of sufficient measured data. Thus, it is important that systematic efforts are pursued to ensure that quality monitoring data is obtained for watershed ecosystems in Iowa and across the U.S.

Appendix A. Iowa Conservation Practice Cost and Usage Estimates by County

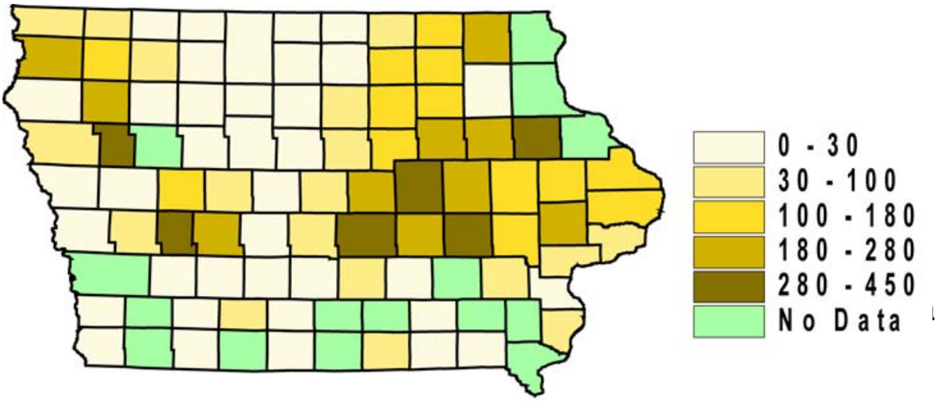
In this appendix, we present the estimates of costs and usage for the following practices: grassed waterways, terraces, water and sediment control basins, grade stabilization structures, filter strips, contour buffer strips, riparian buffers, wetland restoration, nutrient management, no till, and general CRP. The usage information is based on the data source that is considered to be most interesting for the practices.

Grassed Waterways

Map A.1. Average Cost per Acre

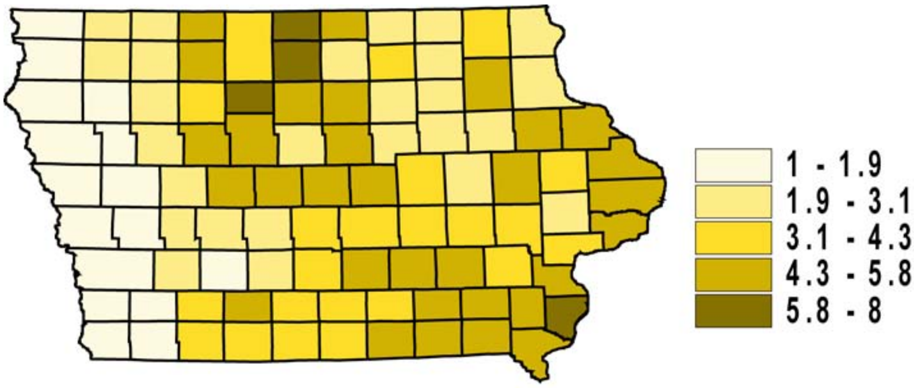


Map A.2. Total Acres Installed (IFIP)

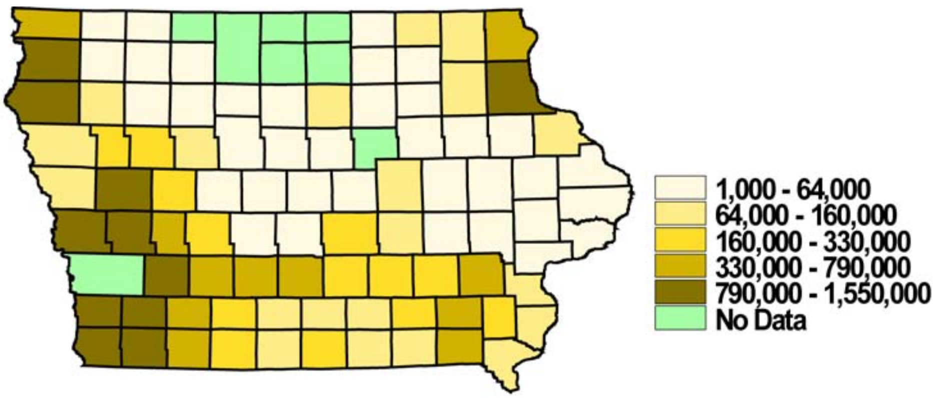


Terraces

Map A.3. Average Cost per Foot

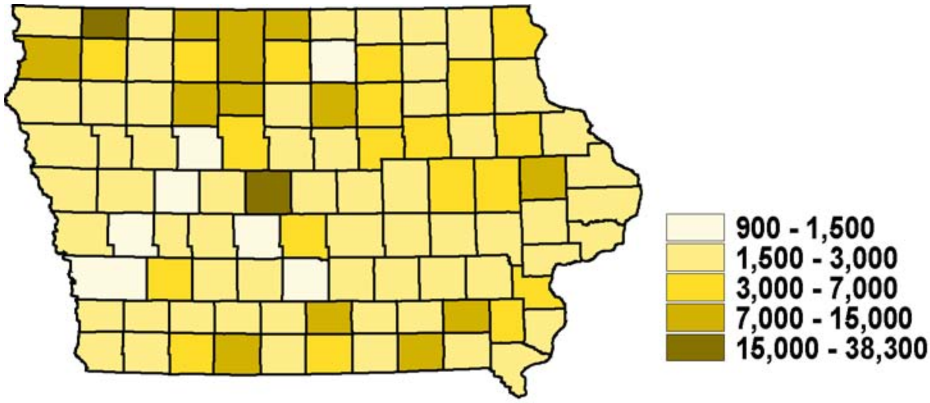


Map A.4. Total Feet Installed (IFIP)

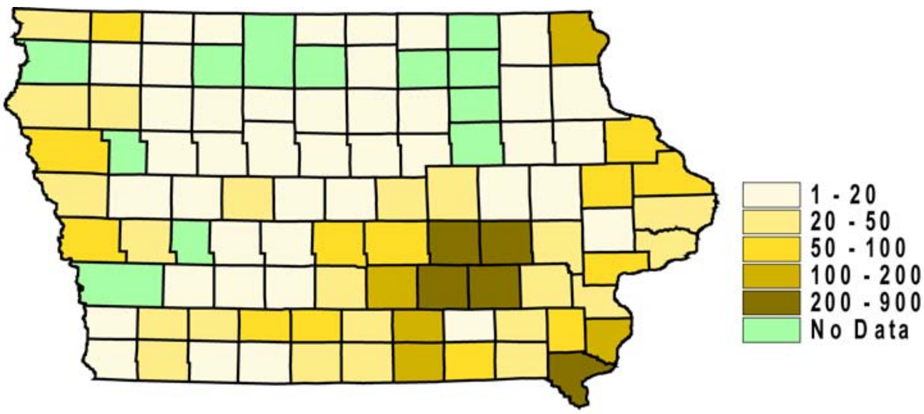


Water and Sediment Control Basins

Map A.5. Average Cost per Structure

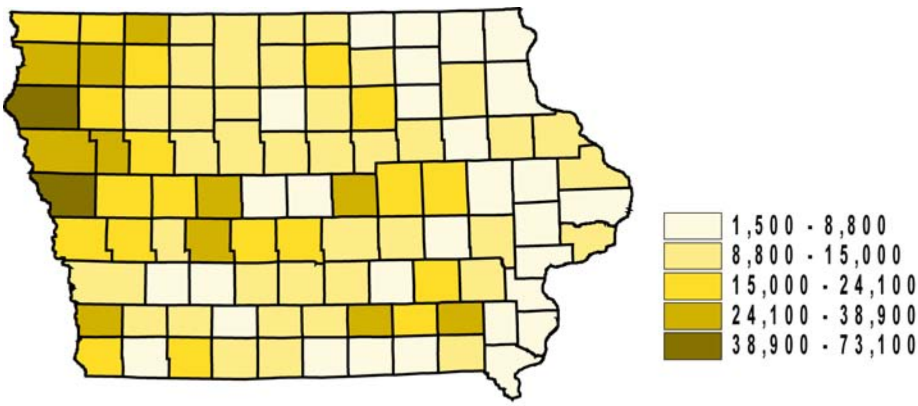


Map A.6. Total Structures Installed (IFIP)

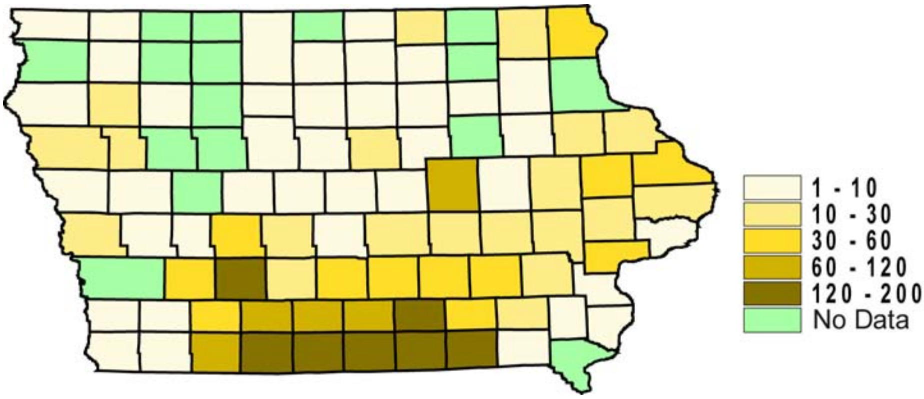


Grade Stabalization Structures

Map A.7. Average Cost per Structure

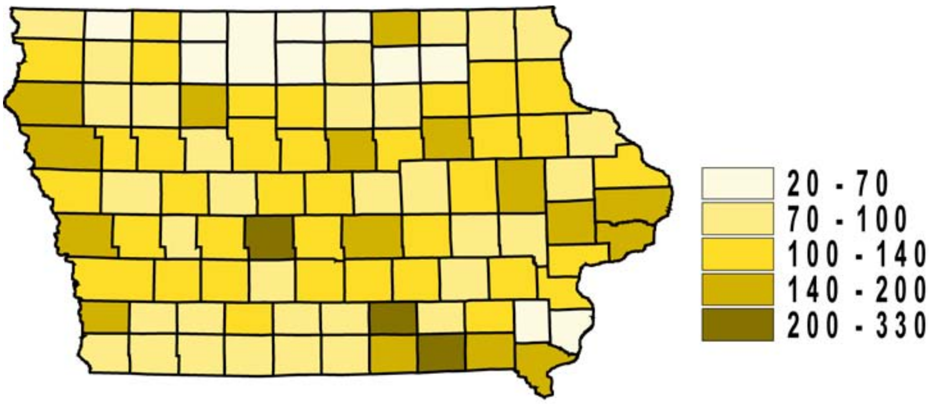


Map A.8. Total Structures Installed (IFIP)

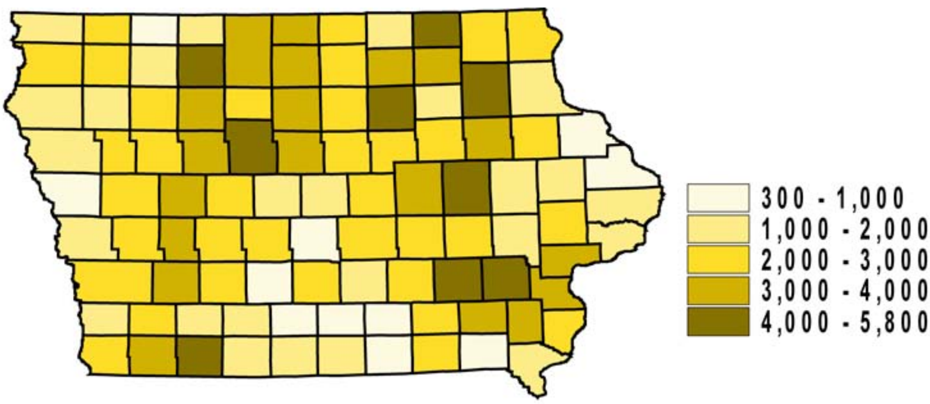


Filter Strips

Map A.9. Average Cost per Acre

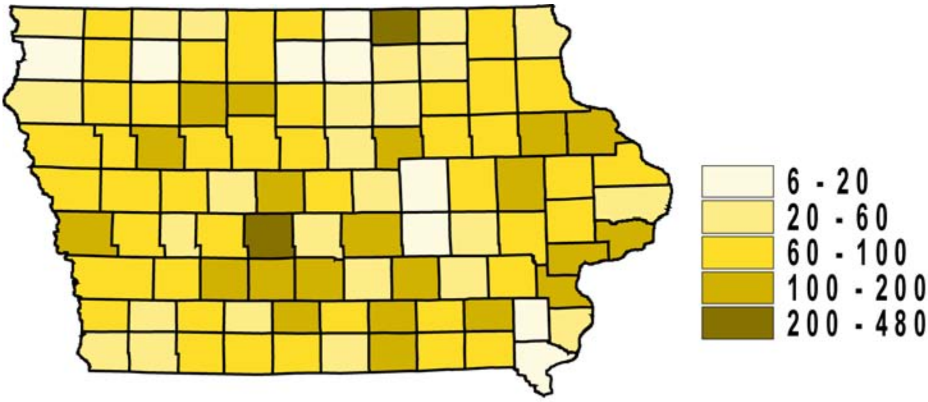


Map A.10. Total Acres Installed (CRP)

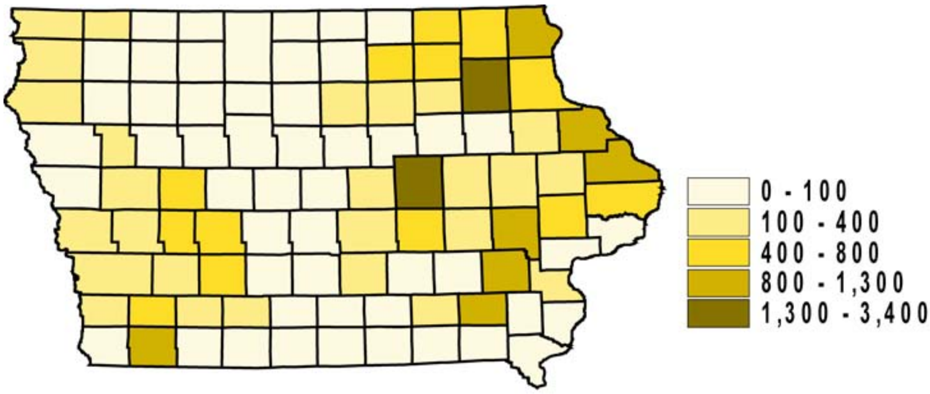


Contour Buffer Strips

Map A.11. Average Cost per Acre

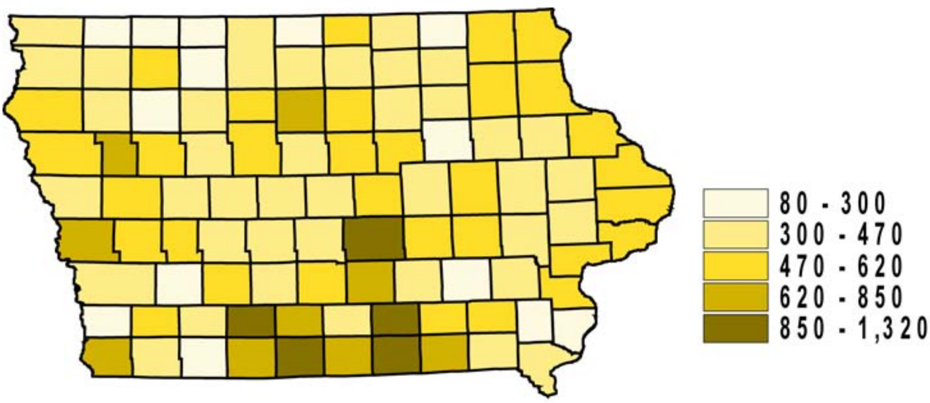


Map A.12. Total Acres Installed (CRP)

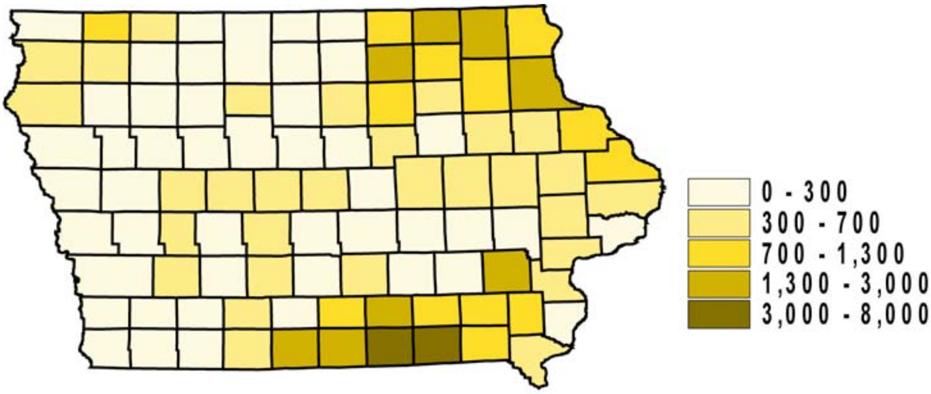


Riparian Buffers

Map A.13. Average Cost per Acre

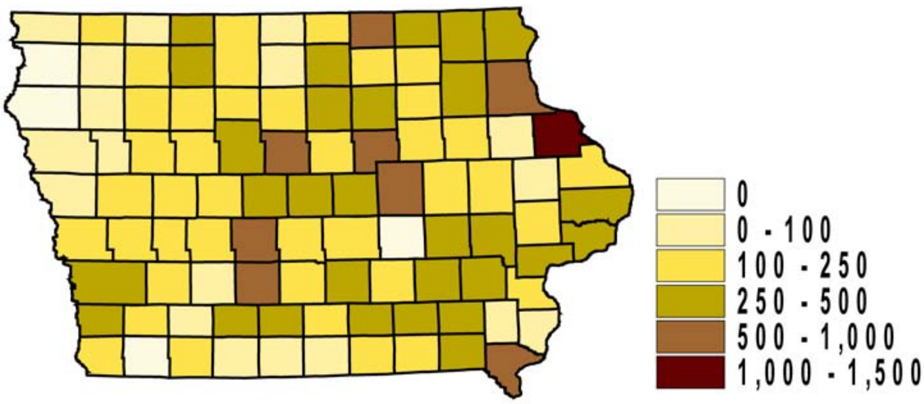


Map A.14. Total Acres Installed (CRP)

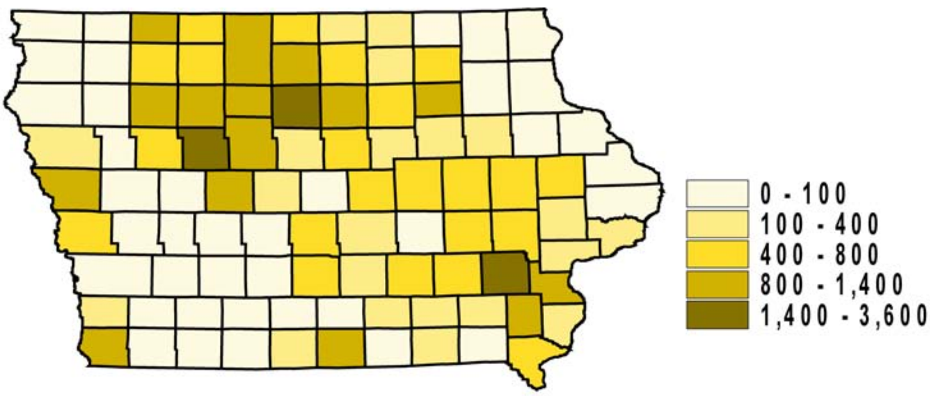


Wetland Restoration

Map A.15. Average Cost per Acre

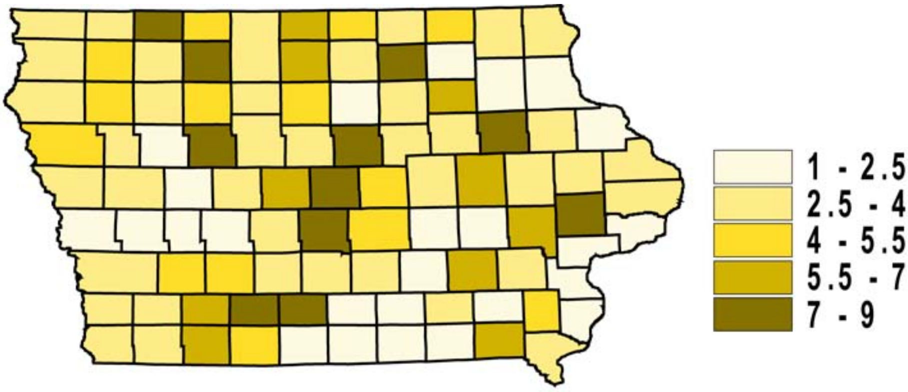


Map A.16. Total Acres Installed (CRP)

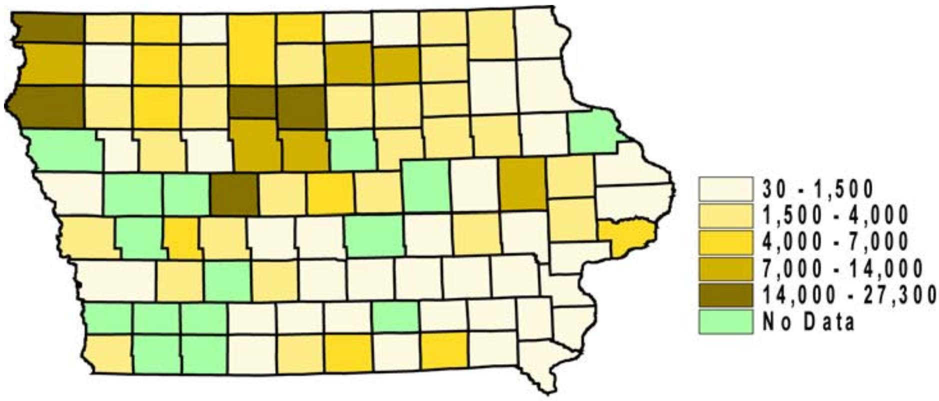


Nutrient Management

Map A.17. Average Cost per Acre

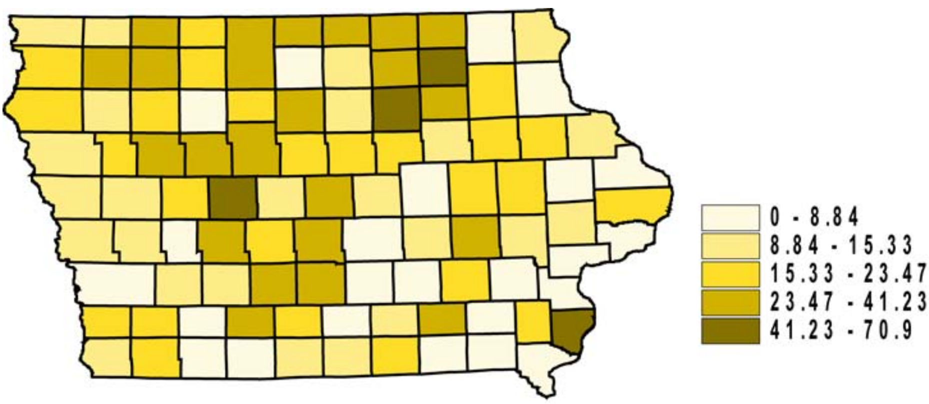


Map A.18. Total Acres Implemented (EQIP)

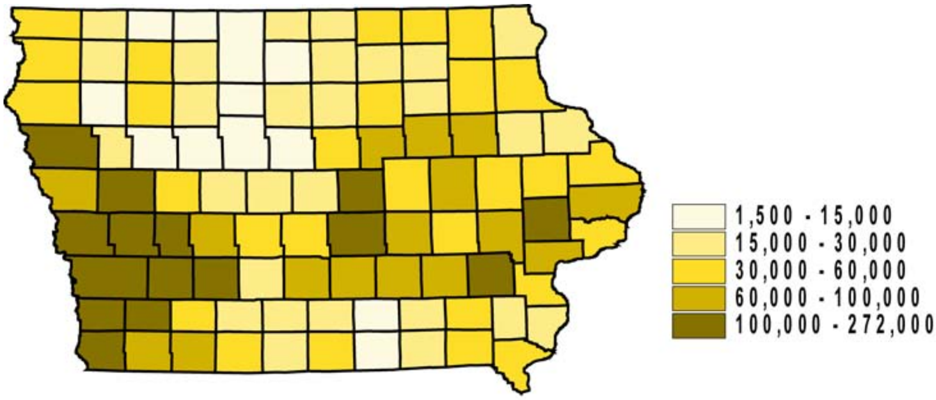


No Till

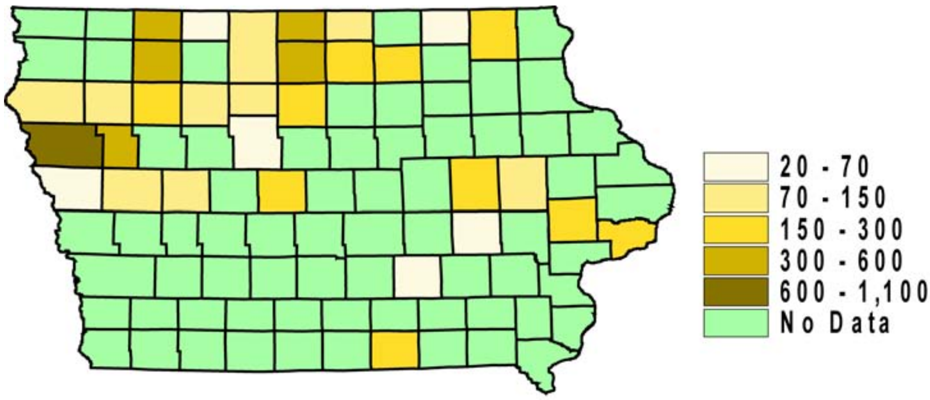
Map A.19. Average Cost per Acre



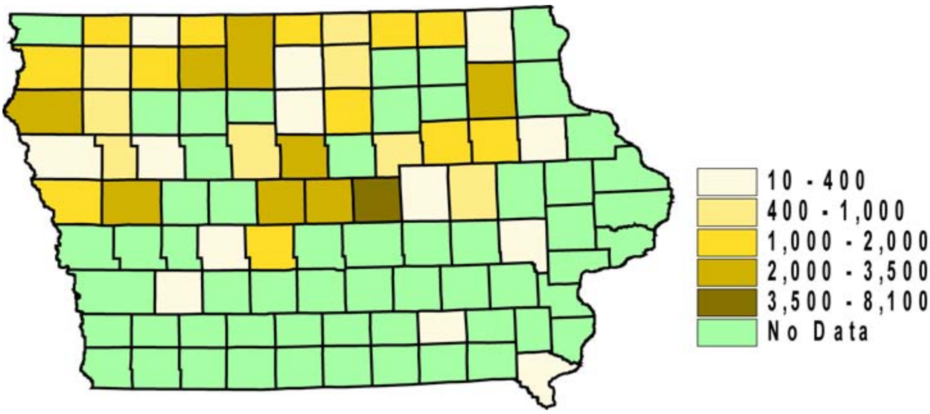
Map A.20. Total Acres (CTIC)



Map A.21. Total Acres Implemented (IFIP)



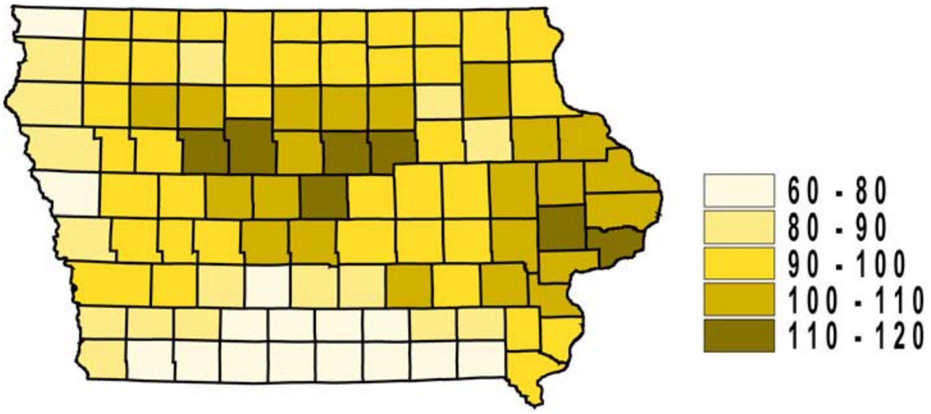
Map A.22. Total Acres Implemented (EQIP)



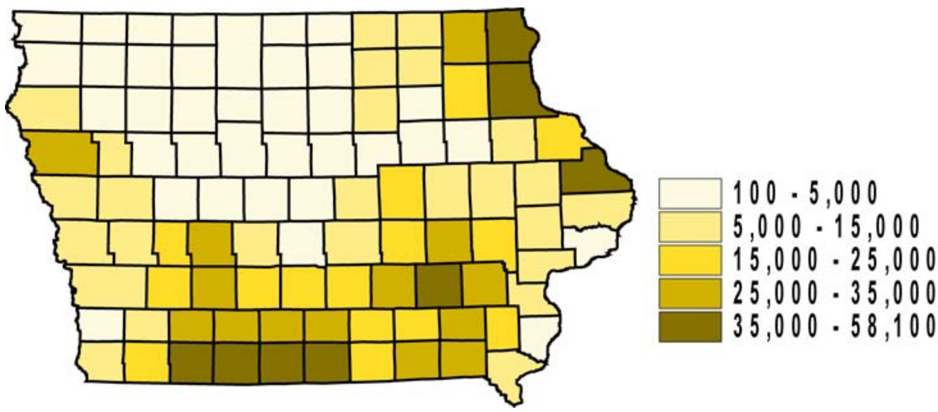
* Note: IFIP and EQIP were not used for no till average cost estimates; maps of the no till acres implemented under the programs are listed for comparison to CTIC.

General CRP

Map A.23. Average Annual Rental Payment per Acre



Map A.24. Total Acres (CRP)



APPENDIX B. SWAT MODEL CALIBRATION AND VALIDATION

B.1 The Soil and Water Assessment Tool (SWAT)

In SWAT, surface runoff from daily rainfall is estimated with the modified SCS curve number method (Mishra and Singh, 2003), which estimates the amount of runoff based on local land use, soil type, and antecedent moisture condition. The Green-Ampt method (Green and Ampt, 1911) of estimating infiltration is an alternative option of estimating surface runoff and infiltration that requires sub-daily weather data. Melted snow is treated the same as rainfall for estimating runoff and percolation. Channel routing is simulated using either the variable-storage method or the Muskingum method; both methods are variations of the kinematic wave model (Chow et al., 1988). Three methods of estimating potential evapotranspiration are available: Priestley-Taylor (Priestley and Taylor, 1972), Hargreaves (Hargreaves and Samani, 1985), and Penman-Monteith (Allen et al., 1989).

Erosion and sediment yield are estimated for each HRU with the Modified Universal Soil Loss Equation (MUSLE) (Williams, 1995). The channel sediment routing equation uses a modification of Bagnold's sediment transport equation (Bagnold, 1977), which estimates the transport concentration capacity as a function of velocity. The model either deposits excess sediment or re-entrains sediment through channel erosion depending on the sediment load entering the channel.

A generic crop growth submodel used in SWAT is a simplified version of the crop growth functions developed for the Environmental Impact Policy Climate (EPIC) model (Arnold and Forher, 2005; Gassman et al., 2007). A wide range of crop rotations can be simulated in the model, as well as different grassland and forest systems. Yields and/or biomass output are estimated at the HRU level in SWAT. SWAT simulates the complete nutrient cycle for nitrogen and phosphorus. The nitrogen cycle is simulated using five different pools; two are inorganic forms (ammonium and nitrate) while the other three are organic forms: fresh, stable, and active (Figure B.1). Similarly, SWAT monitors six different pools of phosphorus in soil; three are inorganic forms and the rest are organic forms (Figure B.2). Mineralization, decomposition, and immobilization are important parts in both cycles. These processes are allowed to occur only when the temperature of the soil layer exceeds 0ºC.

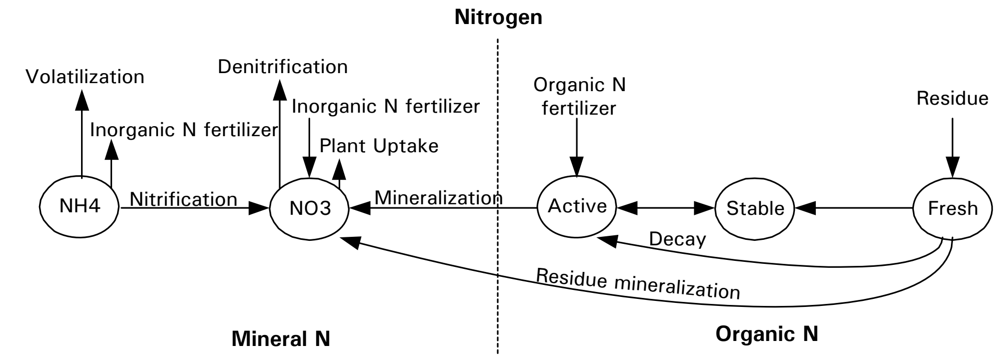


Figure B.1 Nitrogen Cycle as Simulated in SWAT (Adaption from SWAT Theoretical Document)

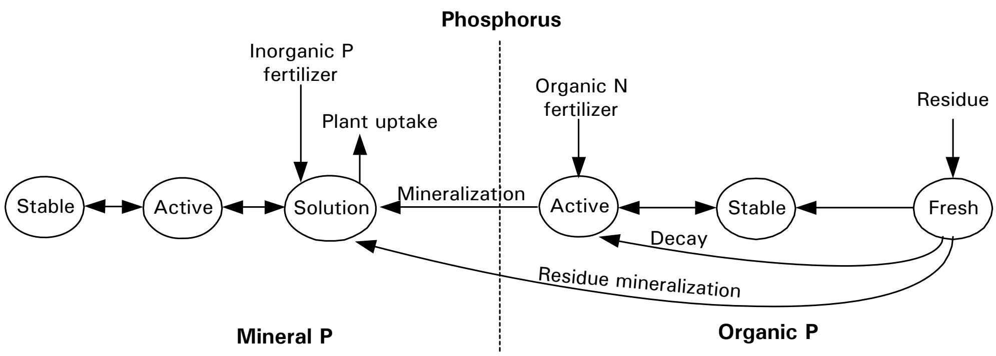


Figure B.2 Phosphorus Cycle as Simulated in SWAT (Adapted from SWAT Theoretical Document)

Nitrate export from runoff, lateral flow, and percolation are estimated as products of the volume of water and the average concentration of nitrate in the soil layer. Organic N and

Organic P transport with sediment is calculated with a loading function developed by McElroy et al. (1976) and modified by Williams and Hann (1978) for application to individual runoff events. The loading function estimates daily Org N and P runoff loss based on the concentrations of constituents in the top soil layer, the sediment yield, and an enrichment ratio. The amount of soluble P removed in runoff is predicted using labile P concentration in the top 10 mm of the soil, the runoff volume, and a phosphorus soil partitioning coefficient. In-stream nutrient dynamics are simulated in SWAT using the kinetic routines from the QUAL2E in-stream water quality model (Brown and Barnwell, 1987).

A detailed theoretical description of SWAT and its major components can be found in Neitsch et al. (2002). SWAT has been widely validated across the U.S. and in other regions of the world for a variety of applications, including hydrologic, pollutant loss, and climate change studies. An extensive set of SWAT applications are documented in Gassman et al. (2007).

B.2 Calibration and Validation: Hydrology and Streamflow

Calibration and validation of water quality models are typically performed with data collected at the outlet of a watershed. Daily streamflow data were collected from the U.S. Geological Survey website (USGS, 2007). Table B.1 lists all USGS gauging stations, which were used for the calibration and validation of the SWAT model for streamflow.

Table B.1. USGS Gauging Stations used in The SWAT Streamflow Calibration

Watershed	USGS station #	USGS station name	Latitude	Longitude	Drainage Area (km^2)
Boyer	6609500	Boyer River at Logan, IA	41.6425	-95.7828	2,256
Des Moines	5490500	Des Moines River at Keosauqua, IA	40.7278	-91.9596	36,358
Floyd	6600500	Floyd River at James, IA	42.5767	-96.3114	2,295
Iowa	5465500	Iowa River at Wapello, IA	41.1781	-91.1821	32,375
Little Sioux	6607500	Little Sioux River near Turin, IA	41.9644	-95.9728	9,132
Maquoketa	5418500	Maquoketa River near Maquoketa, IA	42.0834	-90.6329	4,022
Monona	6602400	Monona-Harrison Ditch near Turin, IA	41.9644	-95.992	2,331
Nishnabotna	6810000	Nishnabotna River above Hamburg, IA	40.6325	-95.6256	7,268
Nodawy	6817000	Nodaway River at Clarinda, IA	40.7394	-95.013	1,974
Skunk	5474000	Skunk River at Augusta, IA	40.7537	-91.2761	11,168
Turkey	5412500	Turkey River at Garber, IA	42.74	-91.2618	4,002
Upper Iowa	5388250	Upper Iowa River near Dorchester, IA	43.4211	-91.5088	1,994
Wapsipinicon	5422000	Wapsipinicon River near De Witt, IA	41.767	-90.5349	6,050

The calibration process was initiated by calibrating the water balance and streamflow for average annual conditions. Once the water balance and annual streamflow were considered correctly calibrated, the monthly calibration process was performed. Baseflow is an important component of the streamflow and had to be calibrated before the model was fully calibrated for streamflow and other components. An automated digital filter technique (Arnold and Allen, 1999) was used to separate baseflow from the measured streamflow. This approach estimated the baseflow to be centered on 60% for most of the watersheds except for Nodaway and Skunk where it is around 50% of the streamflow on an average annual basis for the period 1986 to 2005. SWAT was then executed for a total simulation period of 20 years, which includes 1986-1995 as a calibration period and 1996-2005 as a validation period. Parameter adjustment was performed only during the calibration period; the validation process was performed by simply executing the model for the different time period using the previously calibrated input parameters. The streamflow calibration process varied several SWAT hydrologic calibration parameters within their acceptable ranges to match the model predicted baseflow fraction, average annual streamflow, and monthly streamflow time series with corresponding measured values. These parameters include the curve number (CN2), soil available water capacity (SOL_AWC), evaporation compensation coefficient (ESCO), groundwater delay (GW_DELAY), groundwater recession coefficient (GW_ALPHA), surface runoff lag coefficient (SURLAG), and snow parameters. A model was also attempted to calibrate for the daily streamflow.

The model predictions were evaluated for both the calibration and validation periods using two statistical measures: coefficient of determination (R^2) and Nash-Sutcliffe simulation efficiency (E). The R^2 value is an indicator of strength of relationship between the measured and simulated values. The E value measures how well the simulated values agree with the measured values. The model prediction is considered unacceptable if the R^2 values are close to zero and the E values are less than or close to zero. If the values equal one, the model predictions are considered perfect. Generally, R^2 and E values greater than 0.5 are considered acceptable; however, explicit standards have not been specified for assessing model predictions using these statistics. Results of the calibration and validation of the SWAT model for streamflow for all 13 watersheds are presented in Table B.3. Statistical evaluation of the streamflow simulation underscore that SWAT accurately replicated the streamflows for each watershed in both the calibration and validation periods.

B.3 Calibration and Validation: Water Quality

Water quality data such as nitrate, Org N, Org P, and Min P were collected from the Environmental Protection Agency (EPA) database called STORET (STOrage and RETrieval). Iowa’s STORET database is managed by the water monitoring section of the Iowa Geological Survey Bureau. These water quality samples have been collected under the Enhanced Ambient Surface Water Monitoring program from 2000 on a monthly basis. Table B.2 lists STORET water quality measurement sites that were used in this study for the calibration and validation of the SWAT model for water quality parameters for all 13 Iowa watersheds. All STORET stations may not necessarily be at the watershed outlet all the time. So, the regression equations were developed at stations that are closed to the watershed outlets. Then, these equations were used in conjunction with the streamflow data at the watershed outlet to estimate water quality data at the watershed outlet and were used in the model verification.

Table B.2. STORET Water Quality Measurement Stations Points used in this Study

Watershed	STORET ID #	STORET station name
Boyer	10430001	Boyer River near Missouri Valley
Des Moines	10560001	Des Moines River near Keokuk
Floyd	10750001	Floyd River near Sioux City
Iowa	10580001	Iowa River at Columbus Junction
Little Sioux	10970001	Little Sioux River near Smithland
Maquoketa	10490002	Maquoketa River near Maquoketa
Monona	10670001	Monona-Harrison Ditch
Nishnabotna	10360001	East Nishnabotna River near Shenandoah
Nodaway	10730001	West Nodaway River near Shambaugh
Skunk	10620001	South Skunk River near Oskaloosa
Turkey	10220001	Turkey River near Garber
Upper Iowa	10030001	Upper Iowa River near Dorchester
Wapsipinicon	10820001	Wapsipinicon River at De Witt

These monthly samples of water quality data were extrapolated into continuous monthly data using the U.S. Geological Survey (USGS) Load Estimator (LOADEST) regression model (Runkel et al., 2004). LOADEST estimates constituent loads in streams and rivers by developing a regression model, given a time series of streamflow, constituent concentration, and additional data inputs. LOADEST is based on two previous models: LOADEST2 (Crawford, 1996) and ESTIMATOR (Cohn et al., 1989). The model is well documented in scientific publications and is accepted as a valid means of calculating annual solute load from a limited number of water quality measurements. However, the load estimation process of the model is complicated by retransformation bias, data censoring, and non-normality. Similar uncertainties are also inherent in other approaches. For example, Ferguson (1986) reported that the rating curve estimates of instantaneous load are biased and may underestimate the true load by as much as 50%.

There are a lot of uncertainties associated with the measured water quality data in addition to the error associated with the measurement process. First of all, measurement data were not available at the watershed outlet and so needed to be extrapolated. Furthermore, once-a-month samples were used to generate continuous monthly and then annual water quality data. Therefore, this study did not attempt to calibrate and validate the SWAT model based on the measured water quality data, but attempted to verify the model prediction with seasonal trends and annual totals. Nutrient calibration parameters were adjusted to bring annual totals of the simulated values into close agreement with the measured data. Model parameters used in the calibration were initial soil nutrient concentrations, biological mixing efficiencies, nitrogen and phosphorus percolation coefficients, phosphorus soil partitioning coefficient, residue decomposition factor, and in-stream nutrient transformation parameters.

Table B.3. Statistical Evaluation of The SWAT2005 Streamflow Calibration and Validation Results

	Statistical Evaluation							
	Annual				Monthly			
	Calibration (1986-1995)		Validation (1996-2005)		Calibration (1986-1995)		Validation (1996-2005)	
Iowa Watersheds	R^2	E	R^2	E	R^2	E	R^2	E
Boyer	0.89	0.84	0.8	0.78	0.68	0.66	0.75	0.75
Des Moines	0.98	0.92	0.78	0.65	0.81	0.78	0.67	0.65
Floyd	0.8	0.68	0.64	0.46	0.45	0.41	0.59	0.55
Iowa	0.98	0.97	0.63	0.88	0.84	0.83	0.69	0.74
Little Sioux	0.9	0.88	0.71	0.46	0.72	0.69	0.75	0.59
Maquoketa	0.91	0.89	0.83	0.74	0.79	0.79	0.77	0.76
Monona	0.86	0.83	0.49	0.52	0.55	0.51	0.52	0.4
Nishnabotna	0.95	0.94	0.88	0.84	0.73	0.72	0.84	0.83
Nodaway	0.9	0.9	0.9	0.88	0.8	0.79	0.84	0.82
Skunk	0.99	0.98	0.98	0.95	0.93	0.93	0.87	0.87
Turkey	0.95	0.92	0.88	0.8	0.8	0.79	0.73	0.71
Upper Iowa	0.88	0.83	0.65	0.52	0.81	0.79	0.7	0.59
Wapsipinicon	0.96	0.95	0.65	0.92	0.87	0.86	0.89	0.88

REFERENCES

Arnold, J.G., and N. Fohrer. 2005. SWAT2000: current capabilities and research opportunities in applied watershed modeling. *Hydrol. Process.* 19(3): 563-572.

Arnold, J.G., R. Srinivasan, R.S. Muttiah, and J.R. Williams. 1998. Larger area hydrologic modeling and assessment part I: model development. *J. Amer. Water Resour. Assoc.* 34(1): 73-89.

Baumer, O., P. Kenyon, and J. Bettis. 1994. MUUF v2.14 User's Manual. U.S. Department of Agriculture, Natural Resources Conservation Service, National Soil Survey Center, Lincoln, Nebraska.

Claassen, R., and M. Ribaudo. 2006. Conservation Policy Overview. Agricultural Resources and Environmental Indicators, 2006. U.S. Dept. Agr., Econ Res. Serv., EIB-16. Available at: http://www.ers.usda.gov/publications/arei/eib16/eib16_5-1.pdf.

Claassen, R., V. Breneman, S. Bucholtz, A. Cattaneo, R. Johansson, and M. Morehart. 2004. Environmental Compliance in U.S. Agricultural Policy: Past Performance and Future Potential. U.S. Dept. Agr., Econ Res. Serv., AER832, June.

Claassen, R., P. Johnston, M. Peters, and D. Gadsby. 2000. Compliance Provisions for Soil and Wetland Conservation. Agricultural Resources and Environmental Indicators, 2003. U.S. Dept. Agr., Econ Res. Serv., AH722. Available at: http://www.ers.usda.gov/publications/arei/ah722/arei6_3/AREI6_3compliance.pdf.

Conservation Technology Information Center (CTIC). Crop Residue Management Survey. Available at: http://www.conservationinformation.org/pdf/2006CRMSurvey SummaryHiRes.pdf.

Costanza, R., R. d'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, S. Naeem, K. Limburg, J. Paruelo, R.V. O'Neill, R. Raskin, P. Sutton, and M. van den Belt. 1997. The value of the world's ecosystem services and natural capital. *Nature* 387: 253-260.

Deacon, T.R., D.S. Brookshire, A.C. Fisher, A.V. Kneese, C.D. Kolstad, D. Scrogin, K.V. Smith, M. Ward, and J. Wilen. 1998. Research Trends and Opportunities in Environmental and Natural Resource Economics. *Environ. Resour. Econ.* 11(3-4): 383-397.

FAPRI-UMC. 2007. Estimating Water Quality, Air Quality, and Soil Carbon Benefits of the Conservation Reserve Program. FAPRI-UMC Report01-07, Food and Agricultural Policy Research Institute (FAPRI), Universtiy of Missouri-Columbia, Columbia, MO.

Gassman, P.W., M. Reyes, C.H. Green, and J.G. Arnold. 2007. The Soil and Water Assessment Tool: Historical development, applications, and future directions. *Trans. ASABE*. (forthcoming).

Gassman, P.W., E. Osei, A. Saleh, J. Rodecap, S. Norvell, and J. Williams. 2006. Alternative practices for sediment and nutrient loss control on livestock farms in northeast Iowa. *Agric. Ecosys. Environ.* 117(2-3): 135-144.

Gassman, P.W., S. Secchi, M. Jha, and L. Kurkalova. 2006. Upper Mississippi River Basin modeling system part 1: SWAT input data requirements and issues. In: *Coastal Hydrology and Processes* (Eds. V.P. Singh and Y.J. Xu), pp. 103-115. Water Resources Publications, Highland Ranch, CO.

Gassman, P.W., J.R. Williams, V.R. Benson, R.C. Izaurralde, L.M. Hauck, C.A. Jones, J.D. Atwood, J.R. Kiniry, and J.D. Flowers. 2005. Historical Development and Applications of the EPIC and APEX models. Working paper 05-WP 397. Ames, Iowa: Center for Agricultural and Rural Development, Iowa State University. Available at: http://www.card.iastate.edu/publications/DBS/PDFFiles/05wp397.pdf.

Gassman, P.W., T. Campbell, S. Secchi, M. Jha, and J. Arnold. 2003. The i_SWAT software package: a tool for supporting SWAT watershed applications. In: SWAT2003: The 2nd International SWAT Conference, 1-4 July, Bari, Italy. Instituto di Ricerca sulle Acque, IRSA-CNR, Bari, Italy. pp. 229-234.

Heimlich, R., D. Gadsby, and P. Johnston. 2000 a. Overview of Conservation Programs and Expenditures. Agricultural Resources and Environmental Indicators, 2003. U.S. Dept. Agr., Econ Res. Serv., AH722. Available at: http://www.ers.usda.gov/publications/arei/ah722/arei6_1/AREI6_1consoverview.pdf.

Heimlich, R., D. Gadsby, Claassen R., and Wiebe K. 2000 b. Wetlands Programs. Agricultural Resources and Environmental Indicators, 2003. Ch 6.5. U.S. Dept. Agr., Econ Res. Serv., AH722. Available at: http://www.ers.usda.gov/publications/arei/ah722/arei6_5/AREI6_5wetlands.pdf.

Hyberg, B., G. Frisvold, and P. Johnston. 1997. Conservation Compliance. Agricultural Resources and Environmental Indicators, 1996-97. pp. 297-309. U.S. Dept. Agr., Econ Res. Serv., AH712. Available at: http://www.ers.usda.gov/publications/arei/ah712/AH7126-4.PDF.

Iowa Department of Agriculture and Land Stewardship (IDALS). 2005. 2006 Budget Request-Infrastructure. Available at: http://www.legis.state.ia.us/lsadocs/SC_Materials Dist/2005/SDDLR015.PDF.

Iowa Department of Natural Resources (IDNR). 2001. "Iowa's water—Ambient monitoring program." Water fact sheet 2001-12. Iowa City, Iowa.

Iowa Environmental Mesonet (IEM). 2007. NWS COOP. Iowa Environmental Mesonet, Department of Agronomy, Iowa State University, Ames, Iowa. Available at: http://mesonet.agron.iastate.edu/COOP/.

Jha, M., P.W. Gassman, S. Secchi, R. Gu, and J. Arnold. 2004. Impact of watershed subdivision level on flows, sediment loads, and nutrient losses predicted by SWAT. *J. Amer. Water Resour. Assoc.* 40(3): 811-825.

Lee K.H., T.M. Isenhart, R.C. Schultz, and S.K. Mickelson. 1999. Nutrient and sediment removal by switchgrass and cool-season grass filter strips in Central Iowa, USA. *Agrofor. Sys.* 44:121-132.

Montgomery, W.D. Markets in licenses and efficient pollution control programs. 1972. *J. Econ. Theory* 5: 395-418.

Nusser, S. M., and J.J. Goebel. 1997. The national resources inventory: a long-term multisource monitoring program. *Environ. Ecol. Stat.* 4: 181-204.

Potter, S., S. Andrews, J. Atwood, R. Kellogg, J. Lemunyon, L. Norfleet, D. Oman. 2006. Model Simulation of Soil Loss, Nutrient Loss, and Change in Soil Organic Carbon Associated with Crop Production. U.S. Dept. Agr., Natural Resources Conservation Service. Washington, D.C.

Ribaudo, M.O. 1986. Consideration of off-site impacts in targeting soil conservation programs. *Land Econ.* 62: 402–11. 1989. Targeting the conservation reserve program to maximize water quality benefits. *Land Econ.* 65: 320–32.

Ribaudo, M., R. Horan, M. Smith. 1999. Economics of Water Quality Protection from Nonpoint Sources: Theory and Practice. U.S. Dept. Agr., Econ Res. Serv., AER782, Nov.

Ropke, I. Trends in the development of ecological economics from the late 1980s to the early 2000s. 2005. *Ecol. Econ*. 55: 262-290.

Secchi, S., P.W. Gassman, M. Jha, L. A. Kurkalova, H. Feng, T. Campbell, and C.L. Kling. 2007. The cost of cleaner water: assessing agricultural pollution reduction at the watershed scale. *J. Soil Water Conserv*. 62.1 (Jan-Feb 2007): 10(12).

Takasugi, P. 2006. Idaho State Department of Agriculture: Annual Report 2006. Idaho State Department of Agriculture, Boise, ID.

U.S. Department of Agriculture (USDA). 2006. Conservation and the Environment. Available at: http://www.usda.gov/documents/FarmBill07consenv.pdf.

U.S. Department of Agriculture, Farm Service Agency (USDA/FSA). 2006. Estimating Wildlife Response to the Conservation Reserve Program: Bobwhite and Grassland Birds. Available at: ftp://ftp-fc.sc.egov.usda.gov/NHQ/nri/ceap/quailandsongbirds.pdf.

———. 2007. General Conservation Reserve Program fact sheet. Available at: http://www.fsa.usda.gov/pas/publications/facts/html/crp03.htm.

U.S. Department of Agriculture, Economic Research Service (USDA/ERS). 2002. Conservation and Environmental Policy: Questions and Answers. Available at: http://www.ers.usda.gov/briefing/ConservationAndEnvironment/Questions/consenvwq4.htm.

U.S. Department of Agriculture, Natural Resources Conservation Services (USDA/NRCS). 2003. Environmental Quality Incentives Program: Benefit Cost Analysis. Final Report, USDA, Natural Resources Conservation Service.

———. 2007. Total EQIP Program Data, 1997-2001 Practices by State. Available at ftp://ftp-fc.sc.egov.usda.gov/Economics/Costs/EQIP_installed_truecost.xls.

U.S. Department of Agriculture, Natural Resources Conservation Services (USDA/NRCS). 2007a. National Resources Inventory. Available at: http://www.nrcs.usda.gov/TECHNICAL/NRI/.

U.S. Department of Agriculture, Natural Resources Conservation Services (USDA/NRCS). 2007b. Conservation effects assessment project (CEAP). Available at: http://www.nrcs.usda.gov/Technical/nri/ceap/

Williams, J. R. 1990. The erosion productivity impact calculator (EPIC) model: a case history. *Phil. Trans. R. Soc. Lond.* 329: 421-428.

Williams, J.R., W.L. Harman, M. Magre, U. Kizil, J.A. Lindley, G. Padmanabhan, and E. Wang. 2006. APEX feedlot water quality simulation. *Trans. ASAE* 49(1): 61-73.

Williams, J.R. and R.C. Izaurralde. 2006. The APEX model. In *Watershed Models*, 437-482. Singh, V.P. and D.K. Frevert, ed. CRC, Taylor & Francis.

Wischmeier, W. H. and D. D. Smith. 1978. Predicting rainfall erosion losses – A guide to conservation planning. USDA Handbook No. 537. Washington, D.C.

References (Appendix B)

Allen, R.G., M.E. Jensen, J.L. Wright, and R.D. Burman. 1989. Operational estimates of evapotranspiration. *Agron. J.* 81: 650-662.

Arnold, J.G., and P.M. Allen. 1999. Automated methods for estimating baseflow and groundwater recharge from stream flow records. *J. Amer. Water Resour. Assoc.* 35(2): 411-424.

Arnold, J.G., and N. Fohrer. 2005. SWAT2000: current capabilities and research opportunities in applied watershed modeling. *Hydrol. Process.* 19(3): 563-572.

Bagnold, R.A. 1977. Bedload transport in natural rivers. *Wat. Resour. Res.* 13: 303-312.

Brown, L.C., and T.O. Barnwell, Jr. 1987. The Enhanced Water Quality Models QUAL2E and QUAL2E-UNCAS Documentation and User Manual. EPA Document EPA/600/3-87/007. USEPA, Athens, GA.

Chow, V.T., D.R. Maidment, and L.W. Mays. 1988. *Appl. Hydrol.* New York, NY: McGraw-Hill.

Cohn, T.A., L.L. Delong, E.J. Gilroy, R.M. Hirsch, and D.K. Wells. 1989. Estimating constituent loads. *Wat. Resour. Res.* 25(5): 937-942.

Crawford, C.G. 1996. Estimating mean constituent loads in rivers by the rating-curve and flow duration, rating-curve methods. Unpublished PhD diss. Bloomington, IN: Indiana University.

Ferguson, R.I. 1986. River loads underestimated by rating curves. *Wat. Resour. Res.* 22(1): 74-76.

Gassman, P.W., M. Reyes, C.H. Green, and J.G. Arnold. 2007. The Soil and Water Assessment Tool: Historical development, applications, and future directions. *Trans. ASABE*. (forthcoming).

Green, W.H., and G.A. Ampt. 1911. Studies on soil physics, 1: The flow of air and water through soils. *J. Agric. Sci.* 4: 11-24.

Hargreaves, G.H., and Z.A. Samani. 1985. Reference crop evapotranspiration from temperature. *Appl. Eng. Agric*. 1: 96-99.

McElroy, A.D., S.Y. Chiu, J.W. Nebgen, A. Aleti, and F.W. Bennett. 1976. Loading functions for assessment of water pollution from nonpoint sources. Environmental Protection Technology Services, EPA 600/2-76-151.

Mishra, S.K., and V.P. Singh. 2003. *Soil Conservation Service Curve Number (SCS-CN) Methodology*. Dordrecht, The Netherlands: Kluwer Academic Publishers.

Neitsch, S.L., J.G. Arnold, J.R. Kiniry, and J.R. Williams. 2002. Soil and Water Assessment Tool Theoretical Documentation, Version 2000 (Draft). Blackland Research Center, Texas Agricultural Experiment Station, Temple, Texas. Available at: http://www.brc.tamus.edu/swat/ swat2000doc.html.

Priestley, C.H.B., and R.J. Taylor. 1972. On the assessment of surface hear flux and evaporation using large-scale parameters. *Mon. Weather Rev*. 100: 81-92.

Runkel, R.L., C.G. Crawford, and T.A. Cohn. 2004. Load Estimator (LOADEST): A FORTRAN Program for Estimating Constituent Loads in Streams and Rivers. U.S. Geological Survey Techniques and Methods Book 4, Chapter A5. Available at: http://pubs.usgs.gov/tm/2004/tm4A5.

U.S. Geological Survey (USGS). 2007. “USGS Real-Time Water Data for the Nation.” Available at http://waterdata.usgs.gov/nwis/rt.

Williams, J.R. 1995. The EPIC model. In *Computer Models of Watershed Hydrology*, 909-1000. Singh, V.P., ed. Water Resources Publications.

Williams, J.R., and R.W. Hann. 1978. Optimal operation of large agricultural watersheds with water quality constraints. Technical Report No. 96, Texas Water Resources Institute, Texas A&M University, College Station, TX.

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Chapter 4

PUBLIC SUPPORT FOR RENEWABLE ELECTRICITY: THE IMPORTANCE OF POLICY FRAMING*

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ABSTRACT

Individuals' contribution to electricity generation based on renewable energy sources can be channelled in two ways. The "green" market approach relies on an 'unconditional' contribution to renewable power while the certificate scheme represents a corresponding 'conditional' support (i.e., I can only contribute if the scheme is at place, and if so many others will also contribute). In both systems the support to renewable power is made possible through a price premium paid for these types of energy sources. In this chapter we draw on the economics literature on individual contributions to public goods and empirically test the overall hypothesis that the framing of renewable power support in a 'conditional' and an 'unconditional' scenario, respectively, will tend to trigger different types of moral deliberations. In the former case the deliberations concern mainly the division of efforts between individuals, while the deliberations in the latter case relate more to the characteristics of the public good in question and the perceived personal responsibility and ability to contribute to this good. This implies also that the variables determining the willingness to accept price premiums for renewable power may differ across schemes considered. We analyze the responses to dichotomous willingness to pay (*WTP*) questions from two different versions of a postal survey sent out to 1200 Swedish house owners. A random effects binary logit model is applied, and the estimated marginal effects support the notion that different types of factors tend to dominate choices depending on the support scheme considered. From these results follow a number of important implications for measures undertaken to increase the public's valuation of renewable power as well as the legitimacy of measures to increase renewable power production.

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1. INTRODUCTION

An essential component of the European Union's energy policy is the promotion of renewable energy sources in its member countries, and the Renewable Energy Directive (2001/77/EC) outlines quantitative goals for the development of renewable energy in each country. The Directive provides however substantial freedom on the part of national governments to select the policies needed to fulfil these goals. In countries where the electricity markets have been deregulated different ways of making consumers pay a price premium to support the still more expensive – but supposedly more environmentally benign – renewable power sources (such as wind, biofuels etc.) are becoming increasingly common.

Sweden provides an illustrative example of this overall picture. As a result of the deregulation of the electricity market in 1996 electricity consumers can now choose to sign contract with any of the electricity suppliers connected to the grid, and in this way product differentiation has become a way of attracting consumers. In order to facilitate "green" consumer choice in the market, the Swedish Society for Nature Conservation (SSNC) initiated a system for the labelling of "green" electricity. However, few Swedish households have been active in the "green" electricity market, and overall "green" consumer choices have not promoted enough of new renewable power.[2] Most of the "green" electricity sold has instead been purchased by state enterprises such as the Swedish railroad companies SJ and Green Cargo, and a few private firms (e.g., McDonalds). Partly as a result of the limited success of the voluntary "green" electricity market, a Swedish system for renewable energy certificates was introduced in 2003 and since then all households (as well as a majority of the non-residential electricity consumers) are obliged to pay a premium for renewable power on their electricity bills.[3] Generators of renewable power are awarded a certificate for every MWh they generate. These can then be sold and the distributors are obliged by law to purchase certificates that correspond to a certain percentage of their electricity consumption. This quota obligation will increase annually and the current (2007) share corresponds to 15 percent.[4]

Both the above approaches towards promoting environmentally benign power sources rely on private contributions to the same public good – the environmental benefit of replacing non-renewable power – but it is less clear that people in the two cases apply the same type of reasoning process when deciding whether they are willing to pay a premium to ensure public good provision or not. The "green" market approach relies largely on an 'unconditional' contribution to the public good while the certificate scheme represents a corresponding 'conditional' support (i.e., I can only contribute if the scheme is at place, and if so is the case many others will also contribute). The overall purpose of this chapter is to investigate to what extent the support for these two renewable power schemes can be explained by different types of factors and underlying motives.

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[2] In a survey investigation (Swedenergy, 1999), only 1 percent of the households stated that they purchase "green" electricity in spite of the fact that some "green" power contracts include only electricity from existing hydropower with sometimes modest price premiums of 0.5-1.0 öre per kWh (corresponding to about 0.06-0.12 US cents per kWh).

[3] It should be noted that the definition of "green electricity" in the labelling scheme differs slightly from the definition of "renewable electric power" in the certificates scheme, the most notable difference being that the latter scheme does not include large-scale hydropower.

[4] See Kåberger et al. (2005) for a critical discussion of the Swedish certificates system.

Methodologically we approach this research task by analyzing the responses to dichotomous willingness to pay (*WTP*) questions from two different versions of a postal survey sent out to a large sample of Swedish house owners. The overall hypothesis of the empirical investigation is that the framing of renewable power support in 'conditional' and 'unconditional' scenarios, respectively, will tend to activate different types of norms – informal rules about what one ought to do – among the respondents (Biel and Thogersen, 2007). This implies also that the variables determining the willingness to accept price premiums for renewable power may differ across schemes considered. Such differences may in turn provide different implications for the design of, for instance, information campaigns encouraging people to negotiate "green" electricity contracts on the one hand and for policy measures taken to increase legitimacy and public trust in mandatory certificate schemes on the other. In addition, a shift in policy focus from one support system to another may induce an increase in one set of moral deliberations but also 'crowd-out' other types of norm-based deliberations. This notion is, for instance, well in line with findings from the economic psychology literature, which show that public policy may 'crowd-out' specific morally based motivations (e.g., Frey and Jegen, 2001; Nyborg and Rege, 2003).

In previous economic research on private support to the provision of public goods a number of different behavioural models has been investigated, incorporating concepts such as selfishness, altruism, "warm-glow", social norms, fairness, reciprocity and commitment.[5] Many of these models represent extensions of the standard neoclassical model of consumer behaviour (e.g., Andreoni, 1990; Holländer, 1990), while other scholars assume that individuals are committed to different types of preference orderings (i.e., 'behavioural models') in different contexts (e.g., Nyborg, 2000; Sen, 1977; Margolis, 1982). In this chapter we assume the existence of different preference orderings for the same public good, but – in contrast to many previous studies – we also provide empirical tests of the hypothesis that different framings/contexts in which public good provision is presented will trigger different types of moral deliberations and stated behaviour.

In section 2 we discuss two sets of behavioural models for public good provision, which provide appropriate starting points for the analysis of the support for the two types of support system, and from which empirically testable hypotheses can be derived. Section 3 presents the survey design and the different variables included in the empirical analysis. Section 4 discus-ses the survey responses and presents the results from the econometric analysis of the binary choice questions included in the survey, while section 5 provides some concluding remarks.

2. Two Models of Individual Contributions to Public Goods

The methodological challenge in this chapter lies mainly in designing an empirical approach, which can be used to investigate differences in the public support for the two means of promoting renewable electric power sources; 'conditional cooperation' within the realms of a green certificate system and 'unconditional' contributions' in the electricity market. Thus, we do not empirically test a specific theoretical model of household contributions

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[5] See, for instance, Nyborg and Rege (2003) for a nice overview and discussion of a number of such models.

to a public good; instead we discuss the different features of two such models in an attempt to determine which of these features are of particular importance for our cases.

According to the welfare economics literature public goods, i.e., goods characterized by non-rivalry and non-excludability in consumption, are underprovided in the market place. This conclusion can be derived from a standard economic model of utility-maximizing behavior. As a starting-point in our analysis we specify an individual's *i* utility function as:

$$u_i = u_i(x_i, Z) \qquad i = 1,\ldots, N \tag{1}$$

where *Z* is the quantity of a pure public good and x_i represents *i*'s effort to contribute to the public good. It is assumed that

$$\frac{\partial u_i}{\partial Z} > 0 \quad \text{and} \quad \frac{\partial u_i}{\partial x_i} < 0 \tag{2}$$

Thus, x_i is a 'bad'. In the case of a *monetary* contribution to the public good, as is the case when consumers pay a higher price for renewable electric power, this specification amounts to the standard consumer model in which the individual derives utility from the own consumption of private goods as well as from the public good. Bergstrom et al. (1986) show that this type of model specification implies an equilibrium that is not Pareto-efficient. Because of the indivisibility and non-excludability properties of *Z*, individual *i* will receive the benefits of any *Z* purchased by other individuals. This tends to diminish the incentives to contribute to *Z*. Since the 'renewable component' of the consumption of electricity (and any environmental goods attached to this) has strong public characteristics, the 'homo economicus' model of consumer behavior would predict very meager support for renewable power programs.

The above shows that the economic analysis of private contributions to public goods would need to consider the possibility of extending the traditional model to include, for instance, issues of fairness, altruism, commitment, and/or norms (e.g., Nyborg and Rege, 2003). For our purposes it is useful to focus the attention on two different model extensions that in various ways incorporate the interplay between economic and moral-based motivation; we call these the 'reciprocity model' and the 'self image model', respectively.

The reciprocity model considered here was originally outlined by Sugden (1984).[6] He essentially assumes that individuals pursue their own self-interest – as modeled in equation (1) – subject to a moral constraint saying that if everyone else contributes [...] to the production of a public good, you must do the same. This is Sugden's reciprocity principle, and it can be derived by extending the above simple 'homo economicus' model.

In Sugden's model the production of the public good is expressed as:

[6] In the experimental economics literature on public good provision alternative models of reciprocity are used (e.g., Fischbacher et al., 2001; Fehr and Falk, 2002; Sethi and Somanathan, 2003), but the underlying micro-economic structure of these models are often simplified so as to permit laboratory tests. One example is Rabin (1993), who assumes that each player in a public good game is assumed to choose a strategy that maximizes his utility, which simply comprises a 'material payoff' and a 'reciprocity payoff'.

$$Z = f\left(\sum_{i=1}^{N} \alpha_i x_i\right) \tag{3}$$

where α_i indicates the 'productivity' of individual *i* in transforming his/her effort to public good outputs. Individual *i* is a 'member' of a group of *N* individuals. If all *N* individuals contribute φ to the public good we can define the function $F(N,\varphi)$, which shows the specific amount of the public good that is produced given this particular contribution. We have thus:

$$F(N,\varphi) = f\left(\sum_{i=1}^{N} \alpha_i \varphi\right) \tag{4}$$

In order to derive the 'reciprocity principle' we can think of an individual *i* who considers what level of φ that would maximize his utility $u_i[\varphi, F(N,\varphi)]$. "If *i* could choose a single level of contribution for all [...], this is the level he would choose," (Sugden, 1984, p. 777). Let use define this optimal level as x_i^N; the reciprocity principle then states that *i* is obliged to contribute at least x_i^N to the public good provided that all other *N*-1 individuals do the same. Thus, if others do not contribute the individual does not need to that either, but he/she should not take a free ride if others are contributing.

Nyborg and Rege (2003) notes that Sugden's model is what one may refer to as a 'commitment model' of individual contributions to public goods (see also Sen, 1977); it is a departure from the 'homo economicus' model since it opens up the possibility for personal sacrifices. It is however worth noting that even though Sugden's model explicitly incorporates a moral constraint on behavior, this constraint 'only' guides behavior in the case of public goods that the individual cares for himself/herself. In the empirical section we find, however, that regardless of the support scheme chosen no evidence of such an 'egoistic public good provision' can be presented (see section 4.2). In the analysis we focus rather on the fact that according to Sugden's model the individual's moral deliberations concern mainly the division of efforts between individuals, and tend to relate less to the characteristics of the public good in question and the perceived personal responsibility and ability to contribute to the same good. This differs from the second model of public good contributions to be discussed now, namely the self-image model.

The self-image model draws on the literature on integrating norm-based behavior into neoclassical economic theory (Brekke et al., 2003; Nyborg et al., 2006). In this strand of research it is assumed that individuals have a preference for keeping a self-image as a responsible person, and contributions to public goods will be determined by how such behavior affects this self-image. We therefore consider an individual *i* with the utility function:

$$u_i = u_i\left[x_i, Z, S_i\left(E_i^Z\right)\right] \tag{5}$$

where S_i represents the individual's self-image as a morally responsible person, defined as a person who conforms to certain norms of responsible behavior (Brekke et al., 2003). This model assumes thus that individuals have preferences for a positive self-image, and self-image is therefore treated as an argument in the utility function. The more willing the individual is to acknowledge his/her own responsibility to contribute to the public good, the higher is S_i. Following Nyborg et al. (2006) S_i is assumed to be affected by E_i^Z, which is the individual's belief about the positive externalities arising from his/her contribution to the public good, *Z*. In a companion paper (Ek and Söderholm, 2005) we argue that E_i^Z will be determined by both the individual's (a) perception of his/her ability to affect the outcome in a positive way; and (b) assessment of the importance – the value – of the public good. In the literature the first of these points is sometimes summarized in the concept 'perceived consumer effectiveness' (PCE) (e.g., Ellen et al., 1991). In our case, PCE reflects the extent to which the individual perceives that his/her decision to pay for renewable electric power will in fact lead to an increase in the production of renewable energy sources, which in turn is more environmentally benign than other energy sources. The second point suggests that implicit in E_i^Z is also some valuation of the environmental benefits following the individual's choice. Even if individuals in general believe that their choices imply more renewable power generation and that this improves the environment, they may perceive the *importance* of this change differently. Overall this type of model specification is inspired by the way in which moral decision-making is understood in the field of social psychology, and in which awareness of consequences and ascription of responsibility are identified as important factors determining such decisions (Schwartz, 1970).

The model outlined in equation (5) suggests that from the individual's point of view there exists an explicit trade-off between increases in self image following a morally respon-sible decision and the costs in terms of increased efforts. Let us assume that a representative individual is considering whether to increase efforts from its initial level x_i^0 to $x_i^0 + \tau = x_i^1$ (where $\tau > 0$). The individual will pursue this if and only if:

$$u_i\left(x_i^0, Z^0, S_i^0\right) < u\left(x_i^1, Z^0, S_i^1\right) \tag{6}$$

where $S_i^1 - S_i^0 > 0$ and we assume that there are no *direct* personal benefits for the individual in terms of public good provision, i.e., *Z* remains unchanged.

In contrast to Sugden's reciprocity model the self-image model assumes that the individual's moral deliberations primarily concern his/her own personal responsibility to contribute to a specific good. According to our model specification the decision to take on such a responsibility implies increases in the individual's self image, which in turn is affected by the perceived external effects. Thus, here the individual is assumed to reflect explicitly on the impact on public good provision following his/her choice. Although others' behavior plays no direct role in this specification, one can easily integrate this by assuming the presence of *descriptive* norms, i.e., the individual perceives others' contributions as a guide to what is morally appropriate (e.g., Moscovici, 1985). Thus, one can assume that *S* is also an increasing function of the extent to which others contribute (e.g., Nyborg et al., 2006). Although both this approach and Sugden's approach predict that the individual's contribution will increase if he/she perceives that others also increase their contributions, there are major differences in the way in which interactions between individuals affect behavior. In the 'reciprocity model' one can think of contributions as conditional on expectations that others will contribute, while in the 'self-image model' (with others' behavior included) others' behavior play more the role of a compass to define ones own moral responsibility. In Ek and Söderholm (2005) we test for the presence of descriptive norms in the empirical context of Swedish household's willingness to purchase "green electricity", but find only meager support for this hypothesis. Instead the empirical results suggest that individuals who express that they face *prescriptive* social norms (i.e., norms that directly guide how one *ought to* act) in the sense that people who are close to them expect them to buy "green", are more inclined to report a positive willingness to pay a premium for "green" electricity. A likely explanation for this result, we argue, is that others' purchases of "green" electricity are (essentially) not observable (Nyborg et al., 2006), and others' behavior in general may therefore not play a role in the individual decision. For the above reasons – and to maintain clear-cut distinctions between the two model specifications – we choose here not to include perceptions about other's behavior in the 'self-image' model outlined in equation (5).

In the next section we discuss briefly the methodological approach of the investigation and outline a number of specific hypotheses to be tested empirically. Before proceeding, however, it is useful to make clear that we do not assume that in a specific situation people "turn on" one behavioral model and "turn off" all other models. We simply claim that the framing of the choice situation can to *various extents* trigger different types of moral deliberations, and the two models outlined above provide a useful starting point for testing this overall hypothesis.

3. Empirical Strategy, Survey Design and Variable Definitions

3.1 Choice Scenarios and Hypotheses

In May 2005, 1200 questionnaires were sent out to a sample of randomly drawn Swedish house owners. The main reason for directing the investigation towards house owners (and not, for instance, to people living in apartments) is that all house owners in Sweden have the possi-bility to actively choose between different electricity suppliers and contracts, and constitute thus a significant proportion of the potential market for differentiated electricity. House owners are therefore generally more familiar with some of the choice situations we intend to mimic in our study, and for many of them the electricity expenses constitute a significant proportion of total household expenditures.

All questionnaires included a section briefly describing the current green certificate system in Sweden such as the different energy sources eligible for certificates. The respon-dents were informed that the prevailing (2005) quota amounted to (about) 10 percent of electricity consumption, and as a result all households pay a certificate fee of approximately 3 Swedish öre per kWh (about 0.4 US cents per kWh) to finance this contribution to renewable power generation. After this the respondents were asked whether they were willing to support renewable power generation *beyond* the present (10 percent) quota stipulated in the certificate scheme, but half the sample (600 respondents) faced a choice scenario that we will refer to as the ‘conditional scenario’ while the remaining half (600 respondents) faced the so-called ‘unconditional scenario’. The wordings of the two scenarios (translated from Swedish) were:

Conditional Scenario

“We are now interested in the extent to which you are prepared to support a proposal for an increase in the generation of renewable power beyond the pre-sent share, 10 percent, through a raise in the certi-ficate fee for all electricity consumers. Envisage a situation in which the government has to choose between two options, A and B. Alternative A represents electricity generated with the existing share, 10 percent, of renewable energy sources, while alternative B represents electricity genera-ted with a higher share of renewable power sources, 20 percent, primarily at the expense of reduced nuclear power generation. “

Unconditional scenario

“We are now interested in the extent to which you are prepared to voluntarily support increased generation of renewable power, beyond the contribution you already provide by paying the certificate fee. Envisage that at the next time when you are to renew the contract with your electricity supply company, you will only be able to choose between two options, A and B. Alternative A represents the existing mix of power sources in Sweden today, 10 percent from renewable energy sources. If you choose alternative B you will pay a higher price for the electricity, but you also provide additional support to renewable power. Then electricity corresponding to 20 percent of your household’s total consumption of electricity will be produced by renewable energy sources.”

In the ‘conditional scenario’ the respondents were shown a table outlining the resulting mix of power generation sources in the A and B alternatives. All 1200 respondents then faced three different choices between alternatives A and B; the choices differed only with respect to the price premiums (i.e., the extra cost) paid for “green” electricity. The price premiums included in the three choice sets were 1, 3, and 5 öre per kWh, corresponding to approximately 1, 4 and 7 percent increases in the average household electricity price (including the prevailing certificate fee).[7] In the ‘conditional scenario’ the respondents were asked whether they would vote in favour of the B-alternative given the above price premiums incurred for all consumers. In the ‘unconditional scenario’ the choices confronting the respondent were presented in a traditional dichotomous willingness-to-pay framework, in which he/she either ticks the A-alternative (the status quo option) or the B-alternative (inducing higher prices for the household but increased contribution to renewable power).

In order to facilitate comparisons between the economic impacts of the different price bids, the choice sets were preceded by a simple calculation translating the price bids into an

[7] In comparison to other studies of willingness to pay for “green power” (e.g., Ek and Söderholm, 2005) we chose to include relatively low price premiums primarily because respondents are asked to contribute *in addition* to the certificate fee that they already pay.

annual total cost in SEK for the household. The calculations were based on the electricity consumption of an average house with and without electricity heating. Since the price bids were equally high in both sub-samples, this information did not differ across samples.

The choice sets were followed by a number of follow-up questions concerning the respondents' motives when choosing whether to support renewable power or not (see further section 4.2). In addition to the above, the questionnaire also collected information about the environmental attitudes of the respondents, in general and in particular with respect to the possible negative environmental and health impacts of non-renewable electric power production. Questions concerning the perceived responsibility of different actors (e.g., the state, the electricity companies, consumers, etc.) for increasing the share of renewable power sources were also included, as were questions about whether the respondent had prior experiences of renewable power. At the end of the questionnaire socio-economic information was collected.

As was noted above, we expect that the two different scenarios tend to induce different types of moral deliberations as outlined in the 'reciprocity' model and 'self-image' model, respectively. Specifically, the 'conditional scenario' (CS) is assumed to frame a situation in which the behaviour of the average individual is *relatively* more in line with the 'reciprocity model', and he/she is thus more inclined to set the own effort in relation to the contribution of others but does not necessarily reflect so much on, for instance, if and how the personal effort implies improved environmental quality. The 'unconditional scenario' (US) instead, we hypothesize, induces a behaviour that in *relative* terms resembles that of the 'self-image' model; the individual knows little about others contributions and engages instead in a moral deliberation primarily about the own responsibility to contribute and the impacts of contributing to the public good. Given this overall hypothesis, a number of specific hypotheses can be derived. We hypothesize that the probability that an individual chooses to express support for renewable power will be higher:

- the more people agree that the production of non-renewable power sources impose a (health and environmentally related) threat to society as a whole. This impact will be more pronounced in the 'unconditional scenario', since in the 'self-image' model people's self image is explicitly assumed to increase with their valuation of the external environmental benefits.
- the more people feel that their contribution will be *effective* in solving the external environmental and health problems associated with non-renewable power sources. This impact will be more pronounced in the 'unconditional scenario'.
- the more willing people are to assume a personal responsibility for contributing to renewable power. This impact is believed to be stronger in the 'unconditional scenario', which is assumed to induce such moral deliberations to a larger extent than the 'conditional scenario'.
- the more inclined people are to express preferences for reciprocity, stating a willingness to contribute only if others also contribute. We hypothesize that this influence on the choice will be more pronounced in the 'conditional scenario'.

Overall we believe it is fair to assert that overall our hypotheses are consistent with the view that in the case of a voluntary market for renewable power people tend to be more careful in evaluating the benefits of such a scheme, and the impact on these benefits of ones own decision to contribute. In the case of the renewable certificate scheme, however, the individual is confronted with a ready-made policy package for all (i.e., someone else has already made all the assessments), and if the policy makers are perceived to be trustworthy there is less need to deliberate on ones own responsibility and ability to contribute to the public good.

The two behavioural models outlined above both build on the assumption of norm-based behaviour. In the case of electricity purchases and personal voting behaviour – which are hard for most others to observe – it may be reasonable to assume the existence of so-called moral norms (e.g., Nyborg et al., 2003). In contrast to social norms, which are enforced by explicit approval or disapproval from others, the presence of a moral norm implies that individuals sanction themselves. The maintained hypothesis of the analysis is that moral norms are present, but in the empirical section we also explicitly test for the presence of (prescriptive) social norms communicated by people close to the respondent.

We also test for the impact of variables such as the importance of contributing to other environmental public goods, prior experience of renewable power, different socio-economic factors (education, age, gender etc.), as well as, of course, the electricity price. For these variables, however, it is hard to state any *a priori* hypotheses as to whether the respective impacts ought to differ across the two scenarios. In the case of electricity price sensitivity one needs to recognize that ‘Economic Man’ is very much present in both the ‘reciprocity’ and the ‘self-image’ model; price increases are hypothesized to imply a reduced willingness to accept an increased reliance on renewable power, but we see no apparent reason for the existence of a significant difference in price responses in these two model specifications.

3.2 Variables Included in the Analysis

The binary choice between “increased renewable power” (B) and the “status quo” option (A) represents the dependent variable in the empirical investigation. The independent variables to be included in the model can be divided into five different main categories: (a) variables affecting the cost of purchasing “green” electricity; (b) factors influencing the extent to which purchases of “green” electricity gives rise to self-image improvements; (c) variables addressing the importance of reciprocity; (d) socio-economic characteristics; and (e) other variables.

As was noted above, we expect that the willingness to support increased renewable power should be decreasing with the level of the price premium associated with the B-alternative. As was noted above, three levels of price changes were included in the discrete choice situations presented to the respondents permitting thus estimations of the (average) price sensitivity of the responding households. Respondents were also asked about whether their homes are heated with electricity or not; this may be of profound importance for the impact of certain electricity price increases on the total (absolute) cost increase for households.

The improvement in self-image from choosing the B-alternative is determined by the perceptions of the personal responsibility to contribute to renewable power, the ability to affect the outcome in a positive way, as well as of their perception of the importance of the positive environmental externalities following the individual choice (E_i^Z). We expect that respondents that are more inclined to acknowledge a personal responsibility for reducing the negative environmental impact associated with electricity production are more likely to choose the B-alternative offered in the survey. The respondents were therefore confronted with the statement: “I feel a personal responsibility to contribute to increased production of renewable power,” and were asked to mark to what degree they agree or disagree with this statement on a scale ranging between 1 and 5 (where 1 corresponds to “disagree entirely” and 5 to “agree entirely”). The respondents were also asked to what extent they agreed to two statements intended to capture different aspects of PCE, the extent to which renewable power is perceived as more environmentally benign compared to non-renewable power and whether respondents believe that the higher price paid will actually imply increased production of renewable power. Five statements about the environmental impacts of non-renewable power were posed to the respondents in order to facilitate a test of whether the degree of environmental concern affects the willingness to support non-renewable power. Based on the responses we constructed an “environmental impact” index by adding the scores reported on each of the five statements; the index variable thus ranges between 5 and 25.

In order to measure the relative importance of reciprocity two different variables were used. First, in the follow-up questions to the choice sets, the respondents were asked whether they agreed or disagreed to the statement: “I can only consider paying a premium for renewable power if I know that all other electricity consumers also pay”. The responses were measured on a five-point scale with the end points “disagree entirely” (1) and “agree entirely” (5). Second, in another question we asked to what extent the respondents perceive that households (in general) have a responsibility to take measures towards increasing the share of renewable power generation. Even though this variable does not *explicitly* concern the issue of reciprocity we believe it will possess some reciprocity features and thus have different impacts on the choice probability in the two scenarios. In the US sample – based on the ‘self image model’ – people are assumed to be primarily concerned with their personal responsibility with few conditions attached to what others do, but in the CS sample the overall responsibility of all households is hypothesized to matter because of the presumed weight given to reciprocity in this case.

In order to test for the presence of a social norm the questionnaire included the following statement: “Important persons, who are close to me, expect me to contribute to increased production of renewable power”. The responses were measured on a five-point scale with the end points “disagree entirely” (1) and “agree entirely” (5). In our case, where an individuals’ choice between electricity with different environmental characteristics is difficult to observe for others, social sanctioning may be limited but if it is present it may perhaps be significant only between, say, people in the same family. We also included binary choice questions to obtain information about prior personal experience of renewable power sources (wind power, biofueled power, and hydropower), and a statement (to which respondents could agree or disagree along a five-point scale): “I would rather support environmental projects in other areas”.

The different socio-economic variables included in the questionnaire (and ultimately used in the econometric model estimated) were gender, age and the education level of the respondent. We do not have any a priori hypothesis about whether women or men should in general be more willing to pay a premium for renewable power. Earlier work suggests that the willingness to purchase "green" electricity often decreases with age. We also tested for the possibility that households with children (dummy 1/0 variable) are more or less prone to support renewable power, but our initial model estimations showed that this had no statistically significant impact on this choice, and the variable was removed from the estimations. Table 1 summarizes the variables used in the empirical investigation, including definitions, coding and some descriptive statistics based on the responses from each of the two sub-samples.

Table 1. Variables Included in the Analysis: Definitions and Descriptive Statistics

Variables	*Coding/definitions*	*Mean*	*Std. Dev.*	*Min*	*Max*
Dependent variable		US/CS	US/CS	US/CS	US/CS
Green choice	1 if "more renewable power", 0 otherwise	0.41/0.45	0.49/0.50	0.00/0.00	1.00/1.00
Cost of additional renewable power					
Electricity Price (-)	Price increases of 1, 3 and 5 öre per kWh	3.00/3.00	1.63/1.63	1.00/1.00	5.00/5.00
Electricity heating (-)	1 for electricity heated house, 0 otherwise	0.59/0.58	0.49/0.54	0.00/0.00	1.00/1.00
Self-image variables					
Personal responsibility (+)	1 for "disagree entirely", and 5 for "agree entirely"	3.81/3.14	1.28/1.27	1.00/1.00	5.00/5.00
Environmental impact (+)	Index ranging from 5 to 25 based on responses presented in Table A1	17.47/17.23	4.91/5.27	5.00/5.00	25.00/25.00
No impact on renewable production of price premium (-)	1 for "disagree entirely", and 5 for "agree entirely"	4.14/4.08	0.96/1.03	1.00/1.00	5.00/5.00
Renewable power not "green" (-)	1 for "disagree entirely", and 5 for "agree entirely"	2.60/2.50	1.26/1.20	1.00/1.00	5.00/5.00
Reciprocity variables					
Conditional cooperation (+)	1 for "disagree entirely", and 5 for "agree entirely"	3.20/3.44	1.48/1.45	1.00/1.00	5.00/5.00
Consumer responsibility (+)	1 for "disagree entirely", and 5 for "agree entirely"	2.95/2.98	1.35/1.34	1.00/1.00	5.00/5.00
Socio-economic variables					
Gender (?)	1 for female, 0 otherwise	0.24/0.27	0.43/0.44	0.00/0.00	1.00/1.00
Age (?)	Age in years	55/57	13.1/13.5	20/25	81/81
Education (?)	1 for university degree, 0 otherwise	0.32/0.33	0.47/0.47	0.00/0.00	1.00/1.00
Other variables					
Social norm (+)	1 for "disagree entirely", and 5 for "agree entirely"	2.49/2.40	1.88/1.12	1.00/1.00	5.00/5.00
Prior experience (?)	1 if personal experience of renewable power, 0 otherwise	0.46/0.47	0.50/0.50	1.00/1.00	5.00/5.00
Support to other areas (-)	1 for "disagree entirely", and 5 for "agree entirely"	3.25/3.41	1.16/1.04	1.00/1.00	0.00/0.00

It is worth noting that the underlying basic model for empirically analyzing our dichotomous responses is the random utility model that was originally set up by Hanemann (1984). The model is often a good approximation to any arbitrary utility specification and it is frequently adopted for econometric applications to contingent valuation binary choice questions. However, the model assumes that the marginal income is constant across the scenarios set up in the willingness to pay question and therefore eliminates income as a determinant of the choices.

4. EMPIRICAL RESULTS

4.1 Survey Responses

After two reminders we received 608 responses corresponding to an overall response rate of 51.6 percent.[8] After removing all questionnaires with incomplete answers, 564 questionnaires remained, 302 in the US sample and 262 in the CS sample. In the majority of the incomplete questionnaires respondents refused to state their preferred alternative in one or more of the choice sets. There were also respondents that rejected the lowest bid and did not state their opinion in the subsequent two choices (11 in the US sample and 9 in the CS sample) or that accepted only the highest bid without stating their preferred alternative in the preceding two choice sets associated with lower price premiums (5 in sample A and 4 in sample B). We interpreted these answers as either refusing all price bids (the first case) or as accepting all price bids (the second case).

So as to evaluate whether the results are reasonably representative, the socio-economic characteristics of the respondents need to be compared with the populations from which they were drawn. The proportion of women is relatively low in our sample, 26 percent, while the corresponding proportion among house owners in general equals 45 percent (Statistics Sweden, 2005). This may reflect the fact that men in general are more likely to be responsible for administering electricity bills, and choosing supplier. In the official statistics there exists directly comparable estimates either for the household income, age or the level of education among house owners in general to which we can compare our sample estimates. The proportion of people older than 65, the proportion of people with a university degree, as well as the reported household income were however similar to the ones reported in a similar recent study on Swedish house owners (Ek, 2005).

Before the choice scenario was outlined in the questionnaire the prevailing certificate system in Sweden was briefly introduced and described, and the respondents were asked to what extent they had prior knowledge of the system. In both samples about 70 percent responded that they had heard about the certificate system before reading the questionnaire, and 22 percent claimed that they knew well how the system functions. However, in neither of the two samples the degree of prior knowledge had statistically significant impacts on the probability of accepting to pay a price premium for increased renewable power, and for this reason this variable was not included in the binary choice analysis presented in section 4.2.

In the US sample, 40 percent of the respondents chose the status quo option in all three choices, while the corresponding share in the CS sample was 23 percent. However, the proportion that accepted all three price bids was slightly higher in the US sample (24 percent) than in the CS sample (20 percent). Based on the econometric results presented in section 5.2 we calculated the expected willingness to pay (*WTP*) for the stipulated increase in renewable

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[8] Three questionnaires were undeliverable and 18 respondents were unable to return the questionnaire due to death or severe illness.

power in each of the two samples. The estimates are based on the mean of the explanatory variables and they equal the sum of the estimated coefficients times their corresponding mean values divided by the estimated coefficient of the price premium variable.[9] The results show that the expected *WTP* is estimated at 2.9 öre per kWh in the US sample, and 2.5 öre per kWh in the CS sample. Thus, overall the average willingness to contribute monetarily to increased reliance on renewable power does not differ much between the two choice scenarios.

Table 2 is based on the responses to the statements immediately following the choice scenarios, and here the respondents were asked to indicate to what extent they agreed (or disagreed) to a number of statements concerning their motives when choosing whether or not to support increased renewable power. Specifically, Table 2 shows the share of the respondents that agreed (partly or entirely, i.e., marked "4" or "5") to each statement in each of the two scenarios, and the third column presents a 95 percent confidence interval for the differences in responses across scenarios.

Table 2. Support for Selected Statements Concerning the Motives Behind Choices Made

	US sample share	*CS sample share*	*95 % confidence interval for the difference (US-CS)*
I feel a personal responsibility to contribute to increased production of renewable power	0.50	0.45	0.002 – 0.074
Renewable power is not more environmentally benign than is non-renewable power	0.26	0.18	0.040 – 0.099
I can only consider paying a premium for renewable power if I know that all other electricity consumers do the same	0.52	0.59	-0.119 – -0.047
I would rather support environmental projects in other areas.	0.44	0.47	-0.077 – -0.005
Renewable power should not cost more than non-renewable power	0.68	0.57	0.057 – 0.127

In Table 2 we include only those statements where we found statistically significant differences (at the five percent level) in responses between the two scenarios. For the remaining statements in this section of the questionnaire we could not reject the null hypothesis of equal support for the respective statements regardless of scenario faced. Among these we find, for instance, the statement that a price premium does not guarantee increased production of renewable power and the statement capturing the presence of a social norm.

It should be noted that the reported differences in Table 2 do not in themselves tell us anything about whether we should reject (or not) our hypothesis about differences concerning what factors influences choices in the two scenarios. Nevertheless, the results do suggest that some – for our purposes – interesting framing effects seem to be present. Most importantly perhaps, we find that those confronted with the US sample are more likely to express a personal responsibility for contributing to increased renewable power, while the importance of reciprocity is given a higher weight among the respondents in the CS sample. We believe it is fair to conclude that this supports our conjecture that the different framings in the scenarios

[9] See Haab and McConnell (2002) for a detailed discussion of *WTP* calculations in dichotomous contingent valuation studies.

trigger different deliberations among those responding, but it remains an empirical question to what extent these motives provide important explanations to why people accept or reject the price bids in the two scenarios.

Table 2 also shows that the respondents in the US sample are more prone to agree with the statements that renewable power: (a) is not more environmentally benign than non-renewable power; and (b) should not cost more than non-renewable power. The theoretical discussion in section 2 provides no clear clues to why such differences would occur, but we believe that one plausible explanation could simply be that people tend to be careful in acting consistently with previously expressed responses. The theory of cognitive dissonance suggests that people are not generally willing to take positions that conflict with previous behaviour.[10] In the US sample 41.9 percent of the respondents accepted at least one of the price bids, while the corresponding share in the CS sample was 45.4 percent, and as noted above in the US sample a considerably large share (40 percent) rejected all three price premiums compared to the CS sample (23 percent). Put bluntly, this may suggest that some of the respondents who refuted all bids may have been looking out for a ‘politically correct’ excuse when confronted with the follow-up questions, and these respondents were somewhat over-represented in the US sample.

Overall, the differences in responses reported in Table 2 may both reflect true differences in moral deliberations among the respondents as well as *ex post* justifications for choices already made. For variables that are included in the binary choice analysis below and for which we suspect the latter may be the case – most likely the variables “renewable power not “green”” and “support to other areas” – one should be careful in interpreting the results since they may not entirely reflect the result of deliberations pursued prior to the choice.

Before proceeding with the econometric analysis it is useful to discuss an additional caveat with the two scenarios framing the choices in the different samples. This concerns the fact that currently power generation in Sweden is heavily dominated by hydro and nuclear power, but the Swedish parliament has decided to gradually phase out nuclear. Fossil fuels only contribute a few percent to total power generation in the country. Therefore, any increase in renewable power will primarily be at the expense of reduced nuclear power generation. This “nuclear replacement effect” is built into both scenarios, but the respondents in the CS sample explicitly choose between two power generation mixes where one of these implies more renewable power and less nuclear. Thus, any negative or positive preferences towards nuclear power are more likely to come to the surface in the CS sample compared to the US sample. Based on our results it is difficult to tell whether this effect was important in practice. The results showing that the respondents in the US sample were more prone to state that renewable power should not cost more and is unlikely to be more environmentally benign than non-renewable power, may support the existence of a “nuclear replacement effect” if the average respondent is against nuclear. If this is the case, the respondents in the CS sample may have perceived the “increased renewable” alternative as offering two benefits (more renewables and less nuclear), and for this reason they are more inclined to accept the fact that renewable power only comes with a price premium and that its environmental qualities can be questioned. Still, overall the public support for nuclear power is fairly strong in Sweden, especially among older men (a group which is over-represented in our samples). In addition,

[10] The seminal study in this field is Festinger (1957), and for more recent elaborations on cognitive dissonance theory, see, for instance, Aronson (1992) and Johansson-Stenman and Svedsäter (2003).

as is illustrated by the econometric results below, the above does not imply that the perceived "greenness" of renewable power had a stronger impact on the choice probability in the estimations based on the CS sample.

4.2 Results of the Binary Choice Analysis

The empirical analysis is based on the responses of 564 individuals (302 in the US sample and 262 in the CS sample). Each respondent was asked to do three repeated choices, each associated with three different price premiums, and consequently the analysis is based on a total of 906 and 786 observations, respectively. The econometric model estimated is the random effects binary probit model. All respondents were asked to choose between the alternatives in three repeated choices each associated with three different price premiums, and the chosen specification allows the error term to be correlated within individual choices while it is assumed to be independent between individuals. The model is thus less restrictive than the ordinary binary probit model that ignores the correlation altogether although it is still restrictive in the sense that it assumes equal correlation between the choices of each individual. However, since there were only three relatively simple choices facing each individual we expect this assumption to be plausible (Butler and Moffit, 1982; Greene, 2000). The hypothesis of no individual-specific correlation is tested empirically by evaluating the statistical significance of the coefficient, rho(ρ).

The results from the estimation of the random effects binary probit model are presented in Table 3. The overall performance of the estimated models differs between the two samples. Although we can – based on the log-likelihood measures – reject the hypothesis of all coefficients being equal to zero at the one percent significance level (the critical value equals 31) the 'explanatory power' (i.e., the chi-square measures) is considerably higher in the US sample. The statistical significance of the estimate of the average correlation of choices within individuals, rho(ρ), indicates that the random effects model is preferred over a simpler model that ignores the correlation within individuals; this is the case for both samples.

The results display that in both samples the respondents were sensitive to price increases; the coefficients representing the electricity price are highly statistically significant. Somewhat surprisingly, though, the fact that some houses are electricity-heated does not appear to affect choices. Overall, felt personal responsibility to contribute and variables related to PCE all influence the probability of choosing the "increased renewable" alternative. Still, in the CS sample we could not reject the null hypothesis that concern for the environ-mental impacts of non-renewable energy sources did not influence this choice. The coefficients representing the two reciprocity variables are highly statistically significant in the CS scenario, but less so in the US scenario.

The estimation results also show that the different socio-economic variables overall contribute little to our understanding of choices made. Highly educated people tend to be more prone to support increased renewable power, but this impact is only statistically significant in the CS sample. In the US sample, on the other hand, there is a tendency that older people are less willing to contribute to renewable power. The presence of social norms did not seem to matter for the choices made, and it is worth noting that only around 15-16 percent of the respondents agreed (partly or entirely) to the statement concerning the presence of people who expect them to make contributions to increased renewable power.

Table 3. Random Effects Binary Probit Parameter Estimates

	Unconditional scenario (US)			*Conditional scenario (CS)*		
Variables	Coefficient		*t*-statistics	Coefficient		*t*-statistics
Constant	1.747		1.161	1.255		1.561
Cost of additional renewable power						
Electricity price	-0.758	***	-8.716	-0.553	***	-9.753
Electricity heating	-0.226		-0.676	-0.191		-1.138
Self image variables						
Environmental impact	0.108	**	2.511	0.011		0.570
Personal responsibility	1.089	***	5.103	0.228	**	2.473
No impact on renewable production	-0.461	**	-2.549	-0.181	*	-1.874
Renewable power not “green”	-0.474	***	-2.867	-0.172	*	-1.946
Reciprocity variables						
Conditional cooperation	0.263	*	1.851	0.320	***	4.627
Consumer responsibility	-0.086		-0.572	0.176	**	2.421
Socio-economic variables						
Gender	0.495		1.194	-0.164		-0.788
Age	-0.025	*	-1.863	-0.005		-0.672
Education	0.633		1.530	0.676	***	3.505
Other variables						
Social norm	-0.164		-0.967	0.064		0.724
Other areas	-0.489	***	-2.921	-0.287	***	-3.155
Experience	0.273		0.788	-0.386	**	-2.071
Rho (ρ)	0.813	***	18.371	0.468	***	5.714
	Log-likelihood: -335 Restricted log-likelihood: -401 Chi-squared: 133			Log-likelihood: -367 Restricted log-likelihood: -385 Chi-squared: 36		

*, **, *** Coefficients statistically significant at the ten, five and one percent level, respectively.

Respondents in the CS sample who claim that they have prior personal experience of renewable power are less likely to express support for renewable power,[11] but this impact is statistically insignificant in the US sample. Finally, in both samples we find that those who are more inclined to express support for other environmental projects, are also less willing to pay a premium for increased reliance on renewable power.

Since our main interest in this empirical section lies in analyzing to what extent *the sizes of the impacts* it is important to note that the estimated coefficients of the binary choice model cannot be directly interpreted in any meaningful way analogous to the slope coefficients in the linear regression model. Therefore we also calculate the so-called marginal effects; they

[11] It is worth noting that when we instead included the variable “prior personal experience of non-renewable power”, this variable had no statistically significant impact in neither of the two samples.

are the partial derivatives of the probability function, evaluated at the sample mean values of the explanatory variables and can be interpreted as the marginal (percentage) change in the probability of choosing the increased renewable alternative for a unit change in the value of the actual explanatory variable (e.g., Gujarati, 2003). Table 4 presents the marginal effects for the estimates together with their corresponding *t*-values and elasticities, while Table 5 then shows confidence intervals for the differences in estimated marginal effects between the two samples (the numbers indicate the respective marginal effects in the US sample minus the corresponding effects in the CS sample).

Table 4. Estimated Marginal Effects and Corresponding Elasticities

	Unconditional scenario (US)				*Conditional scenario (CS)*			
	Marginal effect		*t*-statistic	Elasticity	Marginal effect		*t*-statistic	Elasticity
Cost of renewable power								
Electricity price	-0.123	***	-6.999	-1.010	-0.158	***	-9.197	-1.109
Electricity heating	-0.036		-0.677	-0.059	-0.055		-1.139	-0.074
Self image variables								
Environmental impact	0.018	**	2.448	0.846	0.003		0.570	0.128
Personal responsibility	0.177	***	4.716	1.541	0.065	**	2.459	0.480
No impact on renewable prod.	-0.075	***	-2.529	-0.848	-0.052	*	-1.871	-0.494
Renewable power not "green"	-0.077	***	-2.799	-0.547	-0.049	*	-1.946	-0.288
Reciprocity variables								
Conditional cooperation	0.042	*	1.886	0.374	0.092	***	4.580	0.737
Consumer responsibility	-0.014		-0.573	-0.113	0.050	**	2.407	0.350
Socio-economic variables								
Gender	0.080		1.181	0.052	-0.047		-0.787	-0.031
Age	-0.004	*	-1.879	-0.621	-0.001		-0.673	-0.180
Education	0.103		1.504	0.089	0.194	***	3.490	0.148
Other variables								
Social norm	-0.027		-0.969	-0.181	0.018		0.725	0.103
Other areas	-0.079	***	-2.833	-0.708	-0.082	***	-3.146	-0.653
Experience	0.044		0.781	0.052	-0.111	**	-2.065	-0.120

*, **, *** Estimates statistically significant at the ten, five and one percent level, respectively.

The results show that the responsiveness to price raises does not differ between the two scenarios investigated. The estimated 'price elasticity' is slightly higher in the CS sample compared to the US sample, but – as Table 5 shows – these differences are not statistically different from zero even at a 'generous' significance level (i.e., 10 percent). Thus, even though respondents in the US sample were more inclined (at least *ex post*) to stress the importance of cheap renewable power, this does not imply that they show evidence of a more price responsive choice behaviour in the experiment.

As expected we do, however, find important differences between samples in the case of the self-image variables. First, the impact of personal responsibility on the willingness to support renewable power is more pronounced in the US sample, and the statistical test shows that we can reject the null hypothesis of equal marginal effects at the five percent level. This result is thus consistent with the theoretical prediction that the unconditional scenario triggers a moral deliberation about the personal responsibility to contribute to the public good. Second, in the theoretical section we also suggested that self image would be affected by the external environmental impacts incurred by the choice to support renewable. This implies that if the respondent believes that: (a) it is important to avoid the environmental effects of replacing non-renewable power; (b) his/her choice will in fact imply increased renewable power; and (c) this increase implies environmental impacts that are more benign than those incurred by non-renewable power; he/she will be more willing to support renewable power. All these impacts appear to be more pronounced in the US sample, but it is only in the first case – the environmental impact variable – that there is a statistically significant difference (at the ten percent level) between the marginal effects in the two samples. In the CS case we cannot even reject the null hypothesis of a zero marginal effect for this variable.

Table 5. Confidence Intervals for Sample Differences (US-CS) in Marginal Effects

	95 percent confidence interval	*90 percent confidence interval*
Cost of renewable power		
Electricity price	-0.013 – 0.084	-0.005 – 0.076
Electricity heating	-0.124 – 0.160	-0.101 – 0.137
Self image variables		
Environmental impact	-0.003 – 0.032	0.0001 – 0.029*
Personal responsibility	0.021 – 0.202**	0.036 – 0.187*
No impact on renewable prod.	-0.102 – 0.057	-0.089 – 0.044
Renewable power not “green”	-0.101 – 0.045	-0.089 –0.033
Reciprocity variables		
Conditional cooperation	-0.108 – 0.010	-0.099 – -0.0001*
Consumer responsibility	-0.127 – -0.001**	-0.117 – -0.012*
Socio-economic variables		
Gender	-0.050 – 0.305	-0.021 – 0.276
Age	-0.008 – 0.002	-0.007 – 0.001
Education	-0.264 – 0.082	-0.236 – 0.053
Other variables		
Social norm	-0.117 – 0.028	-0.105 – 0.016
Other areas	-0.073 – 0.078	-0.060 – 0.065
Experience	0.002 – 0.308**	0.027 – 0.283*

*, ** Differences in marginal effects between US and CS samples statistically significant at the ten and five percent level, respectively.

Finally, in order to test for the presence of ‘personal preferences’ for the public good in question, we divided the environmental impact variable into two components: one concerned external impacts (index based on the first four statements in Table A1) and one concerned only the perceived personal environmental threat connected with the production of non-renewable power (fifth statement in Table A1, Appendix A). However, this alternative estimation added few new insights. The ‘personal’ environmental impact variable was highly statistically insignificant in both samples, while the coefficient corresponding to the ‘external’ variable showed the same pattern as in the original estimations shown in Table 4. Hence, overall these results are not consistent with the reciprocity model’s prediction that people are more likely to acknowledge moral constraints on behaviour when contributing to public goods that they have strong personal preferences for. Instead in both sample estimations the results support the presence of a ‘warm-glow’ type of preferences, for which external environmental effects matter more.

The estimates of the marginal effects are also fairly consistent with respect to the impact of the two reciprocity variables. Both effects are – as hypothesized – significantly stronger (at the ten and five per cent significance level, respectively) in the CS sample. This scenario tends to stimulate deliberations concerning others' responsibility to contribute, and when it is made clear to the respondents that the price premium is uniform across all consumers he/she does not want to free ride but acknowledges his/her personal responsibility. In the US sample, however, evidence that people who are concerned about reciprocity also contribute more to the public good is much scarcer.

For the socio-economic variables we found no significant differences between the two samples. This is also the case for the remaining variables with the exception of that relating to the respondents' prior personal experience of renewable power. This impact is – as noted above – negative, but also much more pronounced in the CS sample although it is difficult to assert why this difference is present.

5. Concluding Remarks

In this chapter we have investigated the main driving forces behind individuals' willingness to accept a price premium to support increased reliance on renewable power generation. The results indicate that different types of factors tend to dominate choices depending on the support scheme considered, and this may suggest some important implications for measures undertaken (either by policy makers or electricity companies) to increase the public's valuation of renewable power as well as the legitimacy of measures to increase renewable power production. In the ”green” market case it is particularly important to increase the ”perceived consumer effectiveness”; consumers need to feel confident that the purchases they make imply significant positive (i.e., environmentally appealing) outcomes. Today the public confidence is low in this respect, and this probably provides an important explanation to why households have been inactive in the “green” electricity market. Moreover, measures to increase household demand for “green” electricity must also acknowledge the fact that individual's perception of the importance of environmental problems as well as of the personal moral responsibility to take action seem to have a significant impact on their willingness to purchase “green” power. Overall the impacts of the above factors are – although far from unimportant – less pronounced in the case of public support for a more ambitious certificate scheme. In this latter case it becomes less important to appeal to personal obligations as well as to stress the environmental importance of renewable power. Instead references to the certificate scheme as a ‘collective' undertaking in which all consumers do their share, can be more effective in spurring the public's support for the system. Finally, however, we need also to recognize that regardless of the support scheme considered, individuals appear equally price sensitive, and at the end of the day the must significant push to renewable power will probably emerge if/when these energy sources become commercial and thus are in no need of additional economic support.

In the recent past the renewable certificate system has largely taken over the role of the “green” electricity market as the main vehicle through which individuals contribute to the public goods associated with renewable power sources. While previous studies suggest that such a ‘policy takeover' can crowd-out moral-based motivations to contribute, one should note that in the case of “green” electricity in Sweden there is not much household demand to crowd-out and a shift to a mandatory system may be necessary to reach the desired policy goals. Nevertheless, the experiences from the Swedish green electricity scheme – administerred by the SSNC – suggest that when the certificate scheme was introduced in 2003 the annuals sales of (voluntary) green electricity feel from almost 14 percent to less than 5 percent of electricity consumption (Ekengren, 2005). Our results suggest that one of the policy costs of relying more heavily on mandatory certificates schemes (rather than strengthening the voluntary market) is that individuals are less likely to deliberate on the environmental benefits of renewable power as well as to seriously consider ones own personal responsibility to contribute to these benefits. In a democratic society such deliberations are important, not the least in environ-mental policy. Environmental issues often have a broad ethical content, and since ethics are a matter for argument, public discourse over what is worth promoting and why becomes important. If people routinely contribute to renewable power by paying the certificate fee without thinking much about the impacts of their efforts, the long-run legitimacy of the system may quickly deteriorate in the case of, say, substantial electricity price increases. This suggests that it may be appropriate to promote voluntary and mandatory approaches towards renewable energy support in parallel.

REFERENCES

Andreoni, J. (1990). Impure altruism and donations to public goods: A theory of warm glow giving. *The Economic Journal, 100,* 464-477.

Butler, J., and Moffit, R. (1982). A computationally efficient quadrature procedure for the one factor multinomial probit model. *Econometrica*, *50*, 761-764.

Aronson, E. (1992). The return of the repressed: Dissonance theory makes a comeback. *Psychological Inquiry, 3*, 303-311.

Bergstrom, T., Blume, L., and Clotfelter, C. T. (1986). On the private provision of public goods. *Journal of Public Economics, 29,* 25-49.

Biel, A., and Thogersen, J. Activation of social norms in social dilemmas. A review of the evidence and reflections on the implications for environmental behaviour. *Journal of Economic Psychology*, *28,* 93-112.

Brekke, K. A., Kverndokk, S., and Nyborg, K. (2003). An economic model of moral motivation. *Journal of Public Economics, 87*, 1967-1983.

Directive 2001/77/EC of the European Parliament and the Council of 27 September 2001 on the promotion of electricity produced from renewable energy sources in the internal electricity market. Official Journal L283, 27/10/2001.

Ek, K. (2005). Public and private attituder towards "green" electricity: the case of Swedish wind power. *Energy Policy*, *33,* 1677-1689.

Ek, K., and Söderholm, P. (2005). Green electricity consumption in Swedish households: The role of norm-motivated consumer behavior. Unpublished manuscript.

Ekengren, T. (2005). Swedish Society for Nature Conservation (SSNC), Personal Communication, May 2.

Ellen, P. S., Wiener, J. L., and Cobb-Walgren, C.(1991) The role of perceived consumer effectiveness in motivating environmentally conscious behaviors. *Journal of Public Policy and Marketing*, *10,* 102-117.

Fehr, E., and Falk, A. (2002). Psychological foundations of incentives. *European Economic Review, 46,* 687-724

Festinger, L. (1957). *A Theory of Cognitive Dissonance*, Evanston, IL: Row, Peterson.

Fischbacher, U., Gächter, S., and Fehr, E. (2001). Are people conditionally cooperative? Evidence from a public goods experiment. *Economics Letters, 71,* 397-404.

Frey, B. S., and Jegen, R. (2001). Motivation crowding theory. *Journal of Economic Surveys, 15*, 589-611.

Greene, W. H. (2000). *Econometric Analysis* (4th edition). New Jersey: Prentice Hall.

Gujarati, D. N. (2003). *Basic Econometrics* (4th edition). New York: McGraw-Hill.

Haab, T. C., and McConnell, K. E. (2002). *Valuing Environmental and Natural Resources: The Econometrics of Non-market Valuation.* Cheltenham and Northampton: Edward Elgar.

Hanemann, W. M. (1984). Welfare Evaluations in Contingent Valuation Experiments with Discrete Responses. *American Journal of Agricultural Economics*, *66,* 332-341.

Holländer, H. (1990). A social exchange approach to voluntary cooperation. *American Economic Review, 80,* 1157-1167.

Johansson-Stenman, O., and Svedsäter, H. (2003). Self image and choice experiments: Hypothetical and actual willingness to pay. Working Papers in Economics no 94, Göteborg University.

Kåberger, T., Sterner, T., Zamanian, M., and Jurgensen, A. (2005). Economic efficiency of compulsory green electricity quotas in Sweden. *Energy & Environment, 15*, 675-697.

Margolis, H. (1982). *Selfishness, altruism, and rationality*. Chicago: University of Chicago Press.

Moscovici, S. (1985). Social Influence and Conformity. In Gardner, L. and Aronson, E. (Eds) *The Handbook of Social Psychology.* New York: Random House. 347-412.

Nyborg, K. (2000). Homo economicus and homo politicus: Interpretation and aggregation of environmental values. *Journal of Economic Behavior & Organization, 42*, 305-322.

Nyborg, K. and Rege, M. (2003). Does public policy crowd out private contributions to public goods? *Public Choice, 115,* 397-418.

Nyborg, K., Howarth, R. B., and Brekke, K. A. (2006). Green consumers and public policy: On socially contingent moral motivation. *Resource and Energy Economics*, *28*, 351-366.

Rabin, M. (1993). Incorporating fairness into game theory and economics. *American Economic Review, 83,* 1281-1302.

Schwartz, S. H. (1970). Moral decision making and behaviour. In Macaulay, J., and Berkowitz, L. (Eds) *Altruism and helping behavior. Social Psychological of Some Antecedents and Consequences.* New York and London: Academic Press, 127-141.

Sehti, R., and Somanathan, E. (2003). Understanding reciprocity. *Journal of Economic Behavior & Organization, 50*, 1-27.

Sen, A. K. (1977). Rational fools: A critique of the Behavioural Assumptions of Economic Theory. *Philosophy and Public Affairs*, *6,* 317-344.

Statistics Sweden (2005). Internet: www.scb.se, September 2005.

Sugden, R. (1984). Reciprocity: The supply of public goods through voluntary contributions. *The Economic Journal*, *94*, 772-787.

Sveriges Elleverantörer (Swedenergy) (1999). *Vad tycker elkunderna om grön el*. Stockholm.

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Chapter 5

A FRAMEWORK FOR THE ECONOMIC VALUATION OF LAND USE CHANGE

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ABSTRACT

There is a broad recognition that sustainable land management (SLM) is crucial for ensuring an adequate, long-term supply of food, raw materials and other services provided by the natural environment to the human society. However, to date, SLM practices are the exception rather than the rule in many parts of the world. Among the causes for unsustainable land management is a general lack of understanding of the economic costs of land degradation and the benefits of sustainable land management. This paper presents a methodological framework for analyzing the benefits of sustainable land management. The framework comprises three complementary types of assessment: partial valuation, total valuation and impact analysis. The first two allow for static assessment of selected respectively all economic benefits from a certain land use. The third approach is dynamic, and allows for analyzing the costs and benefits related to changes in land use. Each approach requires the application of a number of sequential methodological steps, including (i) ecosystem function and services identification; (ii) bio-physical assessment of ecosystem services; (iii) economic valuation; and (iv) ecological-economic modeling. The framework is demonstrated by means of a simple case study in the Guadalentin catchment, SE Spain.

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1. INTRODUCTION

There is a broad recognition that sustainable land management (SLM) is crucial for ensuring an adequate, long-term supply of food, raw materials and other services provided by the natural environment to the human society. SLM involves both the long-term maintenance of the productive capacity of agricultural lands, and the sustainable use of natural and semi-natural ecosystems, such as semi-arid rangelands or forests. Nevertheless, SLM practices are the exception rather than the rule in many parts of the world. A whole range of social, institutional and economic factors play a role with regards to the lack of sustainability in the management of natural resources. For instance, farmers and local ecosystem users may be driven by immediate food and income requirements and may have limited possibilities to adjust harvest levels to the carrying capacity of the ecosystem.

One of the factors that is often identified as being critically important is that the various economic benefits that are provided by multifunctional agricultural landscapes and natural ecosystems tend to be underestimated in decision making. Agricultural and natural ecosystems provide a whole range of valuable goods and services, ranging from the supply of food or medicinal plants, to the regulation of water flows and biochemical cycles, to the provision of sites for recreation or cultural events. Many of these services directly or indirectly contribute to human welfare and, as such, have economic value.

The general lack of recognition of these values in decision making is caused by a range of factors. First, these benefits are often difficult to specify, as they are widely varying in terms of the type of benefit supplied, and as they operate over a range of spatial and temporal scales. Second, several of these benefits have a public goods character and/or are not traded in a market. In spite of their welfare implications, they therefore do not show up in economic statistics. Third, there is often a mismatch between the stakeholders that pay the (opportunity) costs of maintaining an environmental benefit (e.g. by not converting a forest to cropland) and the beneficiaries of that benefit (e.g. downstream water users benefiting from the regulation of water flows).

Through assessment of the economic value of the multiple benefits provided by land and ecosystems, it is possible to increase the awareness of stakeholders and decision makers of the economic benefits resulting from sustainable land management. Since economic considerations generally play a key role in decision making, it is anticipated that economic valuation of environmental benefits can contribute to a more sustainable and a more efficient decision making. Analysis and valuation of ecosystem services can also guide the setting up of mechanisms to compensate the suppliers of ecosystem services for the costs related to providing those benefits in a Payment for Ecosystem Services (PES) mechanism.

To date, a wide range of assessments of the costs of land degradation and the benefits of sustainable land management have been carried out (e.g. Lal and Stewart, 1990; Pimentel et al., 1995). In addition, a number of conceptual frameworks have been proposed for analyzing the economic aspects of environmental change. The Millennium Ecosystem Assessment (2003) has provided a general framework for ecosystem services assessment, and, for instance, Barbier et al., 1997 presents a framework for the economic valuation of wetlands. However, to date, to the best knowledge of the authors, a comprehensive and systematic framework for analyzing the economic consequences of changes in land management is still lacking.

This study presents a conceptual framework for analyzing the costs of land degradation and the benefits of SLM. The framework supports three main approaches (partial valuation, total valuation and impact assessment), each of them is illustrated with a case study. Note that these case studies are based on limited data and simplified methodologies, and their only purpose is to illustrate the application of the framework.

2. A FRAMEWORK FOR ANALYZING THE ECONOMIC IMPACTS OF LAND USE CHANGE

A general framework for analyzing the economic impacts of land use change is presented in Figure 1. The framework can be used for three distinct approaches to analyzing the economic benefits of SLM: (i) Partial Valuation; (ii) Total Valuation; and (iii) Impact Assessment.

(i) Partial valuation. Partial valuation involves the economic valuation of only one or a limited set of environmental benefits. It can be used where only few environmental benefits supply the large majority of benefits to society, and where appraisal of only few benefits is required to support decision making. This approach can be applied, for instance, in case the impact of SLM on food security needs to be assessed.

(ii) Total valuation. The second approach is ‘Total valuation’. This approach is appropriate where a full accounting of the benefits provided by an area under a certain management system is required. In this case, all services need to be identified and valued. For instance, in case a decision needs to be taken involving the selection of one of two land use conversion options, it may be important to analyze all benefits provided in the two options. Note that, in specific cases, it may be clear that some services only generate a very minor part of the total benefits, as in the case of carbon sequestration in a system that absorbs only minimal amounts of carbon over time. In this case, it may be decided to skip these minor services and include them only as a pro memory post.

(iii) Impact assessment. The third approach is ‘Impact assessment’. It involves analyzing the impacts of changes in environment and land management on the supply of benefits to society. This approach needs to be applied in case of a change in the management of an area (e.g. through the adoption of various SLM practices). In this case, it is necessary to analyze both the economic value of the benefits generated by the system under consideration, *and* how the supply of these benefits will change following a change in management practices. This approach is also relevant for the prediction of the impact of environmental pressures, e.g. pollution, that may cause a change in the state of the environmental system. Hence, compared to the two previous approaches, this approach requires an additional step, dealing with how the impact of the change in management or pressures can be analyzed or modeled.

The framework comprises five complementary steps: (i) Problem definition and selection of the assessment approach; (ii) Identification of ecosystem functions and services; (iii) Biophysical assessment of ecosystem services; (iv) Economic valuation of ecosystem services; and (v) Ecological-economic modeling. These steps are described below.

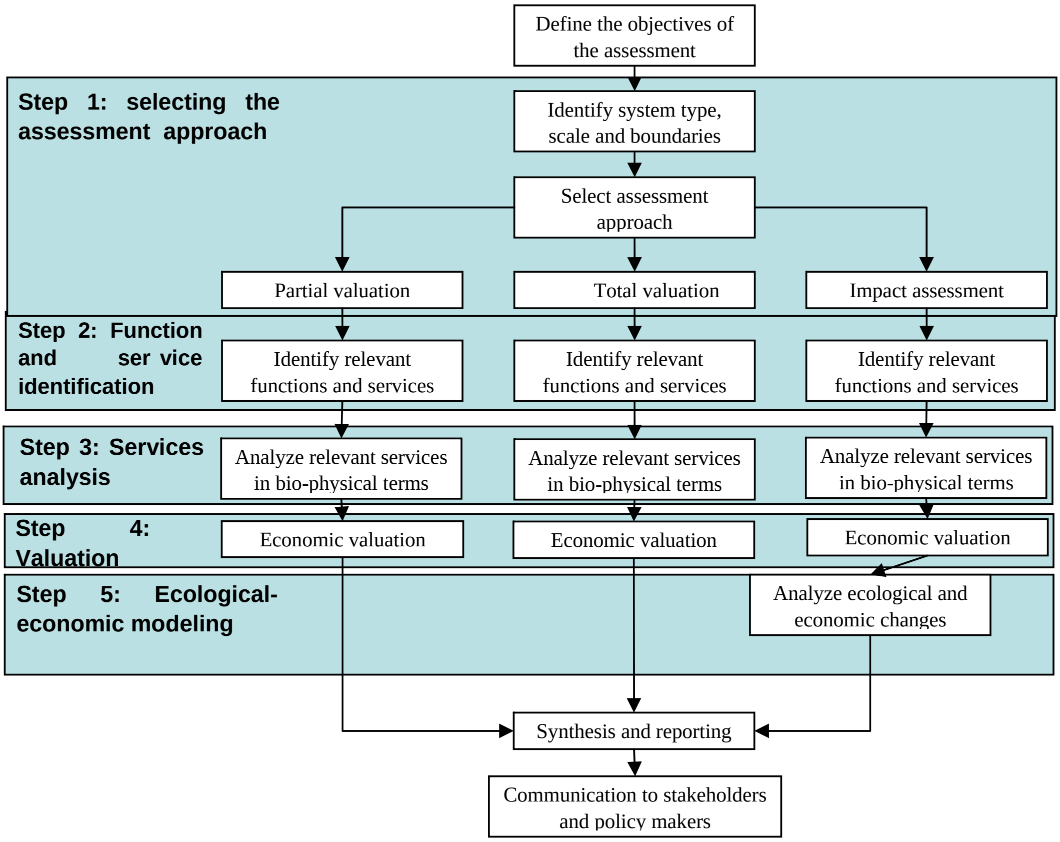

Figure 1. Framework for analyzing the costs of and degradation and the benefits of sustainable land management.

2.1 Problem Definition and Selection of the Assessment Approach

Economic valuation of land use changes (as any other analysis) requires that the object of the valuation is clearly defined. Hence, it is necessary to define the system to be analyzed, in terms of its spatial and temporal boundaries. The ecosystem is the entry point often used for valuation of ecosystem services and environmental benefits. Following Likens (1992), ecosystems are defined as: 'the individuals, species and populations in a *spatially defined area*, the interactions among them, and those between the organisms and the abiotic environment'. This spatial approach makes it easier to define the physical boundaries of the area to be analyzed. Following the Millennium Ecosystem Assessment, ecosystems may comprise both natural and/or strongly man influenced systems such as agricultural fields.

As ecological and institutional boundaries seldom coincide, stakeholders in ecosystem services often cut across a range of institutional zones and scales. Whereas for the analysis of land degradation processes, ecosystem services and ecosystem dynamics the ecosystem is the appropriate unit of analysis, in the identification of policy measures the administrative and institutional contexts need to be explicitly considered. This incongruence between ecological and political boundaries is very common in environmental management, and flexible solutions need to be identified on a case-by-case basis.

Following the definition of the system boundaries, the appropriate valuation approach needs to be selected. As explained in the Introduction section, the user may be interested in (i) Partial valuation; (ii) Total valuation; or (iii) Impact assessment. Table 1 provides some examples of analyses that can be carried out with these three approaches.

Table 1. Examples of the Potential Applications of the Three Main Valuation Approaches

Valuation approach	Examples
Partial valuation	Valuation of the production of non-timber forest products by a forest.
	Valuation of the hydrological service of an upland forest in order to define a payment vehicle from downstream users to upland managers to maintain this service.
Total valuation	Valuation of the ecosystem services supplied by a forest in order to compare the benefits of timber logging with those of sustainable management
	Valuation of the services provided by a natural area in order to identify which stakeholders benefits from the area and which stakeholders may be expected to contribute to financing the preservation of the area.
Impact assessment	Analyzing the impacts of pollution control measures in a wetland on water quality and ecosystem services supply in order to compare the costs and benefits of pollution control measures
	Analysis of the impact of disturbances (e.g. road construction, or desertification) on the supply of ecosystem services

2.2 Ecosystem Function and Services Identification

In the early 1970s, the concept of ecosystem function was proposed to facilitate the analysis of the benefits that ecosystems provide to society. An ecosystem function can be defined as “the capacity of the ecosystem to provide goods and services that satisfy human needs, directly or indirectly”. Ecosystem functions depend upon the state and the functioning of the ecosystem. For instance, the function ‘production of firewood’ is based on a range of ecological processes involving the growth of plants and trees that use solar energy to convert water, plant nutrients and CO2 to biomass.

A function may result in the supply of ecosystem services, depending on the demand for the good or service involved. Ecosystem services are the goods or services provided by the ecosystem to society (following the definition of the Millennium Ecosystem Assessment, 2003). The supply of ecosystem services will often be variable over time, and both actual and potential future supplies of services should be included in the assessment.

Ecosystem functions, and the services attached to these functions, vary widely as a function of the type of ecosystem and the socio-economic setting involved. For example, the capacity of the ecosystem to provide firewood depends on the forest cover and the amount of woody plant biomass contained in the system, as well as, in the longer term, on the primary productivity of the forest. However, the actual supply of firewood also depends on the *demand* of different stakeholders for firewood. This demand is determined by the need for wood energy as well as the availability of other sources to satisfy household energy needs.

Table 2 provides a comprehensive list of ecosystem services, containing 24 different types of services. By and large, the list follows the MEA (2003). Compared to MEA (2003) some minor adjustments have been made in order to ensure consistency in its application to SLM. The list contains three types of ecosystem services, which are based on a different type of interaction between people and ecosystems: Provisioning services, Regulation services and Cultural services. Contrary to Millennium Ecosystem Assessment (2003), but analogous to Costanza et al. (1997) and Hein et al. (2006), there is no category 'supporting services'. Supporting services represent the ecological processes that underlie the functioning of the ecosystem. Their inclusion in valuation may lead to double counting as their value is reflected in the other three types of services. In addition, there are a very large number of ecological processes that underlie the functioning of ecosystems, and it is unclear on which basis supporting services should be included in, or excluded from a valuation study.

Table 2. List of Ecosystem Services (based on Ehrlich and Ehrlich, 1981; Costanza et al., 1997; DeGroot et al., 2002; Millennium Ecosystem Assessment, 2003; Hein et al., 2006)

Category	Definition	Ecosystem services
Provisioning services	Provisioning services reflect goods and services extracted from the ecosystem	- Food - Fodder (including grass from pastures) - Fuel (including wood and dung) - Timber, fibers and other raw materials - Biochemical and medicinal resources - Genetic resources - Ornamentals
Regulation services	Regulation services result from the capacity of ecosystems to regulate climate, hydrological and bio-chemical cycles, earth surface processes, and a variety of biological processes.	- Carbon sequestration - Climate regulation through control of albedo, temperature and rainfall patterns - Hydrological service: regulation of the timing and volume of river flows - Protection against floods by coastal or riparian systems - Control of erosion and sedimentation - Nursery service: regulation of species reproduction - Breakdown of excess nutrients and pollution - Pollination - Regulation of pests and pathogens - Protection against storms - Protection against noise and dust - Biological nitrogen fixation (BNF)
Cultural services	Cultural services relate to the benefits people obtain from ecosystems through recreation, cognitive development, relaxation, and spiritual reflection.	Provision of cultural, historical and religious heritage (e.g. a historical landscape or a sacred forests) - Scientific and educational information - Opportunities for recreation and tourism Amenity service: provision of attractive housing and living conditions Habitat service: provision of a habitat for wild plant and animal species

Note that, in principle, the user has the choice of valuing services or functions; both express the benefits supplied by the natural environment to society. The main difference is that valuation of services is based on valuation of the flow of benefits, and valuation of functions is based on the environment's capacity to supply benefits. The first expresses clearly the current benefits received, but additional analyses are required if the flow of ecosystem services is likely to change in the short or medium term (e.g. if current extraction rates are above the regenerative capacity of the ecosystem). In this case, calculation of the NPV requires that assumptions are made on the future flows of services.

Functions better indicate the value that can be extracted in the long-term, and their value is not biased by temporary overexploitation. However, it is often much more difficult to assess the capacity to supply a service than to assess the supply of the service itself. For instance, for the function 'supply of fish', this requires analysis of the sustainable harvest levels of the fish stocks involved which needs to be based on a population model including reproduction, feed availability and predation levels. Hence, in most valuation studies, it is chosen to value services rather than functions, and to account for potential changes in services supply in the assessment.

An important issue in the valuation of ecosystem services is the *double counting of services* (Millennium Ecosystem Assessment, 2003; Turner et al., 2003). Specifically, there is a risk of double counting in relation to the regulation services that support the supply of other services from an ecosystem. In general, regulation services should only be included in the valuation if (i) they have an impact outside the ecosystem to be valued; and/or (ii) if they provide a *direct* benefit to people living in the area (*i.e.* not through sustaining or improving another service) (see Hein et al., 2006, for more information on double counting). A prerequisite for applying this approach to the valuation of regulation services is that the ecosystem is defined in terms of it's spatial boundaries – otherwise the external impacts of the regulation services can not be precisely defined.

2.3 Ecosystem Services Assessment

The next step in the economic assessment is the quantification, in bio-physical units, of the relevant ecosystem services identified in the previous step. For *production* services, this involves the quantification of the flows of goods harvested in the ecosystem, in a physical unit. For most *regulation* services, quantification requires spatially explicit analysis of the bio-physical impact of the service on the environment in or surrounding the ecosystem. For example, valuation of the hydrological service of a forest first requires an assessment of the precise impact of the forest on the water flow downstream, including such aspects as the reduction of peak flows, and the increase in dry season water supply (Bosch and Hewitt, 1982). The reduction of peak flows and flood risks is only relevant in a specific zone around the river bed, which needs to be (spatially) defined before the service can be valued. An example of a regulation service that does usually not require spatially explicit assessment prior to valuation is the carbon sequestration service – the value of the carbon storage does not depend upon where it is sequestered. *Cultural* services depend upon a human interpretation of the ecosystem, or of specific characteristics of the ecosystem. The benefits people obtain from cultural services depend upon experiences during actual visits to the area, indirect experiences derived from an ecosystem (e.g. through nature movies), and more abstract cultural and moral considerations (see e.g. Aldred, 1994; Posey, 1999). Assessment of cultural services requires assessment of the numbers of people benefiting from the service, and the type of interaction they have with the ecosystem involved. Table 3 presents potential indicators for the biophysical assessment of ecosystem services.

Table 3. Potential Indicators for the Biophysical Assessment of Ecosystem Services

Category	Key goods and services provided	Potential indicators
Provisioning services	- Food - Fuel (including wood and dung) - Timber, fibers and other raw materials - Biochemical and medicinal resources - Genetic resources	- For all provisioning services: amount of product harvested per year; Inputs required for harvesting (time, equipment, etc.); Total inputs and outputs in case the good is used as input in a production process
Regulation services	- Carbon sequestration - Climate regulation through control of temperature and rainfall patterns - Hydrological service - Protection against floods by coastal or riparian systems - Control of erosion and sedimentation - Nursery service: regulation of species reproduction - Breakdown of excess nutrients and pollution - Pollination - Regulation of pests and pathogens - Protection against noise and dust - Biological nitrogen fixation (BNF)	- Amount of C sequestered - Differences in average and extreme rainfall and temperature. - Impact of vegetation on water flow, - Height and effectiveness of coastal belt - Amount of sediments generated or erosion avoided - Numbers of juveniles produced per area unit. - Difference in pollutant concentrations between water flowing in, and water flowing out of the system - Crop yields in areas with adequate pollination compared areas with pollination deficit. - Crop yields in areas with and without such control, or health impacts in areas with an without such control. - Noise levels, particulate matter levels, and concentrations of specific pollutants on either side of the vegetation belt - Amounts of N fixed.
Cultural services	Provision of cultural, historical and religious heritage - Scientific and educational information - Recreation and tourism - Amenity service: provision of attractive housing and living conditions - Habitat service: provision of a habitat for wild plant and animal species	For all services: amount of people benefiting from the service; type of benefits people obtain. For the habitat service: number of species; number of red list species; hectares of ecosystem, ecosystem quality versus ecosystem in natural state, biodiversity indices.

2.4. Economic Valuation

Following welfare economics, the economic value of a resource can be determined via individual preferences as expressed by willingness to pay (WTP) or willingness to accept (WTA) for a change in the supply of that resource (Hanemann, 1991). Aggregation of individual welfare impacts is required to obtain the welfare impact on society. Where relevant, this aggregation needs to consider equity issues, for instance where the interests of one stakeholder group (e.g. traditional ecosystem users), are considered to be more important than those of other stakeholder groups.

The appropriate measure of economic value is determined by the specific context of the resources being managed. Care needs to be taken that the valuation method gives a proper indication of the value of the service involved, reflecting a true WTP or WTA, and avoiding the double counting of services or values. It is also important that the user is aware of the concepts of marginal and total value, where marginal value reflects the value of an incremental change in the supply of a resource, and total value the overall value of a resource.

There are several types of economic value. Following the Millennium Ecosystem Assessment (2003), the framework distinguishes the following four types: (i) direct use value; (ii) indirect use value; (iii) option value; and (iv) non-use value. They are elaborated in Table 4. The aggregated economic value of an area, combining these four value types, is often referred to as Total Economic Value (TEV). Note that other authors have proposed different classifications of economic value types, see for example Hanley and Spash (1993) and Kolstad (2000).

Table 4. Types of Economic Value.

Value Type	Description
Direct use value	This value arises from the direct utilization of ecosystems, for example through the sale or consumption of a piece of fruit. All production services, and some cultural services (such as recreation) have direct use value.
Indirect use value	This value stems from the indirect contribution of ecosystems to human welfare. Indirect use value reflects, in particular, the type of benefits that regulation services provide to society.
Option value	Because people are unsure about their future demand for a service, they are normally willing to pay to keep the option of using a resource in the future – insofar as they are, to some extent, risk averse. Option values may be attributed to all services supplied by an ecosystem.
None-use value	Non-use value is derived from knowing that an ecosystem or species is preserved without having the intention of using it in any way. Kolstad (2000) distinguishes three types of non-use value: existence value (based on utility derived from knowing that something exists), altruistic value (based on utility derived from knowing that somebody else benefits) and bequest value (based on utility gained from future improvements in the well-being of one's descendants).

These different values may or may not be reflected in a market value. In most cases, a significant part of the Direct Use Value will be reflected in market transactions, but most of the other value types will not. This is because, for instance, the services have a public goods character, or because a market has not (yet) been established for the service. However, it is clear that, because of the economic benefits they provide, the non-market economic values need to be included in economic cost-benefit analysis.

Two types of approaches have been developed to obtain information about the value of non-market ecosystem services: expressed and revealed preference methods (Pearce and Pearce, 2001). These methods have also been called direct and indirect valuation methods, respectively. With *expressed valuation methods*, either market prices or various types of questionnaires are used to reveal the willingness-to-pay of consumers for a certain ecosystem service. The most important direct approaches are the Contingent Valuation Method (CVM) and related methods. The *revealed preference methods* use a link with a marketed good or service to indicate the willingness-to-pay for the service. They use either physical or behavioral linkages to a marketed good. With physical linkages, estimates of the values of ecosystem services are obtained by determining a physical relationship between the service and something that can be measured in the market place. For instance, with the damage-function (or dose-response) approach, the damages resulting from the reduced availability of an ecosystem service are used as an indication of the value of the service (Johanson, 1999). In the case of behavioral linkages, the value of an ecosystem service is derived from linking the service to human behavior – in particular expenditures to offset the lack of a service, or to obtain a service. Table 5 presents an overview of the various valuation approaches, detailed descriptions of the various valuation methods can be found in Pearce and Turner (1990, Hanley and Spash, (1993), Munasinghe (1993) and Cummings and Harrison (1995).

Table 5. Valuation Methods and their Applicability to Different Value Types

Valuation method	**Suitable for**	**Value category**			
		direct use value	**indirect use value**	**option value**	**non-use value**
Indirect methods:					
1) averting behavior method	Applicable to services that relate to the purification services of some ecosystems.	x	x		
2) travel cost method	Can be used to value the recreation service.	x			
3) production factor approach	Applicable where ecosystem services are an input into a production process	x		x	
4) hedonic pricing	Applicable where environmental amenities are reflected in the prices of specific goods, in particular property.	x		x	
Direct methods:					
5) CVM	The use of CVM is limited to goods and services that are easily to comprehend for respondents – excluding most regulation services	x		x	x
6) market valuation	Ecosystem goods and services traded on the market	x	x	x	

If the value of ecosystem services is expressed as NPV (instead of as an annual flow), the discount rate is a crucial factor. Discounting is used to compare present and future flows of costs and benefits derived from the ecosystem. The discount rate to be used in environmental cost-benefit analysis is still subject to debate (e.g. Howarth and Norgaard, 1993; Norgaard, 1996; Hanley, 1999). For instance, Freeman (1993) indicates that the discount rate, based upon the after- tax, real interest rate, should be in the order of 2 to 3% provided that the streams of benefits and costs accrue to the same generation, whereas Nordhaus (1994) argued that a 6% discount rate is most consistent with historical savings data. In general, the use of a high discount rate will favor ecosystem management options that lead to relatively fast depletion of resources, whereas a low discount rate will stress the economic benefits of more sustainable management options (Pearce and Turner, 1990; Tietenberg, 2000).

2.5 Ecological-Economic Modeling

The fifth step involves a dynamic assessment of the impact of land use changes on the supply of ecosystem services, and the resulting economic impacts. It comprises a quantitative analysis of the relation between drivers, ecosystem state and ecosystem services supply, as elaborated below.

(i) Modeling of drivers and management options. This first steps involves the modeling, in physical terms, of the impacts of a driver or management options on the ecosystem. This requires the modeling of the main ecosystem components, the feedback mechanisms between them, and their relation to drivers and management options. Following a systems modeling approach, ecosystem components can be interpreted as sets of connected state (level) variables, and the drivers and interactions as flow (rate) variables interacting with the components. The model should capture the relevant inputs, throughputs and outputs over time. This may comprise a range of theoretical, statistical or methodological constructs, dependent upon the requirements and limitations of the model.

For systems subject to complex dynamics, it is important that these dynamics are reflected in the model. In spite of the large number of ecological processes regulating the functioning of ecosystems, recent insights suggest that the main ecological structures are often primarily regulated by a small set of processes (Harris, 1999; Holling et al., 2002). This indicates that inclusion of a relatively small set of key components and processes in the model may be sufficient to accurately represent the (complex) dynamics of the system.

(ii) Linking ecosystem state to the supply of ecosystem services. In the second step, changes in ecosystem state have to be connected to changes in the supply of ecosystem services. This relation is strongly dependent on the type of ecosystem service. For provisioning services, relevant state indicators include the stock of the harvested produce (e.g. forest standing biomass). Changes in state are directly reflected in the capacity to provide the product. However, in many cases, the dynamics of the system can not be effectively captured with only one state variable, as in the case where forest growth is a function of both standing biomass and soil quality (see for instance Hein and Van Ierland, 2006 for an example). For regulation service, there will often be a range of state variables required to analyze changes in the supply of the service. For instance, the effectiveness of the pollination service depends on the amount of healthy pollinators available. Often, only one or few species of pollinators pollinate the large majority of the crop, but these pollinators depend on the availability of suitable habitat, and the absence of disturbances such as high concentrations of pesticides. For cultural services (the habitat service excluded), it is often not the ecological complexity that maintains the service, but rather specific aspects of it. For instance, the majority of Dutch recreationists values birdlife in a recreational site, but is rather indifferent regarding which species or how much variety of birds is encountered on the site. In view of the large variety among ecosystem services, suitable indicators and relevant processes need to be identified on a case-by-case basis.

(iii) Analysis of the impacts of ecosystem change and of the management options. Once the ecological-economic model has been constructed, it can be used to assess the impacts of the ecosystem change on the supply of ecosystem services, as well as of the efficiency and sustainability of different ecosystem management options. The efficiency of ecosystem management can be revealed through comparison of the net welfare generated by the ecosystem and the costs involved in maintaining and managing the ecosystem (e.g. Pearce and Turner, 1990). Through a simulation or an algebraic optimization approach, efficient management options, *i.e.* management options that provide maximum utility given a certain utility function, can be identified. See for instance Chang (1992) for a theoretical construct of optimization procedures, Hein (2006) for an example of a simulation approach to optimization, and Hein and Weikard (2006) for an example of algebraic optimization. The sustainability of management options can be examined by analyzing their long-term consequences for the state of the ecosystem including its capacity to supply ecosystem services (Pearce et al., 1989; Barbier and Markandya, 1990).

3. Case Study: Costs of Erosion in the Guadalentin, SE Spain

Table 6. Land Use in the Study Area (2003)

Category	Case study area (ha)
Irrigated cropland, of which	1100
- irrigated horticulture	*350*
- irrigated tree crops	*750*
Dryland cropland; of which	10750
- almonds	*4250*
- olives	*900*
- barley	*4500*
- wheat	*1100*
Shrublands	2840
Forests	450
Pine afforestations	450
Villages	50
Total	15640

Source: Comarcal, 2003

The framework is applied to the Puentes Catchment in the Guadalentin Basin in South-eastern Spain. The Guadalentin Basin extends over some 3300 km^2, covering the southern half of the province of Murcia and the eastern tip of the province of Andalucia. It is among the driest areas in Europe, with an average annual rainfall of around 300 to 400mm. The case study area covers the catchment of the Puentes reservoir in the region of Lorca (see figure 2). The study area covers in total around 16,000 ha and has a total population of only 1740 people. Besides arable land, the area consists of a mix of shrublands, stipa lands, Mediterranean deciduous forest lands, *Pinus halepensis* afforestations and, locally, badlands. An overview of the land use in the study area is presented in Table 6.

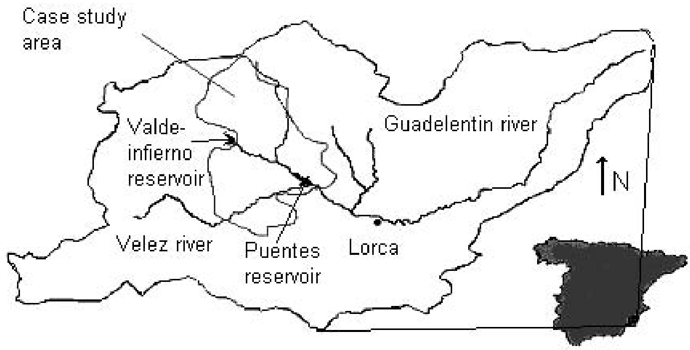

Figure 2. Map of the Puentes Catchment in the Guadalentin Basin, Spain.

(i) Problem definition and selection of the assessment approach. In the case study, the framework is used to assess the costs of erosion in the Puentes catchment. For the assessment, the Impact Assessment approach is selected. The study focuses on erosion in dryland cropland, as erosion rates are highest in this land use unit (Wesemael et al., 2003).

(ii) Identification of ecosystem functions and services. Traditionally, the Guadalentin Basin has been used for dryland agriculture (mainly cereals and tree crops) and grazing (mainly sheep and goats). Since the 1970s, there has been a strong increase in irrigated agriculture, partly driven by subsidies and increased export opportunities provided by the EU. In the Puentes Catchment, the following environmental functions can be distinguished: (1) irrigated agriculture; (2) dryland agriculture; (3) grazing; (4) hunting; and (5) nature conservation. They are strongly linked to the land use units, as described below.

1. *Irrigated agriculture*. As water is the most limiting natural resource for agricultural production, the best, and hence usually the flattest lands, are used for irrigated agriculture. Irrigated agriculture includes a variety of crops including artichokes, broccoli, tomatoes, peppers, melons, grapes, etc. Olives and almonds are the most important irrigated tree crops of the area (they are also grown, in much larger areas, as a dryland crop). In the last five years, the area of irrigated agriculture declined considerably in the case study area because many borewells have dried up or have become saline.

2. *Dryland agriculture*. Barley and wheat are the main dryland crops in the area. The large majority of the barley and wheat (durum) grown is cultivated under rainfed conditions. Usually, a piece of land is cultivated only once in two years, with the fallow year used for the build up of soil moisture. In addition, dryland agriculture includes substantial areas of tree crops, in particular almonds and olives. The area under almond cultivation has increased substantially in the last decade as a result of EU agricultural subsidies.
3. *Grazing*. Grazing, mainly by sheep and goats, is practised on fallow land, shrub land and in the forests. The number of sheep and goats has somewhat decreased in the last decades, as a result of low wool prices, but a more marked difference is the strong growth of the size of the herds, now commonly including several hundreds of animals.
4. *Hunting*. Hunting is practised by locals, but many of the lands are also leased out as hunting lands to hunting clubs from outside the area. Game includes wild boar and deer. Hunting is practised in the forests, shrublands, pine afforestations and the dryland agricultural areas.
5. *Nature conservation*. Nature conservation is another important function. As the area is among the driest parts of Europe, it's flora is particularly interesting, with a large number of drought adapted species. However, the intrinsic value of the nature in the area is difficult to translate into a monetary value – the implementation of a Contingency Valuation Study to determine this value is outside the scope of the study. The nature conservation function is highest for the natural forests occurring in the area, with additional value confined to the extensively used agricultural areas and the pine afforestations.

(iii) Bio-physical assessment of ecosystem services. The agricultural production function has been analyzed by means of the yields of the various crops. The grazing service has been quantified by analyzing the amount of offspring produced per herd, assuming that revenues from wool production are zero because the current very low wool prices. Hunting has been analyzed through assessment of land leased out to hunting organizations, and the nature conservation service through a survey of species diversity based on literature (Consejería de Agricultura, Agua y Medio Ambiente, 2002).

(iv) Economic valuation of ecosystem services. In order to assess the production value of the *agricultural functions*, the net results obtained per hectare have been calculated for each crop. This has been done by first estimating the average production per hectare multiplied with the crop value, to get the gross benefits of the functions. The net benefits have been calculated by subtracting the fixed and the variable production costs from these gross benefits. Fixed costs include the costs of stables, machinery, etc.; variable costs include the costs of agricultural inputs and labour. Labour is valued at the average labour costs for the region (from Comarcal, 2003). Subsidies have not been included in the analysis (which causes the net value of the cereals to be negative). For the irrigated crops, the costs of water have been included at the price farmers pay for it. For the *grazing function*, the same approach has been followed with the added note that the yields relate only to the production of meat (prices of wool are very low and do not justify livestock keeping in the area). *Hunting* is valued by means of the willingness-to-pay of hunters for land leases, as revealed through interviews with farmers and the local hunters' association. The *nature conservation* function is not valued in monetary terms in this study, but table 7 accounts for the non-use value of this function, in a qualitative manner, through a ranking. Based upon the occurrence and diversity of species in different land-use units, the nature conservation function receives the highest score in relation to its non-use value. A non-use value is also attributed to two other functions that contribute to the landscape of the region: grazing and dryland agriculture. An overview of the environmental functions, including an estimate of their relative values, is presented in Table 7.

(v) Ecological-economic modeling. Erosion occurs in all upland areas, with highly varying erosion rates. The most sensitive areas are the steep uplands, with marl baserock, and without vegetation cover, the least sensitive area are the flat or well protected sites. Erosion rates are lowest under natural forest and well-established pine afforestations (0.5 to 2 ton/ha/year), intermediate on the dryland agricultural fields (5- 100 ton/ha/year), high in shrublands (10 – 150 ton/ha/year) and highest on the badlands (100-1000 ton/ha/year) (Kosmas et al., 1997; Hein, 1997). A main difference between the shrublands and the upland agricultural fields is that the shrublands tend to have developed crusts that reduce infiltration rates and increase erosion. In agricultural fields, these crusts are broken through ploughing. This case study considers the costs of erosion in dryland herbaceous and dryland tree crops. It is assumed that the erosion rates under barley are representative for the dryland herbaceous crops, and the erosion rates under almond trees are representative for the dryland treecrops. For these two crops, the erosion rates on different slopes are shown in table 8.

Table 7. Environmental Functions and Approximate Values of the Puentes Catchment

Environmental function	Specification	Economic Value (euro/ha/year)	Non-use Value	Comments
Irrigated agriculture	Horticulture (e.g. broccoli, artichokes, olives)	1348 281		Approximation of net value added
Dryland agriculture	Almonds, olives, cereals	175 136 -30	+	Approximation of net value added
Grazing	Sheep and goats grazing shrubland + crop residues	16	+	Approximation of net value added
Hunting	Fees paid to land owner + local government	14		Willingness to pay for hunting licenses
Nature conservation			+++	No monetary indicator available.

Source: see Annex 1 for details on the economic valuation of the functions 'agricultural' and 'grazing'. The valuation of the function 'hunting' is based upon the average annual rent received by interviewed farmers from hunting associations (n = 15 farmers). The *non-use* value of the different functions is based upon a relative classification by the author, based upon species diversity from

Consejería de Agricultura, Agua y Medio Ambiente (2002). More information on the functions and valuation can be obtained from the author. Only the costs of erosion through its impact on dryland agriculture are calculated. These costs are assessed using a simple replacement costs analysis. It is analysed how much plant nutrients are removed from the soil on an annual basis, and what expenditure the farmer has to make to overcome this loss of nutrients, through the increased use of fertilisers. This analysis requires the assessment of (i) the total soil loss through erosion; (ii) the removal of nutrients; and (iii) the replacement costs for the farmer on the basis of the prices of fertilisers. Step (i) has been discussed above (table 4), and the other two steps are discussed below:

Assessment of the average nutrient content removed through erosion. The Cartographía Ambiental (2000) presents the average nutrient contents of the soils that are used for agriculture in the study area – including calcaric xerosols, calcaric regosols and lithosols – based on five profiles that represent average values for specific soil types. Each profile is based on around 20 to 25 different samples of the soils in each unit. For the current study, it is not known how the different soil types are distributed over the different crops, as the location of dryland and irrigated horticultural crops is often rotated and, hence, differs each year. Therefore, for this study, the average of these soil profiles has been taken – considering the first 22 centimetres of the profile. This depth relates to the depth analysed in the available data set, and takes into account that erosion in the area is a combination of overland flow, rill and gulley erosion. These analyses show that the nutrient contents of the soil are: 0.12 kg N/ton soil, 0.02 kg P/ton soil; and 0.18 kg K/ton soil.

Table 8. Mean Values for Soil Loss (Ton·Ha^{-1}·Year^{-1}) for Dry Herbaceous Crops and Almond Trees in the Region of Murcia. It is Assumed that these Rates are Representative for the Study Area.

Dry crops	**Slope (%)**	**Mean value for soil loss (ton·ha^{-1}·year^{-1})**
Herbaceous crops	<5	7.1
	5-10	22.7
	10-20	92.0
	20-30	166.8
	30-50	206.9
Almond trees	<5	8.3
	5-10	17.2
	10-20	73.7
	20-30	98.2
	30-50	121.5

Source: Ministerio del Medio Ambiente (2002).

Estimation of the costs of erosion by consideration of the price of fertilisers. In order to translate these physical data into an economic indicator, the replacement costs of the nutrients removed through erosion have been assessed, based on the prices that farmers pay for inorganic fertilisers in the area. Only the most commonly applied types of inorganic fertilisers are considered, i.e. ‘triple 15’ for the almond orchards, and a mix of ‘triple 15’, ‘ammonium sulphate’ and ‘superphosphate’ for barley. The replacement costs per unit of nutrients are presented in table 9.

Table 9. Replacement Costs per Unit of Plant Nutrient

Nutrient	Barley (€/kg compound)	Almonds (€/kg compound)
N	0.51	1.33
P	0.67	1.33
K	0.53	1.33

The overall calculations of the per-hectare costs of erosion are summarised in table 10. The loss of nutrients is calculated by multiplying the erosion rates with the nutrient contents of the soil, i.e. 0.12 kg N/ton soil, 0.02 kg P/ton soil; and 0.18 kg K/ton soil. The costs of erosion are calculated by multiplying the loss of nutrients with the costs of each nutrient and, subsequently, summing the costs of the loss of the three types of nutrients.

Table 10. Calculation of the Per-Hectare Costs of Erosion.

Crop	Slope (%)	Erosion (ton/ha/year)	Loss of nutrients (kg N, P, K/ha/year)			Costs of erosion (€/ha/year)
			N	P	K	
Dry herb crops	< 5	7.1	0.85	0.14	1.28	1.2
	5-10	22.7	2.72	0.45	4.09	3.9
	10-20	92.0	11.04	1.84	16.56	15.6
	20-30	166.8	20.02	3.34	30.02	28.3
	30-50	206.9	24.83	4.14	37.24	35.2
Dry tree crops (almond trees)	<5	8.3	1.00	0.17	1.49	3.5
	5-10	17.2	2.07	0.34	3.10	7.4
	10-20	73.7	8.85	1.47	13.27	31.4
	20-30	98.2	11.79	1.96	17.68	41.8
	30-50	121.5	14.58	2.43	21.87	51.7

Source: For the erosion rates, see table 11; for the calculations of the costs of erosion: see text for explanation.

In order to calculate the total costs of erosion for the two crops, the slope distribution of the crops has to be considered (table 8). This slope distribution is derived through superimposing a slope map (from Ministerio de Medio Ambiente, 2002) with the land-use map of the case study area (Lopez-Bermudez et al., 2002), and calculation of the area of each slope category for the two types of land-use.

Table 11. Calculation of the Costs of Erosion Through its Impact on Dryland Agriculture

Crop	**Slope (%)**	**Costs of erosion (€/ha/year)**	**Percentage of the total area (%)**	**Costs of erosion (euro/year)**
Dryland herbaceous crops (in total 5600 ha)	< 5	1.2	46	3,078
	5-10	3.9	37	7,993
	10-20	15.6	15	13,541
	20-30	28.3	2	3,328
	30-50	35.2	0	0
Dryland tree crops (in total 5150 ha)	<5	3.5	30	5,444
	5-10	7.4	35	13,377
	10-20	31.4	26	41,398
	20-30	41.8	7	14,208
	30-50	51.7	2	6,390
				Total: 109,000

In terms of the costs and benefits of erosion control measures, the measures currently applied by farmers are relatively cheap. Questionnaires showed that farmers apply only three erosion control techniques (i) contour ploughing; (ii) gulley control; and (iii) maintenance of terraces (Oñate et al., 2003). Contour ploughing, maintaining terraces, and the construction of small bunds to ensure that water is diverted from the gullies require a limited amount of extra time for the farmer. Interviewed farmers mention these costs to be less than 20 euro/ha/year – hence in the same order of magnitude as the costs of erosion. In particular, these measures are profitable for slopes exceeding around 10%. Hence, the generally low costs of erosion explain the modest interests of farmers in applying erosion control techniques in the area.

4. Conclusions and Recommendations

This paper presents three approaches that can be followed to analyze and value the economic costs of land degradation and the benefits of sustainable land management: (i) partial valuation; (ii) total valuation; and (iii) impact assessment. Partial valuation can be used to analyze the importance of ecosystems, or the benefits of sustainable management, in relation to the provision of a limited set of ecosystem services. Total valuation involves valuing all services provided by an ecosystem, and can be used, for instance, to compare the costs and benefits of different types of land cover options. It can be used to analyze the benefits provided by an ecosystem under sustainable land management with the benefits of an ecosystem under ‘regular’ management. Impact assessment is a more dynamic approach to ecosystem services valuation, which allows analyzing the economic impacts of changes in land management, for instance to assess the economic impacts resulting from the degradation of a specific ecosystem.

A number of general recommendations can be provided for the economic analysis of land degradation and SLM. First, the objective of the study needs to be clear, as the objective determines the scale and the system boundaries, the appropriate valuation methods, and the data requirements. Second, in the implementation of the study, care needs to be taken to analyze both the ecological and the economic aspects of the ecosystem services involved. In particular for the regulation services, it is often as time-consuming to quantify the service in ecological or biophysical terms (Step 3) as it is to conduct the actual valuation itself (Step 4). Third, the uncertainties in the analysis need to be discussed, the impact of the study will depend on the amount of credit it will obtain and it is important to communicate how reliable the study's outcomes are. Fourth, valuation studies require an interdisciplinary approach involving economists, ecologists, hydrologist, sociologists, etc., depending on the functions and environmental setting to be studied.

REFERENCES

Aldred, J. 1994. Existence value, welfare and altruism. *Environmental Values* 3 : 381-402.

Barbier, E.B., M. Acreman and D. Knowler, 1997. *Economic valuation of wetlands; a guide for policy makers and planners.* Ramsar Convention Bureau, Gland, Switzerland.

Cash, D.W. and S.C. Moser, 1998: Cross-scale interactions in assessments, information systems, and decision-making. In: Critical Evaluation of Global Environmental Assessments, *Global Environmental Assessment Project*, Harvard University, Cambridge, MA.

Chiang, A.C., 1992. *Elements of dynamic optimization*. McGraw-Hill Inc., Singapore.

Comarcal, 2003. Oficina Comercial Agricultura Alto Gadelentin, Lorca. Regional Agricultural Statistics.

Consejería de Agricultura, Agua y Medio Ambiente, 2002. La Estrategia Forestal de la Región de Murcia.

Costanza, R., R. d'Arge, R.S. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R.V. O'Neill, J. Paruelo, R.G. Raskin, P. Sutton and M. van den Belt, 1997a: The value of the world's ecosystem services and natural capital. *Nature* 387: 253-260.

Cummings, R.G. and G.W. Harrison, 1995. The measurement and decomposition of nonuse values: a critical review. *Environmental and Resource Economics* 5: 225-47.

De Groot, R.S., M. A. Wilson and R.M.J. Bouman, 2002: A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics*: 393-408.

Diamond, P.A., and J.A. Hausman, 1994. Contingent valuation: is some number better than no number? *Journal of Economic Perspectives* 8: 45-66

Ehrlich, P. and A. Ehrlich, 1981. Extinction: the causes and consequences of the disappearance of species. Random House, New York.

Freeman, A. M., 1993. *The measurement of environmental values and resources: theory and methods.* Resources for the Future, Washington D.C.

Hanemann, W.M., 1995. Contingent valuation and economics. In: K. Willis and J. Corkindale (eds.). *Environmental valuation: new perspectives.* CAB International, Wallingford, UK. pp. 79-117.

Hanley, N., and Spash, C.L., 1993. *Cost-Benefit Analysis and the Environment*. Edward Elgar, Vermont, USA.

Hanley, P., 1999. Cost-benefit analysis of environmental policy and management. In: J.C.J.M. van de Bergh (ed.), 1999: *Handbook of Environmental Resource Economics*. Edward Elgar, Cheltenham, UK, pp. 824-836.

Harris, G., 1999. This is not the end of limnology (or of science): the world may be a lot simpler than we think. *Freshwater Biology* 42: 689-706.

Hein, L., K. van Koppen, R.S. de Groot and E.C. van Ierland, 2006. Spatial scales, stakeholders and the valuation of ecosystem services. *Ecological Economics* 57, 209-228.

Hein, L., Assessing the costs of eutrophication control in a shallow lake ecosystem subject to two steady states. *Ecological Economics* 59, 429-439.

Hein, L. and E.C. van Ierland, 2006. Efficient and sustainable management of complex forest ecosystems. *Ecological Modelling* 190: 351-366.

Hein, L., 1997. The socio-economic impact of erosion in the Guadalentin Basin, Spain. Report 9705, Foundation for Sustainable Development, Wageningen, the Netherlands.

Holling, C.S., 1992. Cross-scale morphology, geometry and dynamics of ecosystems. *Ecological Monographs* 62: 447-502.

Howarth, R.B and R.B. Norgaard, 1993. 'Intergenerational transfers and the social discount rate'. Environmental and Natural Resource Economics 3: 337-358

Howarth, R.B., 1996. Discount rates and sustainable development. *Ecological Modeling* 92: 263-270.

Hufschmidt, M.M, James, D.E., Meister, A.D., Bower, B.T. and Dixon, J.A., 1983. *Environment, natural systems and development, an economic valuation guide*. The John Hopkins University Press, London.

Johansson, P., 1999. Theory of economic valuation of environmental goods and services. In: *Handbook of Environmental Resource Economics*. J.C.J.M. van de Bergh (ed), 1999. Edward Elgar, Cheltenham, UK, pp. 747-754.

Kolstad, C.D., 2000. *Environmental Economics*. Oxford University Press, New York, Oxford.

Kosmas, C., Danalatos, N., Cammeraat, L.H., Chabart, M., Diamantopoulos, J., Farand, R., 1997: *The effect of land-use on runoff and soil erosion rates under Mediterranean conditions*. Catena 29, 45-59.

Lal R, Stewart BA. 1990. *Soil Degradation*. Springer Verlag: New York.

Levin, S.A., 1992. The problem of pattern and scale in ecology. *Ecology* 73: 1943-1967.

Mäler, K.G. and Jeffrey R. Vincent Handbook of Environmental Economics, Volume 2 : Valuing Environmental Changes (*Handbooks in Economics*). North Holland, Amsterdam.

Martinez-Alier, J., G. Munda and J. O'Neill. 1998. Weak comparability of values as a foundation for ecological economics. *Ecological Economics*, 26(3): 277-286.

Millennium Ecosystem Assessment, 2003. Ecosystems and human well-being: a framework for assessment. *Report of the conceptual framework working group of the Millennium Ecosystem Assessment*. Island Press, Washington, DC.

Ministerio del Medio Ambiente, 2002. Inventario nacional de eosion de suelos 2002-2003. Ministerio del Medio Ambiente, Madrid.

Munasinghe, M. and A. Schwab, 1993. *Environmental economics and natural resource management in developing countries*. World Bank, Washington, DC.

Munda, G. 2004. Social multi-criteria evaluation (SMCE): methodological foundations and operational consequences. *European Journal of Operational Research*. 158 (3): 662-677.

Nordhaus, W.D., 1994. *Managing the global commons: the economics of climate change*. MIT Press, Cambridge, USA.

OECD, 1995. *The economic appraisal of environmental projects and policies, a practical guide*. OECD, Paris.

Oñate, J.J., Peco, B., Sánchez, A. and Sumpsi, J.M. 2003. Report on the results of the farm questionnaire survey in the Guadalentín target area. Deliverable 20. Medaction: *Policies for land use to combat desertification*. Madrid.

O'Neill, R.V. and A.W. King, 1998. *Hommage to St Michael: or why are there so many books on scale?* In: D.L. Peterson and V.T. Parker (editors) Ecological scale: theory and applications. Columbia University Press, New York, pp 3-15.

O'Neill, J. 2001. 'Representing People, Representing Nature, Representing the World', *Environment and Planning C: Government and Policy*, 19: 483-500.

O'Riordan, T., C. Cooper, A. Jordan, S. Rayner, K Richards, P. Runci and S. Yoffe, 1998. *Institutional frameworks for political action*. In: Human choice and climate change, Vol. 1: The societal framework. Battelle Press, Columbus, OH, pp. 345-439.

Pagiola, S. and G. Platais. 2002. *Market-based Mechanisms for Conservation and Development:* The Simple Logic of Payments for Environmental Services. In Environmental Matters–Annual Review, July 2001–June 2002 (FY 2002). Washington, DC: World Bank's Environment Department.

Pearce, D.W., A. Markandya, and E.B. Barbier, 1989. *Blueprint for a green economy.* Earthscan, London.

Pearce, D.W., and R.K. Turner, 1990. *Economics of natural resources and the environment.* Harvester Wheatsheaf, Herfordshire, UK.

Pearce, D.W. and C.G.T. Pearce, 2001. *The value of forest ecosystems. A report to the Secretariat of the Convention on Biological Diversity*, Montreal, Canada

Pimentel D, Harvey C, Resosudarmo P, Sinclair K, Kunz D, McNair M, Crist S, Shpritz L, Fitton L, Saouri R, Blair R. 1995. Environmental and economic costs of soil erosion and conservation benefits. Science 267: 1117–1123.

Posey, D.A., 1999. Cultural and spiritual values of biodiversity. *A complementary contribution to the Global Biodiversity Assessment.* Intermediate Technology Publications, London.

Tietenberg, T.H., 2000. *Environmental and Natural Resource Economics (5th edition).* Addison Wesley Longman, Inc., Reading, Massachusetts, USA.

Turner, R.K., C.J.M. van den Bergh, T. Soderqvist, A. Barendregt, J. van der Straaten, E. Maltby, E.C. van Ierland, 2000: Ecological-economic analysis of wetlands: scientific integration for management and policy. *Ecological Economics* 35: 7-23.

Wathern, P., S.N. Young, I.W. Brown and D.A. Roberts, 1986. *Ecological evaluation tecniques. Landscape Planning* 12: 403-420.

Wesemael, B., Cammeraat, E.H., Mulligan, M., Burke, S., 2003. The impact of soil properties and topography on drought vulnerability of rainfed cropping systems in southern Spain. Agric., *Ecosyst. Environ*. 94: 1-15.

ANNEX 1.
ECONOMIC VALUATION OF SELECTED ENVIRONMENTAL FUNCTIONS IN THE PUENTES CATCHMENT, SPAIN

Table A1.1 Agriculture[1]

Function	Average revenue[2] (€/ha/y)	Production costs[3] (€/ha/y)	Economic value generated (€/ha/y)
Irrigated agriculture			
- broccoli, artichoke	7525	6177	1348
- olives	3544	3263	281
Dryland agriculture			
- almonds	404	429	175
- olives	945	809	136
- wheat and barley	131	161	-30

Key:

1. In order to calculate the economic value, subsidies have been excluded from the calculations (which causes the economic value of cereal production to be negative).
2. Market revenues. These are obtained by multiplying average production (kg/ha/year) with the average prices for the crops (euro/kg). Production and price data are from Comarcal (2003) and Lopez Bermudez (1999).
3. Production costs. These include both variable costs (seeds, fertilisers, pesticides, water, labour) and fixed costs (depreciation, at 10% per year, of machinery and the irrigation system). The costs of labour are from Comarcal (2003). The other cost figures have been obtained through interviews with 40 farmers in the study area.

Table A2.2. Grazing

	Sheep	Goats	Total	Source
1. Total number of animals	12,811	3,724		Comarcal, 2003
2. Birth rate	1.3	1.3		Interviews pastoralists
3. Number of animals born	13,879	4,034		Row 1 x row 2
4. Required for replacement (20%)	2,562	745		Interviews pastoralists
5. Animals sold per year	11,317	3,289		Row 3 – row 4
6. Price per animal (euro)	50	60		Interviews pastoralists
7. Total revenues (euro/year)	565,840	197,352	763,192	Row 5 x row 6
8. Revenues supplied through grazing (75%)			572,394	Interviews of the pastoralists showed that 75% of the feed is from grazing, 25% is supplementary feed
9. Labour costs (24 pastoralists)			360,000	Shadow costs of labour per person (Comarcal, 2003)
10. Total (€/year)			212,394	Row 8 – row 9
11. Total (€/ha/year)			15.6 €/ha/year	Divided by 13590 (total area dryland agriculture + shrubland, see table 2)

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Chapter 6

THE ENVIRONMENTAL KUZNETS CURVE HYPOTHESIS AND ITS EVIDENCE FOR THE SPANISH CASE[1]

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ABSTRACT

Since the beginning of the 1990s, analysis of the relationships between economic growth and environmental pressures has been influenced by the hypothesis known as the environmental Kuznets curve or the inverted U relationship between environmental pressure and per capita income. According to this hypothesis, once a certain income level has been achieved, economic growth leads to improvements in environmental quality. This chapter analyzes and discusses the theoretical basis of the hypothesis and the available empirical evidence. The research then analyzes the relationship between per capita GDP and various atmospheric pollutants for the Spanish case. The results show that in general, the empirical evidence for the Spanish case does not support the hypothesis. The evidence found shows that by itself, economic growth does not lead to the reduction in pollution suggested by the hypothesis, but to the opposite effect, at least if the appropriate measures to avoid this are not taken. This is especially evident in the case of greenhouse gases, for which there is a strong contrast between their actual evolution and Spain's international commitments.

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1. INTRODUCTION: THE HYPOTHESIS OF THE ENVIRONMENTAL KUZNETS CURVE

The relationships between economic growth and the various environmental pressures are undoubtedly complex. Economies change over time both with regard to the relative weights of the various activities and the technologies used. We cannot therefore assume that a certain degree of growth in the scale of economic activity would imply an equivalent increase in each one of the flows causing the various environmental problems.

Since the beginning of the 1990s, analysis of the relationships between economic growth and environmental pressures has been influenced by the hypothesis known as the environmental Kuznets curve or the inverted U relationship between environmental pressure and per capita income (Figure 1). According to this hypothesis, economic growth initially has negative environmental effects, but once a critical level of per capita income has been reached, the environmental situation improves as per capita income increases. Although the empirical results are partial, diverse, and sometimes contradictory, some economists have hailed this supposed discovery as proof that "there is a clear evidence that, although economic growth usually leads to environmental degradation in the early stages of the process, in the end the best – and probably the only - way to attain a decent environment in most countries is to become rich." (Beckerman, 1992, p. 48).

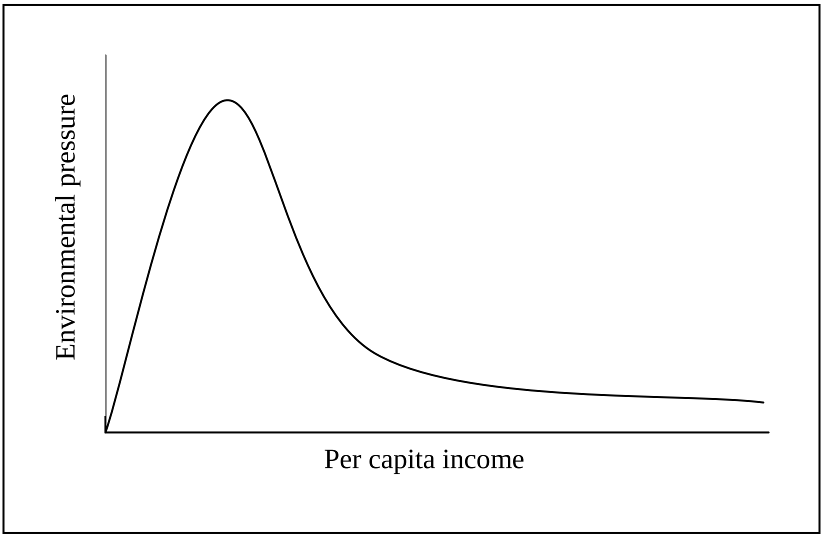

Figure 1. The environmental Kuznets curve

It seems that Panayotou (1993) was the first to use what is nowadays the usual term of environmental Kuznets curve (EKC) to refer to this hypothesis, due to its similarity with the relationship that this author suggested – with many provisos - could exist between the level of inequality and per capita income (Kuznets, 1955). Proof of the influence of the EKC hypothesis on the debate about the environmental effects of economic growth in recent years is that several academic journals devoted special issues to the subject in the 1990s, including *Ecological Economics* (vol. 25, 1998) and *Environment and Development Economics* (vol. 2, 1997). Besides the association with the name of a prestigious economist, the reasons for the widespread dissemination of the hypothesis are undoubtedly connected with, the fact that it gives a soothing perspective. It seems to allow easy harmonization of concerns for "sustainability" with the search for economic growth as the main basis of economic policy. It is significant that the first empirical study upon which the EKC is based (Grossman and Krueger 1991), which analyzed the urban concentrations of different atmospheric pollutants in different countries, was part of a work that discussed the possible environmental consequences of the Free Trade Agreement of North America. As a result, the conclusion – if the greater international trade caused greater economic growth for Mexico, it also would finally imply lesser environmental degradation - could not be more favorable to dominant economic thought. In addition, the second empirical study on the subject, by Shafik and Bandyopadhyay (1992), was particularly influential as it was extensively used in the World Bank's *World Development Report* of 1992.

Although there is empirical evidence that *some* environmental problems have diminished in rich countries, none of the pollutants that have been considered in the literature have been shown to unequivocally follow the EKC hypothesis (Ekins, 1997; de Bruyn and Heintz, 1999). Whether the econometric techniques employed enable the derivation of the casualty relationship assumed by this hypothesis has also been questioned (Stern and Common, 2001). Many authors assert that it is feasible that the EKC hypothesis only was fulfilled in the case of pollutants with local and short-term effects, in which the environmental and health impacts are clearer and the cots of mitigation are lower (case of SO_2). However, in the case of pollutants with effects that are more global, more long term and whose reduction is more complicated (such as CO_2) environmental pressure would increase with the level of income. In fact, the interesting conclusion of the study by Shafik and Bandyopadhyay (1992) was that the relationship between different indicators of environmental pressure or degradation with per capita income followed decreasing, inverted U-shaped, or growing curves depending on the cases. The hypothesis could therefore not be generalized at all to the overall relationship between the economy and the environment. Furthermore, it should be stressed that environmental degradation is not only explained by current emission flows or concentrations of emissions, but also that it depends on the history of environmental pressures that affect the situation of ecosystems and, that these sometimes involve irreversible changes (Arrow *et al.*, 1995).

A particularly important question for evaluation of the data is that one should be always aware that the improvement in an indicator may not only coexist with but also be explained by the negative behavior of another indicator. A relevant example of this possibility would be the case in which emissions associated with the use of fossil fuels were reduced due to the increasing use of nuclear energy. This problem has led some studies to consider some proxy of total environmental pressure. Suri and Chapman (1998) employ the global use of energy. However, in spite that many environmental pressures are linked to the use of energy, not all of them depend on the energy system and, furthermore, from the environmental point of view, composition in terms of energy sources is as important as the total quantity of energy employed. Another alternative is to use the global indicators obtained from the "economic metabolism" analysis from the accounts of the flow of materials that have been developed for various areas, including the Spanish case and some of its regions. In our opinion, this kind of aggregated accountability, which starts with the total flow of materials and then provides the possibility of more disaggregated analysis, is the most appropriate starting point for starting analysis of the relationships between the economy and the environment. Furthermore, given the high degree of ignorance of the environmental problems caused by the use of the various materials, it can be argued that the more materials employed, the greater the probability that more environmental impacts are generated (Spangenberg *et al.*, 1999; Hintenberger *et al.* 1997). However, we think that the aggregated indicators obtained – for both global incomes and global outcomes - cannot be directly considered as total environmental pressure indicators. This is due both to the high level of material aggregation and to the fact that many environmental pressures do not only depend on the total amount of materials and on the composition of the materials that come in an out of the economy, but also on the way that output flows are managed. Two examples related to atmospheric pollution are, firstly that the same quantity and quality of coal will cause very different sulfur dioxide emissions depending on whether there are specific measures for reducing emissions, and secondly, that the same waste generated will cause very different emissions depending on if they are incinerated or they go to a dump, and depending on whether there are systems for re-using the methane generated in the decomposition of waste. A more disaggregated analysis of the material flow should therefore be carried out, although the results should be interpreted cautiously, considering the set of changes that explain the evolution of such flows.

The difficulty in theoretically justifying that the behavior described by the EKC hypothesis dominates the relationship between economic growth and environmental pressure should be emphasized. This hypothesis is usually defined not as the mere possibility or probability that environmental growth coexists with lesser environmental pressures, but as growth itself in per capita income being the factor that explains the decline in environmental pressures. If this were the case, this must be explained by some kind of endogenous change linked to the rise in per capita income. There are three main possibilities. While the two first are, at first sight independent of the changes in individual or collective decisions concerning environmental conservation, the third focuses on these changes.

An initial possibility would be that greater per capita income involves technological innovation with a favorable bias towards the reduction of environmental pressure. There do not seem to be any convincing arguments that allow these kinds of generalizations to be made. New theories of economic growth have firmly stressed the important role of the accumulation of knowledge in economic growth, and it seems reasonable to think that this accumulated knowledge will help the different resources – and particularly natural resources – to be used in a more efficient way. However, technological change goes beyond the more efficient use of resources for basically unaltered techniques, and it involves new processes and new products with associated environmental pressures that need not be less worrying than the pressures associated with previous technologies. In fact, rich countries are not only frequent pioneers in the innovations which enable environmental pressures to be reduced, but also in those that generate the greatest environmental risks (such as the introduction of many new chemical substances or the nuclear power). It should also be stressed that the final effects of technological changes are not always foreseeable. As has been discussed in energy economics, an increase in efficiency in the use of a natural resource would also stimulate the demand for it, thereby reducing – or even, in extreme cases, canceling out - the mitigating effect of the efficiency increase. For example, if motor vehicles were more efficient and used less petrol, this would make car transport cheaper and could stimulate the use of cars and increase gasoline consumption. This has been called the “rebound effect” (Schipper 2000). The term “Jevons paradox” (Giampietro, 1999) is also used for the worst possible case in which the final effect of the increase in efficiency is increased use of the resource, referring to the famous English economist and his book *The Coal Question* (1865), in which he discussed the relationship between technological change and the use of coal.

The second potential explanation would be that the autonomous evolution of the final demand structure involves less environmental pressure as per capita income increases. The evidence justifying this argument is the increasing weight of the demands oriented to the service sector at the expense of those oriented to the industrial sector. However, this argument requires much more empirical research, given that some activities included in the services sector might generate as much or more (direct and/or indirect) environmental pressure than others in the industrial sector (such as long distance tourism, for example). In any event, this argument could at most explain the reduction in environmental pressure *per unit of income* as income increases. However, it would not explain a reduction in this pressure in absolute terms, unless we assume that the most environmentally problematic sectors are those producing inferior goods, which is not at all probable (Torras and Boyce, 1998). The change in the structure of demand would then maybe explain a “relative delinking”, but not an “absolute” one, between economic growth and environmental pressure, using the relevant differentiation of de Bruyn and Opschoor (1997) (see also Roca and Alcántara, 2002). In other words, the income elasticity of environmental pressures could be, according to this argument, lower to unity but not negative.

The third argument is that the individual preferences are those that explain that once a certain income level is achieved, the mix chosen between “producible” goods and environmental quality changes in such a way that individuals decide to consume more “environmental quality” at the expense of lower consumption of other goods and services than the potential one (or with a different composition to the one that would be the most desirable without the environmental factor). However, with different specifications, this is the idea based on the models such as those by McConnell (1997), Selden and Song (1995) and López (1994) (for a broader discussion, see Roca, 2003). All these models consider identical individuals – or, what is more or less the same, an individual that is representative of society - whose utility function depends on both the consumption level and on the pollution level. It is assumed that a “social planner” decides the mix consumption-pollution which given the existing constraints, maximizes the utility of the representative agent. The conclusion is that a high “income-elasticity of environmental quality” – i.e. that the individuals increasingly worry about environmental quality as they become richer - would make it very probable that pollution declines as income increases. However, those models share some important shortcomings.

Environmental quality is almost always a public asset whose provision level cannot be decided at individual level, but is instead mainly resolved in the political arena, and the idea that individuals decide to “buy” environmental quality is a metaphor that cannot be taken too far. Decisions about environmental policy (e.g., implementing regulations or taxes) are decided politically. Furthermore, when there are different individuals, the inequalities in preferences, in income and in the share of the costs of environmental degradation should be taken into account, and conflicts of perceptions and interests that can be solved in various ways arise. The chief conclusion is that even if we consider environmental pressure whose effects will have their entire impact on the current population of the territorial frame in which the decisions of environmental policy are taken, the evolution of per capita income will lead to different decisions, depending on how the costs and benefits of environmental degradation are distributed, how the conflicts generated by it are solved and the institutions that channel these conflicts.

The same definition of which costs and which benefits should be considered and their valuation depends on how rights are defined. This tends to be forgotten by the usual efficiency approach but is fundamental for the tradition of institutional economics of authors such as Kapp and Ciriacy-Wantrup (see Aguilera Klink (Ed.), 1995; and Bromley, 1990).

Furthermore, a country's activities frequently cause environmental pressures that are borne – at least in part - by other countries, so the possible *displacement of environmental costs* between social groups takes on another dimension. The *spatial* displacement to other territories sometimes takes place unavoidably due to the attributes of the environmental problem, such as atmospheric pollution that crosses borders or river pollution that also crosses borders downstream. There are also problems of a global nature, such as the intensification of the greenhouse effect, the effects of which will be felt by everyone, regardless of where are they are from. The greater the share of environmental effects arising beyond the frontier of the political entity that takes the decisions, the less likelihood of economic growth leading to decisions that reduce environmental pressures. In the case of more local environmental problems, there is another indirect – and very relevant - way through which a displacement of environmental costs may occur. This is external trade (Muradian and Martínez-Alier, 2001), which has led to consideration of the possible EKCs not being derived from a genuine environmental improvement, but from an exportation of environmental problems to other territories (Arrow et al., 1995, Stern et al., 1996). This means that one should not only think in terms of the possible migration of polluting industrial activities but, what is certainly much more important, the set of associated impacts on the primary activities aimed at meeting rich societies' enormous requirements for materials and energy. The other very relevant case of displacement of costs is the *intergenerational* one, where problems are transferred to the far future, and assumed preferences over personal consumption of more goods and services or more "environmental quality" are irrelevant. In this case, the incentives for renouncing higher consumption to preserve the environment may either not exist or be the result of attitudes that cannot be positively correlated with the level of per capita income. Indeed, it seems instead that the values that promote the desire of unlimited consumption favor putting these concerns aside.

The fact that some of the environmental pressures that contribute to global and long term problems are those that are more clearly positively correlated with the level of per capita income, even at very high levels, is foreseeable given the considerations mentioned above.

2. ENVIRONMENTAL POLLUTION IN SPAIN (1980-2001): A GLOBAL PERSPECTIVE

Most of the studies that we referred to in the previous section are based on data from various countries. In spite of the interest of this approach, the deduction of curves in the relationship between per capita income and pollution emissions from this kind of data involves the implicit assumption that "though the level of per capita emissions may differ over countries at any particular income level the income elasticity is the same in all countries at a given income level" (Stern, 2003, p. 5). This assumption is particularly problematic

(Dijkgraaf and Vollebergh, 1998). This is one of the reasons that a study of the experiences of each specific country from a historical perspective, like the one that we undertake in this chapter, is necessary.

In this section, we present a global vision of the relationship between per capita income and diverse atmospheric pollutants for the Spanish case during the period 1980-2001. We thus extend and enhance the analysis by Roca et al. (2001). We have considered the total flows of eight atmospheric pollutants for which we have official historical series for the period under consideration.[2] One of the interesting features of the analysis is the variety of pollutants considered. Some have global effects and some have more regional and/or local effects, some have many sources of emissions and some have far more focused emissions. In short, we have considered the main greenhouse gases considered by the Kyoto Protocol on climate change – carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O)-[3], as well as some of the main gases associated with other problems of atmospheric pollution, such as sulfur oxides (measured in units of SO_2 equivalent), nitrogen oxides (NO_x), ammonia (NH_3), carbon monoxide (CO) and non-methanic volatile organic compounds (NMVOC). The relationship between the annual pollution flows (here analyzed) of the latter five pollutants considered and the concentrations of emissions is sometimes very complex, and may depend on the greater or lesser spatial and time concentration of such flows and on the processes of dispersion and transformation among other factors (for example, some “primary” pollutants give rise to other “secondary” pollutants as tropospheric ozone). Furthermore, even though all the “classical” pollutants considered have a clear environmental relevance, there could be an increase in the emissions of a growing range of toxic pollutants for which we frequently do not have continuous data estimates, although they could be as important or more of a threat as those mentioned above (Dasgupta et al. 2002, pp. 162-163).

An initial approach to the trends during the period considered allows us to suggest some conclusions about the supposed delinking process between environmental growth and environmental pressure that would arise from the most optimistic versions of the EKC hypothesis. Global emissions (see Figure 2) during the period increase by a great deal and in the case of methane, they almost double. The emissions also increase quite significantly for other three gases considered (CO_2, NH_3 and NO_x). In the cases of N_2O and NMVOC, the trends are less clear, although the emission flow in 2001 in both cases is higher than that of 1980. Only in the case of SO_2 do emissions diminish continuously and significantly, as would be expected if the EKC was fulfilled and Spain in the early 1980s had achieved a level of per capita income high enough to be placed in the decreasing segment of such curve. The emissions of CO also fall, but in a much less pronounced way, and the reduction is concentrated in the later years.

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[2] The data are taken from the Inventory of Pollutants to the Atmosphere of the Spanish Ministry for the Environment (*Ministerio de Medio Ambiente*). We have subtracted “natural emissions” from global emissions as the relevant emissions for studying the relationship between economic growth and environmental pressure are the anthropogenic emissions. The series have been revised by the Ministry for the Environment for the data subsequent to 1990. In order to analyze the evolution of emissions, we have thus produced the series (with the year 1980 as index=100) establishing a link between the 1980-1990 data from the initial series and the 1990-2001 data from the new series.

[3] The other three gases considered by the protocol, HFCs, PFCs and SF_6, are taken into account in section 4 of this chapter, although we only have data for the period 1990-2001 for them.

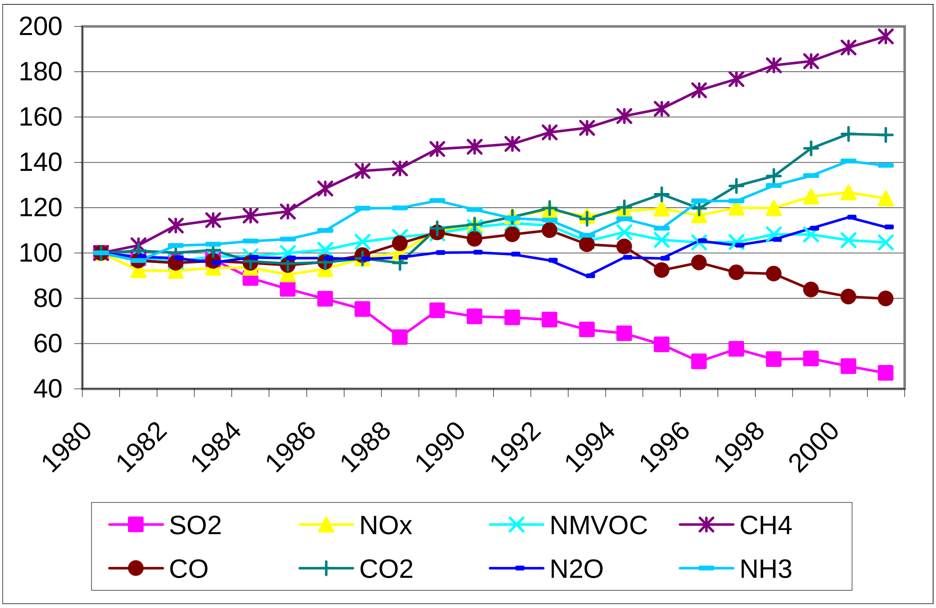

Figure 2. Evolution of emissions in Spain, 1980-2001 (1980=100).

It can be argued that for the debate about the EKC, the data employed should not be those for emissions but for per capita emissions. However, given that the Spanish population between 1980 and 2001 increased very slightly, we can see that the trends in Figure 3 are almost identical to those in Figure 2, although, logically, the index values are always somewhat lower. The important thing to highlight is that there is no reduction trend, except in the case of SO_2, and CO and perhaps a slight decrease for the NMVOC in the 1990s.

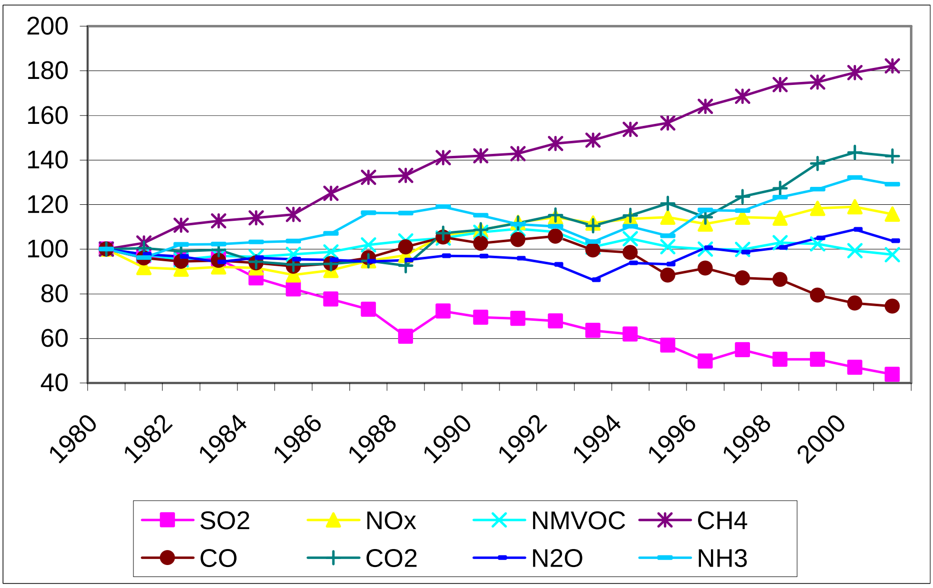

Figure 3. Evolution of per capita emissions in Spain, 1980-2001 (1980=100).

The hypothesis of the EKC does not suggest that it is the passing of years that explains the supposed reduction in environmental pressure – as would be the case if there were technological innovations that were applied in a more or less generalized way but without any direct relation to income level - but that the factor triggering this reduction is the changes involved in economic growth. In 2001, the level of "real" per capita income was considerably higher than that of 1980, but there are very different time periods in the annual variation of such income during these two decades. It is thus interesting to look at the relationship between the data of per capita emission and the data of "real" per capita GDP as can be seen in Figure 4.[4] The resulting figures are more complex but, again, we can assert that there does not seem to be any correlation between economic growth and fewer emissions. The exception is SO_2 – and, to a much lesser extent, CO - whose evolution would be the only ones expected according to the EKC hypothesis, unless one thought that Spain – with a much higher per capita consumption level than the vast majority of the world's population - is not wealthy enough to achieve the positive environmental effects associated with economic growth. We do not believe this, and in any case, this would not invite much optimism.

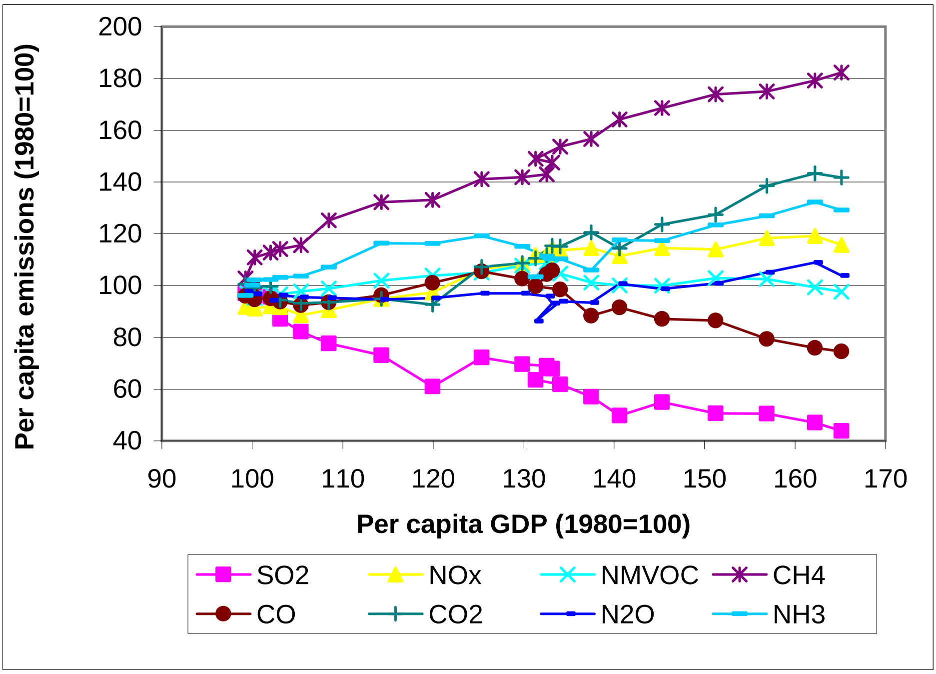

Figure 4. Relationship between "real" per capita GDP and per capita emissions in Spain, 1980-2001.

[4] The per capita GDP are taken from National Statistics Institute (*Instituto Nacional de Estadística, INE*). Given the change of base by the INE, we have linked in 1997 the series at constant 1986 prices with the series at constant 1995 prices in order to have a series at constant prices for the entire period.

3. Analysis of the Evolution of Environmental Pollutants

3.1. Analysis of the Emissions of Carbon Dioxide (CO_2)

There are two stages in the evolution of the emissions of carbon dioxide (Figure 2) during the period. There is a first stage, during most of the 1980s, in which there is a relative stabilization of emissions, and a second stage in which there is an important increase in emissions, except for non-significant temporary reductions.

As for the analysis of the activities generating these emissions, it should be stressed that the most emissions come from burning fossil fuels, for generating electricity, transport and industrial processes. The main emission sources are those of the energy sector, mainly for electricity generation, which account for more than 30%, and the transport sector, which accounted for approximately one third part of emissions in 2001, basically due to road transport. It can be seen that all the activities increased their emissions during the period 1990-2001. This is particularly true of road transport, which rose from 22.6% to 26.4% of the total[5]. As for the industrial processes without combustion, which account for 8% of total emissions, almost three quarters are generated by the cement industry.

In the last decade, several studies estimating the relationship between per capita CO_2 emissions and per capita GDP have been published, using panel data for various countries. The results are diverse, but in general, the hypothesis that greater economic growth involves a reduction in emissions is rejected. Most of the studies find a positive relationship between GDP and emissions (Shafik, 1994), while some estimate that the per capita levels from which the expected de-linking between economic growth and emissions occurs are very high, meaning that the reduction would be only hypothetical in any case (Holtz-Eakin and Selden, 1995). Others, (e.g. Grossman and Krueger, 1995) suggest the possibility of a N-shaped relationship, which would imply that there would be a second inflexion point from which economic growth would imply greater emissions. There is also a longitudinal study of the evolution of per capita carbon emissions in 16 countries of the OECD during the period 1950-1992 (Unru and Moomaw, 1998). Its interesting conclusion is that in all of them (with only one exception) there is clearly a change of behavior compared to the previous trends in the years after 1973. It is evident that this is explained by a common shock that affects economies with very different per capita income levels. The level of this income is therefore not the relevant variable for explaining the change in behavior. The idea that the trajectories of emissions change due to shocks is applicable not only to sudden changes in prices, but to many other reasons such as important changes in environmental policy that impose a new regulation (such as the prohibition of the use of lead in gasoline).

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[5] All the comments about the evolution of the different pollutants disaggregated by activities refer only to the period 1990-2001, given that there are only revised series for these years and a comparison over a longer period would mix data which are not strictly comparable. The big "activities" considered in the disaggregation of the series of the Ministry are: Combustion in production and transformation of energy, Plants of non-industrial combustion, Plants of industrial combustion, Industrial processes without combustion, Extraction and distribution of fossil fuels and geothermal energy, Use of solvents and other products, Road transport, Other modes of transport and mobile machinery, Waste treatment and disposal and Agriculture and livestock farming.

We then estimate the correlation between per capita CO_2 emissions and per capita GDP for the Spanish case using econometric methods.[6] In both this and the other estimations, we take the series in logarithms, so the coefficients can be interpreted in terms of elasticities. The coefficient of each of the independent variables shows the approximate percentage at which the dependent variable changes with a one per cent change in the independent variable.

An initial estimation of the relationship between per capita emissions and GDP showed autocorrelation problems in the estimated residuals. This first result does not show that income and emissions are not correlated, but the relationship can be distorted by the influence of other explanatory variables that are omitted from the estimation. In the production of the econometric model that was finally estimated, we take into account the fact that CO_2 emissions are explained firstly by energy consumption, which are predictably closely correlated with income, and secondly, by the structure of energy supply. Consequently, the changes in these structures act as explanatory factors in the changes in the relationship between income and emissions. Considering the evolution of the energy structure in Spain, the changes that seem most relevant are the use of coal and nuclear energy. In short, the estimated model is:

$$\ln(CO_2/Pop)_t = \beta_0 + \beta_1 \ln(GDP/Pop)_t + \beta_3 \ln Coal_t + \beta_2 \ln Nuclear_t + \varepsilon_t$$

in which besides emissions and per capita GDP, , the variables corresponding to the share that for the different years represented coal in total primary energy ($Coal_t$); and the share of nuclear energy on total primary energy ($Nuclear_t$) appear (data from International Energy Agency, IEA, various years).

Table 1. Results of the Estimation for CO_2 emissions, 1980-2000

Variable		Coefficient	t-statistic
Intercept	(β_0)	-5.04	-65.21
$\ln(GDP/Pop)_t$	(β_1)	1.37	15.13
$\ln(Coal)_t$	(β_2)	0.24	4.15
$\ln(Nuclear)_t$	(β_3)	-0.15	-10.30
Adjusted R^2: 0.97			
Durbin-Watson: 2.49			

Note: The dependent variable is $\ln(CO_2/Pop)_t$.

The results of the estimation show an important positive relationship between per capita GDP and per capita CO_2 emissions. The coefficient shows that the elasticity between both variables is greater than one (1.37). As a result, if we omitted the changes in energy structure, emissions would have tended to increase at an even greater rate than GDP. The results therefore show that far from the delinking between emissions and income posited by the EKC hypothesis occurring, not even a “relative delinking” between emissions and the use of

[6] The results we obtained, both in this section and in the others, should be taken very cautiously, given the scarce number of observations. In all the estimations we have undertaken a time series analysis, checking that there were not spurious regressions. We have checked that the different series were integrated of order one and that the residuals generated by the estimations were stationary.

primary energy has occurred. Instead, energy intensity tends to increase as GDP increases. However, the relationship between emissions and GDP is also significantly influenced by two factors that work in the opposite way. The first is the share of coal with respect to total primary energy, the increase of which causes the emissions to increase. The other is the relative importance of nuclear energy, which works in the opposite way. Of these two factors, the most important in Spain was the second, as during the eighties there was an increase in the use of nuclear energy which explains that total CO_2 emissions increased during the period by less than the increase experienced by GDP, and that in the first years there was no increase in emissions. This is a good example of how an environmental indicator cannot worsen thanks to the worsening of other indicator –in this case nuclear risks.[7]

3.2. Sulfur Dioxide (SO_2) Emissions

The emissions of SO_2 in Spain, as in most rich countries, are much focalized. More than 70% are generated in the production and transformation of energy (65% in traditional thermal energies of electricity generation and 5% in refineries) (see Table 2). Furthermore, these types of emissions can be easily reduced by applying measures that are not very costly (in some cases, simple “end of pipe” measures). Even some economic policy decisions, such as removing subsidies to national coal, would imply a net saving for society, although they are difficult to take.[8]

Table 2. Emissions of SO_2 of the Different Activities

	1990 Tons	% of total	2001 Tons	% of total	% variation
Production and transformation of energy	1,607,362	73.66	1,033,567	72.54	-35.70
Industrial combustion	340,463	15.60	237,640	16.68	-30.20
Others	234,391	10.74	153,699	10.79	-34.43
Total	*2,182,216*	*100*	*1,424,906*	*100*	*-34.70*

Source: Calculated from the Ministry of the Environment data.

Sulfur emissions take place mainly in the burning of coal and when some derivates of oil are burnt. In spite of the fact that emissions from thermal power stations have diminished, they are still the major focus of SO_2 emissions. The existence of highly pollutant thermal power stations of coal, some of which are among Europe’s worst, explains why Spanish emissions are much above the European mean (Eurostat, 1999, p. 18), a fact reflected in the concentration of emissions by province. In 1996, almost 60% of the total emissions were

[7] In 1980, the use of coal accounted for 18.1% of total primary energy, and nuclear energy accounted for only 2%. In 1985 coal's share was 27.2% (the maximum value during the period analyzed) while nuclear energy’s share was 16.6% in 1989.

[8] Before 1998, the electricity tariff included an item that was used for subsidizing autochthonous coal. The European Union considered this practice to be illegal, arguing that any help of this type should be given through the budget of the State. This item disappeared but it was simply substituted by a tax on electricity.

located in four provinces - La Coruña (26.4%), Teruel (18.4%), León (7.2%) and Asturias (6.4%)[9]. Another activity that accounts for a significant percentage, although a much lower one, is industrial combustion. The rest of activities have a much lower share of emissions, which is in all cases lower than 4%, and did not experience significant changes during the period under consideration.

We next carry out an econometric estimation to study the factors behind the reduction in emissions in depth. An initial model in which the dependent variable is per capita SO_2 and the explanatory variable is per capita GDP had autocorrelation problems, so we included various explanatory variables in the estimation. In short, we included indicators of the use of the fuels that contribute most to sulfur emissions - coal and fuel oil. In addition, we introduced a time trend to check whether the passing of time, with the adoption of measures and technological improvements, had greater explanatory power than GDP evolution. After some tests, the equation with greatest explanatory capacity was as follows:

$$\ln(SO_2/Pop)_t = \beta_0 + \beta_1 \ln(coal/Pop)_t + \beta_2 \ln(fuelthermal/Pop)_t + \beta_3 trend + \varepsilon_t$$

The result of the estimation coincides with what was predictable. The per capita use of coal and – to a much lesser extent - the per capita production of electricity in fuel oil thermal power stations have a positive effect on emissions. However, if that was all, the reduction observed in emissions - 50% during the twenty years analyzed - could not be expected.

The negative value of the time trend is very significant, and undoubtedly shows technological changes both as a result of the increasing use of fuels with lower sulfur content as well as the installation of emission control systems. These changes, which are relatively cheap, can be considered characteristic of rich countries. However, it must be stressed there it does not seem to be any clear correlation between the specific level of per capita income of each year and the evolution of emissions, as seen by the the fact that when per capita GDP is introduced into the econometric model, the coefficient is not significant and the explanatory capacity of the model diminishes.

Table 3. Results of the Estimation for SO_2 emissions, 1980-2000

Variable		Coefficient	t-statistic
Constant	(β_0)	1.45	1.65
$\ln(coal/Pop)_t$	(β_1)	0.44	4.21
$\ln(fuelthermal/Pop)_t$	(β_2)	0.08	4.07
Trend	(β_3)	-0.04	-24.28
Adjusted R^2: 0.97			
Durbin-Watson: 1.47			

Note: the dependent variable is $\ln(SO_2/Pop)_t$.

The econometric model does not allow which factors have led to the introduction of technological changes to be seen. One of them has undoubtedly been the existence of international agreements that have affected all the developed countries in general (de Bruyn,

[9] We have taken 1996 as the reference year, because we do not have the province disaggregation for the new revised series of emissions.

1997) and the objectives established at European Union level. Another factor is the pressure in some cases exerted by the affected sectors. A few years ago, there were protests by some affected towns and ecologist groups because of the high emissions from the coal-fired power station of Andorra (in the province of Teruel) that led to a legal action for ecological crime, which ended in a commitment by the firm to make a significant investment in gas desulfuration.

3.3. Nitrogen Oxide (NO_x) Emissions

In contrast to SO_2 emissions, nitrogen oxide emission focuses are much diffuse and therefore more difficult to monitor. The transport sector accounts for about 60% of emissions. In general, most activities have increased their emissions in absolute terms. This evolution is not surprising. Despite that vehicles have significantly reduced their emissions per kilometer, the expansion of road transport (of passengers and goods) in recent decades has been such that increased 'environmental efficiency' has been more than compensated for by larger scale activity. Another sector with an important role is that of energy production and transformation, basically due to the emissions from conventional thermal power stations during electricity generation.

Table 4. NO_x Emissions of the Different Activities

	1990 Tons	% of total	2001 Tons	% of total	% variation
Road transport	533,340	41.99	547,189	38.99	2.60
Other modes of transport	240,552	18.94	274,365	19.55	14.06
Production and transformation of energy	258,599	20.36	313,072	22.31	21.06
Industrial combustion plants	134,373	10.58	163,832	11.67	21.92
Others	103,366	8.13	105,045	7.49	1.62
Total	*1,270,200*	*100*	*1,403,503*	*100*	*10.49*

Source: Calculated from the Ministry of the Environment data.

Again, the estimation taking per capita GDP as the only explanatory variable presents autocorrelation problems. Because of the influence of the transport sector in the level of NO_X emissions, besides a time trend, we have included the per capita consumption of primary energy of the transport sector in the final estimation as an explanatory variable (data from IEA, various years):

$$\ln(NO_X/Pop)_t = \beta_0 + \beta_1 \ln(Transport/Pop)_t + \beta_2 trend + \varepsilon_t$$

Table 5. Results of the Estimation, for NO_X emissions, 1980-2000.

Variable		Coefficient	t-statistic
Intercept	(β_0)	2.06	2.78
$\ln(\text{Transport/Pop})_t$	(β_1)	0.72	7.69
trend	(β_2)	-0.01	-3.61
Adjusted R^2: 0.95			
Durbin-Watson: 2.02			

Note: the dependent variable is $\ln(NO_X/\text{Pop})_t$

The sign of the estimated parameters is the one expected. The inclusion of per capita GDP in the model was not significant, so the probable positive influence of per capita income on emissions is indirect, and shows itself in that at higher levels of income it uses to trigger up the use of private vehicles. The role of transport is vital in explaining the evolution of emissions. The estimation identifies a significant negative coefficient for the time trend, which would show technological changes which would not nevertheless prevent the increase in emissions.

3.4. Carbon Monoxide (CO) Emissions

CO is generated by the incomplete combustion of fuels and its evolution over time shoes that this pollutant is, together with sulfur dioxide, the only one that has clearly decreased in the period under consideration, although its evolution is more erratic than SO_2 (see Figure 2). Analysis by sector confirms that the most emissions originate in road transport. However, both the absolute value and the percentage of this sector fell significantly between 1990 and 2001. This is basically due to the improvement in the combustion of vehicles, which in this case more than compensated for the increase in the number of vehicles. Another sector with an important level of emissions is industrial combustion plants, a sector that maintained its emissions (increasing its share of the total), as well as the sector of industrial processes without combustion, a sector which experienced the greatest increase in the 11 years for which we have disaggregated sectoral data of the revised series. Most of the emissions of this latter sector are generated in the iron and steel and non-ferrous metals industries.

Table 6. CO Emissions of the Different Activities

	1990		2001		
	Tons	% of total	Tons	% of total	% variation
Road transport	2,280,098	60.03	1,362,804	47.70	-40.23
Non-industrial combustion plants	534,803	14.08	483,824	16.93	-9.53
Industrial combustion plants	227,834	6.00	207,326	7.26	-9.00
Industrial processes without combustion	303,000	7.98	358,399	12.54	18.28
Waste treatment and disposal	249,721	6.57	211,487	7.40	-15.31
Others	202,625	5.34	233,320	8.17	15.15
Total	3,798,081	*100*	2,857,160	*100*	*-24.77*

Source: Calculated from the Ministry of the Environment data.

As for the statistical correlation between these emissions and per capita GDP, an initial estimation including only the logarithm of income as an explanatory variable produced autocorrelation problems. The estimation correcting the autocorrelation shows that there is no correlation between GDP and emissions, and that only the variable of the energy employed in road transport was significant, while in the estimation in differences the intercept was significant and negative. In principle, this shows a tendency towards reducing emissions over time. It might be assumed that the improvements made to combustion motors would have contributed to reduce emissions, which is in some cases due to the characteristics of the new models that are sold at the same time in different countries with different per capita income levels.

3.5. The Emissions of Methane (CH_4)

As we have seen above, the emissions of this gas are those that have increased most of all the emissions considered in Figure 2. Table 7 shows that the main responsibility for these emissions lies with activities related to agriculture and livestock farming, with waste treatment in second place. This latter sector has been the one which has experienced the greatest increase, from accounting a quarter of total emissions in 1990 to accounting the third part in 2001. This is not surprising if we take into account the enormous increase in the generation of urban waste linked to the increase in per capita consumption, the scarce re-use of waste and the fact it goes to dumps, as well as the methane generated not being employed for energy use.

Table 7. CH_4 Emissions of the Different Activities

	1990		2001		
	Tons	% of total	Tons	% of total	% variation
Agriculture and livestock farming	912,405	63.27	1,120,597	58.35	22.82
Waste treatment	356,010	24.69	650,644	33.88	82.76
Others	173,771	12.04	149,208	7.77	-14.14
Total	1,442,186	100	1,920,449	100	33.16

Source: Calculated from the Ministry of the Environment data.

The increase in GDP, far from leading to a reduction in emissions, is therefore correlated with greater emissions, although environmental policies could lead to another result. In short, a change in the model of use of materials that generated less waste would be necessary, and more appropriate management of them.

3.6. Emissions of Other Atmospheric Pollutants: Nitrous Oxide (N_2O), Non-Methanic Volatile Organic Compounds (COVNM) and Ammonia (NH_3)

N_2O emissions are basically generated in the agriculture and livestock farming sector, which accounts for more than 60% of the total during the entire period 1990-2001. This is undoubtedly due to the use of fertilizers. None of the other activities exceeds 8%, and a decrease in industrial processes without combustion can be seen. Given the relationship of these emissions with a sector like agriculture which in terms of value, accounts for a very small part of total GDP, it is not surprising that it was not possible to identify any clear relationship between the evolution of emissions and per capita GDP.

The NMVOC are basically emitted by the sectors of agriculture and livestock farming, road transport, and for the use of organic solvents. There is however a change in the relative weights: the sectors of agriculture and livestock farming and road transport lose weight (although agriculture and livestock farming is still the most important), while the weight of the use of solvents and other products increases, and becomes second in importance. The econometric tests carried out show - after correction for autocorrelation problems - a positive correlation between emissions and economic growth, although there is also a negative coefficient in the time trend. This result is confirmed by the estimated regression between the two variables in differences, including a constant for including the effect of the passing of time.

Ammonia emissions are basically generated by processes associated with excretions of animals and, to a lesser extent, the use of nitrogen fertilizers. As a result, approximately 90% are generated in the sector of agriculture and livestock farming. Emissions in this sector have increased in absolute terms, although its weight within total emissions remained constant throughout the 1990s. Econometric analysis, after correcting autocorrelation problems, shows that per capita GDP would have a positive influence on these emissions, an influence that holds true when the use of nitrogen as fertilizer is included in the estimation.

4. The Kyoto Protocol on Greenhouse Gases Emissions: The Contrast Between the Agreements and Reality

Since the Framework Convention of the United Nations on Climate Change was signed at the "Earth Summit" in Rio de Janeiro in 1992, the Kyoto Protocol of 1997 was the first specific quantitative commitment concerning the limitation of greenhouse gas emissions. In this respect, it can be considered an important step (even more so from the current perspective that ratification of the commitment by a sufficient number of countries is not certain and the United States' rejection of it) in spite of the moderate objectives for the affected countries and the fact that, even if the objectives of the protocol were fulfilled, this would not ensure that world emissions were stabilized. The great inequality between different parts of the world and the rejection of rich countries of any agreement that recognized the equality of rights of all the inhabitants of the world to use the atmosphere made any more ambitious agreement impossible.

In the distribution of "emission rights", the European Union, as a "bubble", is committed to the average emissions of 2008-2012 of the six main greenhouse gases being as a whole an 8% lower than those of the base year considered, i.e. 1990.[10] In the subsequent internal

[10] The six gases considered by the protocol are: CO_2, CH_4, N_2O, HFCs, PFCs and SF_6. The CFCs were not included because they were already regulated by another international agreement (the Montreal Protocol). For the three latter gases, the countries could consider 1995 as the base year. The commitment refers to the aggregation of the six gases that are added in CO_2-equivalent tons according to the values of global warming potential established by the Second Report of the IPCC (1995). These values are based on the effects of greenhouse gases in a time

distribution of responsibilities decided upon by the EU, Spain, with per capita emissions lower than the EU average, was allowed to increase its emissions by 15%, while other countries are committed to much greater reductions. This is the case of Germany and Denmark, which have to reduce their emissions by 21%. In this section, we can see that the limit on the increase has been far exceeded in the Spanish case, so there is a great contrast between what is said in international forums and is ratified by parliamentary institutions and what really happens. We are faced with a global problem – about which individual governments do not have, in principle, many incentives to act - and one that is related with multiple production and consumption activities and, above all, with the energy system. The changes needed for limiting emissions – and even more so for reducing them - should therefore be far-reaching.

Figures 5 and 6 show the evolution of the emissions of the six gases considered since 1990. A correct interpretation needs to consider that the present relative importance of each gas in the increase of the greenhouse effect is very different, as shown by Table 8, which is based on the Spanish data for 2001, and where the IPCC conversion factors have been applied.[11] The evolution of total emissions depends above all on the three greenhouse gases already analyzed in the previous sections, and particularly on CO_2. The role of the hydrofluorocarbons (HFCs, substitutes for CFCs in various uses such as refrigeration and air-conditioning equipment or aerosols) and above all, sulfur hexafluorur (SF_6, which is used in electric equipment) and the perfluorocarbons (PFCs, linked to the production of aluminum) is currently very marginal. In any event, total emissions of these three gases have increased, in spite of the heavy reduction of PFC emissions in recent years.

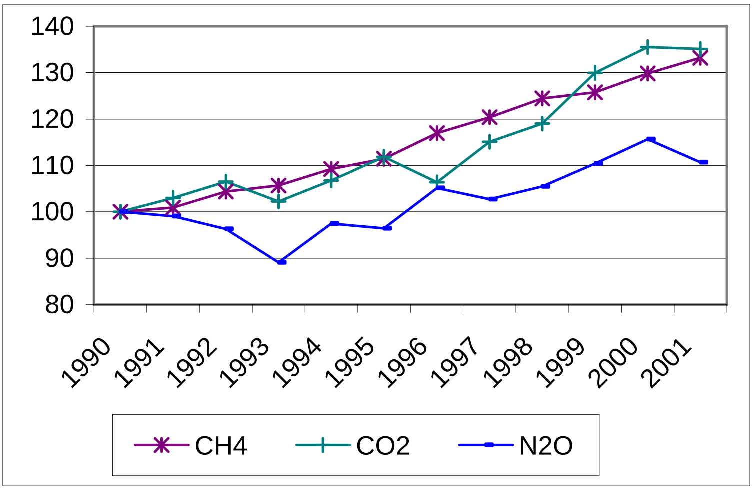

Figure 5. Evolution of Greenhouse Gases Emissions (CO_2, N_2O and CH_4), 1990-2001. (1990=100)

frame of 100 years. We have used these equivalences in the calculations in this chapter. To be more precise, the commitment does not refer to “gross emissions” of greenhouse gases, but to “net emissions”, i.e., taking away the additional carbon fixed by the possible increase of forestry area.

[11] The conversion factors are: 1 for CO_2, 21 for CH_4, 310 for N_2O, 23,900 for SF_6 while for the group of the PFCs, the values oscillate between 6,500 and 9,200 depending on the specific gas, and for the group of the HFCs they vary between 140 and 11,700.

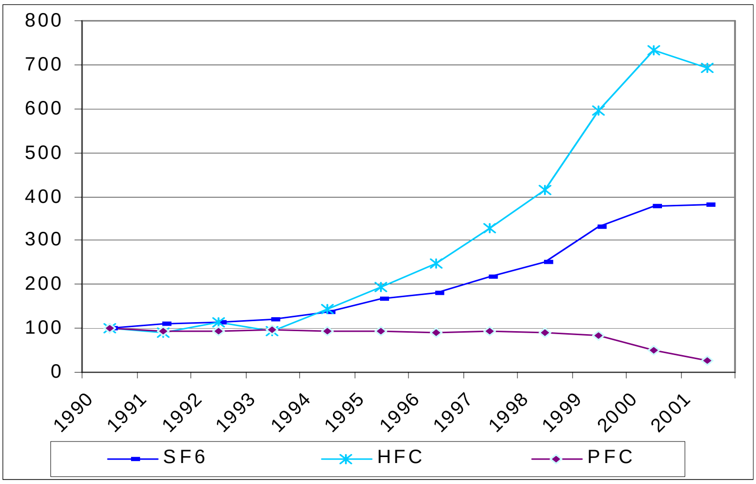

Figure 6. Evolution of the Greenhouse Gases (SF_6, HFCs and PFCs), 1990-2001

Table 8. Percentage with Respect to Total CO_2-Equivalent Emissions in Spain, 2001

CO_2	CH_4	N_2O	SF_6	HFC	PFC	Total
80.97	10.67	6.85	0.06	1.40	0.06	100

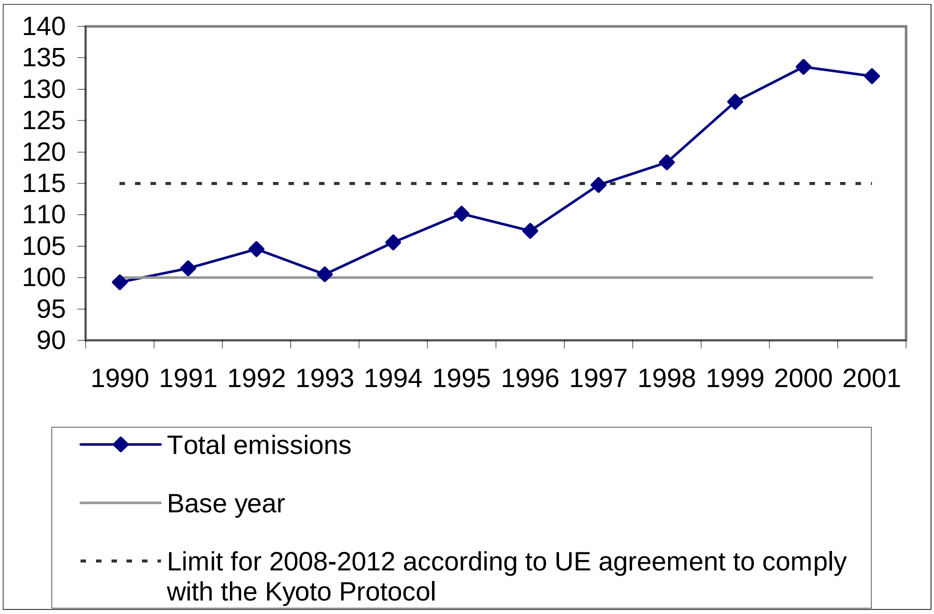

Figure 7. Evolution of Total CO_2-Equivalent Emissions, 1990-2001, Compared to the Base Year of the Kyoto Protocol

Figure 7 shows the evolution of total emissions for all six gases considered and the spectacular increase in emissions: 32% compared to the base year[12]. Together with Ireland, Spain is the EU country that is moving furthest away from its commitment, as highlighted by the European Agency for the Environment in its inventory on emissions published in May 2003.

5. CONCLUSION

Since the early 1990s, analysis of the relationships between economic growth and environmental pressure has been influenced by the EKC hypothesis. According to this hypothesis, once a certain level of income is reached, further economic growth is followed by improvements in environmental quality. In the first section of this chapter we discussed the theoretical and empirical basis for this hypothesis, which at most is able to explain the evolution of a specific environmental problem. In the two next sections, we provided empirical evidence on the relationship between per capita GDP and various atmospheric pollutants in the Spanish case. In short, we analyzed the data for the period between 1980 and 2001 of eight atmospheric pollutants with very different characteristics, some with global effects and some with more regional and/or local effects, some with multiple emission focuses and others with much more concentrated emissions. Only in the case of SO_2, and CO to a lesser extent, do emissions fall as would be expected if the EKC hypothesis was fulfilled and Spain had achieved a per capita level high enough to be already placed in the descending segment of such curve. This result coincides with theoretical predictions and international empirical evidence which indicate that a greater and more effective institutional response may be expected when the effects of emissions are more perceptible at local level and relatively easy to avoid. However, even in these cases, a clear relationship between income increase and the reduction of emissions is not found. In the final section, we analyzed the evolution of greenhouse gases, the expected costs of which are global and long term. These characteristics and the lack of political will to meet the commitments arising from the Kyoto Protocol explain the abrupt rise in these emissions by more than double the weak commitment not to increase them by more than 15% in the period 2008-2012 compared to 1990 levels. In the last months alone, in response to the directive concerning emissions trading within the European Union that might involve a high purchase cost of rights for Spanish firms, there have been many complaints by big firms and employers' organizations, showing a concern that contrasts with the evident lack of concern about participating in any serious strategy for reducing emissions during the previous years.

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[12] According to the data for 2002 of Nieto and Santamarta (2003), the mild reduction for 2001 was followed by a significant increase in emissions in 2002, meaning that the accumulated increase compared to the base year would be 38%. Given that the base year for some gases is 1995, the value for 1990 is not exactly 100, but somewhat lower.

REFERENCES

Aguilera Klink, F. (Ed.) (1995). *Economía de los Recursos Naturales: un Enfoque Institucional. Textos de S.V. Ciriacy-Wantrup y K. W. Kapp*. Madrid: Fundación Argentaria.

Arrow, K., Boling, B., Costanza, R., Dasgupta, P., Folke, C., Holling, S., Jansson, B.-O., Levin, S., Mäler, K.-G., Perrings, C., & Pimentel, D. (1995). Economic growth, carrying capacity and the environment. *Science, 268,* 520-521.

Beckerman, W. (1992). Economic growth and the environment: whose growth? Whose environment?. *World Development, 20*, 481-496.

Bromley, D. W. (1991). The ideology of efficiency: searching for a theory of policy analysis. *Journal of Environmental Economics and Management, 19,* 181-194.

Dasgupta, S., Laplante, B., Wang, H., & Wheeler, D. (2002). Confronting the environmental Kuznets curve. *Journal of Economic Perspectives, 16,* 147-168.

De Bruyn, S. M., & Opschoor, J. B. (1997). Developments in the throughput-income relationship: theoretical and empirical observations. *Ecological Economics, 20,* 255-268.

De Bruyn, S. M., & Heintz, R. J. (1999). The environmental Kuznets curve hypothesis. In J. Van den Bergh (Ed.), *Handbook of Environmental and Resource Economics* (pp. 656-677). Cheltenham: Edward Edgar.

Dijkgraaf, E., & Vollebergh, H. R. J. (1998). Growth and/or (?) Environment: Is there a Kuznets Curve for Carbon Emissions?. Document presented at the 2nd Biennial Meeting of the European Society for Ecological Economics, Geneva, 4-7 March.

Ekins, P. (1997). The Kuznets curve for the environment and economic growth: examining the evidence. *Environment and Planning A, 29,* 805-830.

EUROSTAT. (1999). *Towards Environmental Pressure Indicators for the EU*. Luxembourg: European Commission/Eurostat.

Giampietro, M. (1999). Economic growth, human disturbance to ecological systems, and sustainability. In L. R. Walker (Ed.), *Ecosystems of Disturbed Ground*. Amsterdam: Elsevier.

Grossman, G. M., & Krueger, A. B. (1991). Environmental Impacts of a North American Free Trade Agreement. NBER Working Paper 3914, National Bureau of Economic Research (NBER), Cambridge.

Hinterberger, F., Luks, F., & Schmidt-Bleek, F. (1997). Material flows vs. 'natural capital' What makes an economy sustainable? *Ecological Economics*, *13*, 1-14.

Holtz-Eakin, D., & Selden, T. M. (1995). Stoking the fires? CO_2 emissions and economic growth. *Journal of Public Economics, 57*, 85-101.

International Energy Agency (IEA). (Various years). *Energy Balances of OECD countries*. Paris: OECD.

Kuznets, S. (1955). Economic growth and income inequality. *American Economic Review, 45*, 1-28.

López, R. (1994). The environment as a factor of production: the effects of economic growth and trade liberalization. *Journal of Environmental Economics and Management, 27*, 163-184.

McConnell, K. (1997). Income and demand for environmental quality. *Environment and Development Economics, 2,* 383-399.

Muradian, R. & Martínez Alier, J. (2001). Trade and environment: from a 'Southern' perspective. *Ecological Economics, 36*, 281-297.

Nieto, J. & Santamarta, J. (2003). *Evolución de los gases de efecto invernadero en España (1990-2002)*. Madrid: Comisiones Obreras / Revista WorldWatch.

Panayotou, T. (1993). Empirical Tests and Policy Analysis of Environmental Degradation at Different Stages of Economic Development. Working Paper WP 238, Technology and Environment Programme, International Labour Office, Geneva.

Roca, J. (2003). Do individual preferences explain the Environmental Kuznets curve? *Ecological Economics, 45*, 3-10.

Roca, J., Padilla, E., Farré, M., & Galletto, V. (2001). Economic growth and atmospheric pollution in Spain: discussing the environmental Kuznets curve hypothesis. *Ecological Economics, 39*, 85-99.

Roca, J., & Alcántara, V. (2002). Economic growth, energy use, and CO_2 emissions. In J. R. Blackwood (Ed.), *Energy Research at the Cutting Edge*. New York: Novascience.

Selden, T. M., & Song, D. (1995). Neoclassical growth, the J curve for abatement and the inverted U curve for pollution. *Journal of Environmental Economics and Management, 29*, 162-168.

Schipper, L. (2000). On the rebound: the interaction of energy efficiency, energy use and economic activity. An introduction. *Energy Policy, 28*, 351-353.

Shafik, N. (1994). Economic development and environmental quality: An econometric analysis. *Oxford Economic Papers, 46*, 757-773.

Shafik, N., & Bandyopadhyay, S. (1992). *Economic Growth and Environmental Quality: Time Series and Cross-Country Evidence. Background Paper for World Development Report 1992*. Washington D.C.: World Bank.

Spangerberg, J. H., Hintergerger, F., Moll, S., & Schütz, H. (1998). Material flow analysis, TMR and the mips concept: A contribution to the development of indicators for measuring changes in consumption and production patterns. *International Journal on Sustainable Development, 1*, 491-505.

Stern, D. I., & Common, M. S. (2001). Is there an environmental Kuznets curve for sulfur? *Journal of Environmental Economics and Management, 41*, 162-178.

Stern, D. I., Common, M. S., & Barbier, E. B. (1996). Economic growth, trade and the environment: implications for the environmental Kuznets curve. *World Development, 24*, 1151-1160.

Stern, D. I. (2003). The Environmental Kuznets Curve. Internet Encyclopaedia of Ecological Economics, International Society for Ecological Economics, available at: http://www.ecologicaleconomics.org/publica/encyc.htm

Suri, V., & Chapman, D. (1998). Economic Growth, trade and the environment: implications for the environmental Kuznets curve. *Ecological Economics, 25*, 195-208.

Torras, M., & Boyce, J. K. (1998). Income, inequality and pollution: a reassessment of the environmental Kuznets curve. *Ecological Economics, 25*, 147-160.

Unruh, G. C., & Moomaw, W. R. (1998). An alternative analysis of apparent EKC-type transitions. *Ecological Economics, 25*, 221-229.

World Bank. (1992). *World Development Report*. New York: Oxford University Press.

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Chapter 7

SUSTAINABILITY RANKING AND CRITICAL SUSTAINABILITY FACTORS OF NATIONS

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ABSTRACT

It is nowadays accepted that human welfare cannot be sustained without the preservation of natural resources and without ensuring social coherence and stability. As a result, there has been growing interest among policy-makers and scientists in sustainability and sustainable development. Sustainable decision-making requires systematic methods for measuring and improving sustainability. In this work we present a model which uses fuzzy logic and combines basic indicators of environmental integrity and human welfare to provide a measure of sustainability. Using sensitivity analysis, we identify those indicators that each country should improve if it is to improve its sustainability status. Finally we provide a sustainability ranking of various EU countries together with the most critical indicators.

INTRODUCTION

In the last decades, we have witnessed dramatic environmental changes as a result of human economic activity. Global warming, stratospheric ozone depletion, species extinction, and exhaustion of natural resources provide strong evidence that development is currently unsustainable. We have become accustomed to daily stories about unusual weather phenomena attributed to global warming, toxic spills, depletion of fisheries, deforestation, species extinction, desertification, air, soil, and water pollution and so on. Most importantly, these problems have reached this prominence due to an ever increasing volume of scientific research, which, in most cases, has performed a commendable job in exposing the facets of the environmental predicament.

The present human population of 6.5 billion is expected to reach 8 billion by 2025 [1]. The environmental impact of an additional 1.5 billion of humans is not known precisely, but it is certain that it will be detrimental. The strain of overpopulation is not only environmental. Many societal aspects such as health, economy, education and political systems will be tried.

It is a fast changing world with ever increasing uncertainty about what is in store in the future. Dramatic changes occur in only a few years or just decades and so the time the society is given to adjust to these changes is very short. In the past, climate change progressed over thousands of years, allowing humans and other species enough time to evolve or move to more suitable habitats. In the present global warming era, species have only 50 to 100 years to find new hospitable places. But some plant and animal species move slowly or don't have where to go and therefore, are doomed.

A question thus arises as to the consequences of our actions to us and the environment. How sustainable is our present course? Such a question begs a definition and a measurement scheme of sustainability.

Sustainable development is human development that conserves the natural environment now and for the generations to come. When formulating policies for sustainable development, policy makers need to integrate the two conflicting criteria of human welfare and environmental integrity. Therefore, sustainable decision-making requires a tool for measuring the overall sustainability of the socio-environmental system and a tool for comparing different scenarios of development.

We all would like to develop sustainably provided:

a) We have developed a definition or description of sustainable development
b) We possess mechanisms and we have the political will to do so.

This chapter deals with the first condition and provides some mathematical tools to answer it. The second question, although of the utmost importance and complexity, is outside the scope of this chapter. The main reason is that question 2 is not scientific but has to do with values, ethics, and politics.

People speak of sustainability because the environmental crisis has become obvious to most of them. A society is an extremely complex system and the possibilities of action within this system are many. When a problem arises, its solutions depend on the knowledge, values, and the willingness of the decision makers and the citizens to undertake a certain course of action. Thus complex societal problems such as sustainability are not only technical but primarily cultural and political [2].

Roughly speaking, a society is sustainable if it provides basic necessities and happiness to the present generation and generations to come over a very long time. We shall come back to this.

In the past, sustainability was not a major issue if at all, because the size of human population and the level of consumption, on average, were such that the terrestrial or aquatic ecosystems were not threatened by human actions on a global scale. The scale of the economies was small and there was always a frontier where humans could expand.

Today, in contrast, we have reached or exceeded the capacity of the earth to supply resources and absorb pollution. Anything we do has environmental consequences which, when multiplied by all people and then added up, explain for example why global warming is occurring. In the past, human society relied on the consumption of solar energy to satisfy its needs for food and cover. Plants and animals are the products of photosynthesis and they provide food, building materials, clothing, medicines, a stable climate, a stable atmosphere, esthetic pleasure and so on. Fossil fuels are products of solar energy. Plants and microorganisms trapped in the strata of the earth provide coal, oil, and natural gas, which in essence represent the nuclear energy of the sun which these species received via photosynthesis thousands of years ago.

Today we consume the ecosystem itself to support our growth. We remove forests and fish beyond replenishment, pump water beyond recharging, drive species to extinction, poison the air, water, and soil, sometimes irreversibly.

It has been estimated [3] that 90% of the economic activity in rich countries satisfies or aims to satisfy wants which later become needs. Such is the story of cable TV, cellular phones, fax machines, SUV's, electronic gadgets, large houses, air travel. In some sense we are all trapped in an endless cycle of consumption. Still, consumption patterns differ from place to place despite a tendency to homogenization aided by the enormous advances in communication, transportation, and international trade. People exhibit different consumption behavior in the U.S., Greece, Nepal, Chad, or Peru. This is due to their economic power and in large measure to their personal and social values. Thus sustainability, while dealing with the economic activity, is also about values. The environmental behavior of a nation is also affected by values, among other things. The environmental ethic in Northern Europe for example, is quite advanced, although one may argue that the Northern Europeans enriched themselves by destroying enough of their environment so they can afford now to protect whatever remains of it. Others argue that poor countries supply rich countries with cheap labor, resources, and dumps for their toxic or radioactive garbage becoming heavily polluted in the process. However, the point that sustainability is mainly a matter of values remains valid. There are many nongovernmental organizations, scientists, or citizens who raise awareness about unfair treatment of poor countries and press for legislation and effective measures against this type of exploitation.

WHAT IS SUSTAINABILITY?

The biological and physical environment, in two words the global ecosystem, provides the economy with:

a) **Resources** such as wood, metals, minerals, fuels, food, drugs, water, air, fiber and so on.
b) **Services** as for example the cycles of H_2O, C, CO_2, N, O_2; photosynthesis; soil formation.
c) **Mechanisms to absorb** waste.

According to an attempt to monetize all three global services [4], their total value in 1997 was $33 trillion/year. This number could be disputed on various grounds, but its enormity gives us an idea of the importance of the environment.

Economic growth is based on these three services and since the global ecosystem does not grow, economic growth cannot continue indefinitely. If this is so, and since the human mind cannot stop inventing new things and improving existing systems and processes, what should be done to preserve a basic level of satisfaction of necessities that would provide happiness to our society for a long time? In other words, how can we achieve sustainability?

It is claimed by several economists that substituting one form of capital for another, solves the problem of finiteness of the ecosystem and the concomitant existence of limits. A worker can be replaced by a robot, wood for the shipping industry by iron, or metals for aircraft by synthetic materials. But substitution has its limits too which result from ecological limits. In the long run total output will go down. Replacing harpoon whaling by modern ships depletes the population of whales and substituting traditional nets with drift nets depletes fisheries. More importantly, substitution often is impossible owing to the scale of the global ecosystem or the laws of nature. There is no substitute for a stable climate which we are changing rapidly with direct consequences, as well as there is no substitute for extinct species and their services to nature.

The most basic economic indicator of the state of an economy is the gross national product or GNP, which is the total value of all final goods and services produced in an economy in a given period. If goods and services are produced within a country, then we have the gross domestic product (GDP). GNP is a useful indicator but also one-sided and deceptive. If someone cuts down all forests or depletes fisheries, the GNP goes up even though economic catastrophy is near. Such growth of the GNP is unsustainable. In fact, GNP and consumerism go hand in hand.

There have been attempts to correct the single-sidedness of GNP or GDP. One such attempt [5] defines the Genuine Progress Indicator (GPI). GPI includes the value of household work, parenting, volunteer work, services of customer durables, highways and streets, which GDP ignores. It also subtracts defensive expenses such as auto accidents, social costs such as cost of crime, and depreciation of environmental assets.

As an example, the GDP and GPI per capita for the U.S, in the year 2000 were $35,000 and $11,554 respectively.

Sustainable development on the other hand does not necessarily mean growth, but improvement of the various societal sectors, as for example health, education, or the state of the environment. So what is sustainable development (SD)? To some people it is a contradiction in terms.

There are economists who believe that SD means

- Sustaining economic growth indefinitely while others see it as
- The ability to maintain desired social values, institutions, cultures or other social characteristics or as
- Development (improvement) that can be continued for a long time.

In 1980 the International Union for the Conservation of Nature and Natural Resources (IUCN) wrote in its World Conservation Strategy (WCS) [6] that it espouses "the overall aim of achieving sustainable development through the conservation of living resources." Since then the concept of SD gained momentum as the environmental degradation intensified and became obvious to many. Today most proponents of SD take it to mean:

The existence of environmental and economic conditions needed to sustain human well-being at a given level over a very long time.

- In more general terms, SD according to the Brundtland Report [7] is:
- Development that meets the needs of the present without compromising the ability of future generations to meet their own needs.
- In a slightly different way, IUCN defined SD as "development that improves the quality of human life within the carrying capacity of supporting ecosystems."SD was a central subject of the Earth Summit held in Rio de Janeiro in 1992. Agenda 21, which resulted from the meeting, gives a comprehensive list of actions needed to achieve SD. Leaders from over 150 states committed themselves to undertaking those actions that will render development sustainable.
- It is common to classify SD as strong and weak.
- Strong sustainability is primarily focused on the environment and ignores economics such as the cost of achieving sustainability. It considers pollution, emissions, biodiversity, soil erosion etc.
- Weak sustainability is basically economic sustainability. It ignores environmental inputs to the economy and considers consumption, economic growth, financial value etc.

In this chapter sustainability integrates both aspects, environmental and economic. Two questions arise in the context of SD:

1. *What is the space over which sustainability is considered?*
   Answer: An ecosystem, a region, a country, and since pollution often has no borders, the globe. As the scale becomes smaller, the boundaries of the system the sustainability of which we examine become uncertain. The sustainability of California, for example, depends on regional attributes such as local pollution, water resources, coastal areas, agriculture, education, health etc., but also external factors, e.g., climate, ozone depletion, water resources of the Colorado river, foreign imports and so on. In this chapter the most common scale of SD is that of a country.
2. *What is the time horizon of sustainability?*
   Answer: It depends on the specific attribute. For example, the climate exhibits several periodicities due to a change of the earth's orbit from almost circular to elliptic with a periodicity of 100,000 years. Also the axis of motion of the earth has two periodicities of 40,000 and 20,000 years respectively. Thus the climate naturally oscillates between cold and warm with a variation of 5^{o} C or more. Incidentally such climatic changes occur over periods of thousands of years. Today's global warming occurs within only a few decades.

The fluidity of time scales in the context of SD is shown in two more examples. Pest problems have time scales of the order of 20 years. The life expectancy of species on the other hand is between 1 and 5 million years on average, although from the fossil record we know of species that survived only a few thousand years before becoming extinct.

Perhaps the most important aspect of time is that of uncertainty. We simply cannot plan our society for thousands of years into the future. Often even a few decades are beyond our predictive and planning capabilities. We have only long-term goals and intentions and all the time we make adaptive decisions which bring us as close to our goals as possible.

In the sequel a mathematical methodology will be developed, whereby sustainability will be assessed via fuzzy logic. In effect the model that assesses sustainability serves also as a definition of the concept.

Fuzzy Sets

Fuzzy set theory and its attendant *fuzzy logic* were developed by Lotfi Zadeh in 1965 to handle semantic and subjective ambiguity. In classical logic the number 300 is an integer whereas 300.7 is not. The same number, however, could be considered large, small, very large, very small etc. depending on context and subjective opinion. Therefore, the number 300 could be considered large to a certain degree, very large to another and so on. We then have various *linguistic values* of one *linguistic variable* which are true to some degree. This degree, subjective as it may be, varies from 0 to 1.

In classical set theory an element of a set either belongs or does not belong to the set. In fuzzy set theory an element belongs with a *membership grade* in the interval [0, 1]. All membership grades together form the *membership function*. A classical set is often called *crisp* as opposed to fuzzy.

Definition 1. A set is a collection of elements or members. A set may be an element of another set.

Definition 2. Let X be a set of elements x. A fuzzy set A is a collection of ordered pairs

$(x, \mu_A(x))$ for $x \in X$. X is called the *universe of discourse* and $\mu_A(x)$: $X \rightarrow [0, 1]$ is the membership function.

The function $\mu_A(x)$ provides the degree of fulfillment of x in X. When X is countable, the fuzzy set A is represented as

$$A = \mu_A(x_1)/x_1 + \mu_A(x_2)/x_2 + \ldots + \mu_A(x_n)/x_n$$

This is a common notation in the context of fuzzy sets. It simply states the elements x_i of X and the corresponding membership grades.

Example 1

Consider the temperature of a patient in degrees Celsius. Let $X = \{36.5, 37, 37.5, 38, 38.5, 39, 39.5\}$. The fuzzy set A = "High temperature" may be defined

$$A = \{\mu_A(x)/x \mid x \in X\}$$

$$= 0/36.5 + 0/37 + 0.1/37.5 + 0.5/38 + 0.8/38.5 + 1/39 + 1/39.5$$

where the numbers 0, 0.1, 0.5, 0.8, and 1 express the degree to which the corresponding temperature is high.

DEFINITION 3. The support of a fuzzy set A is the crisp set of all elements of X with nonzero membership in A, or symbolically

$$S(A) = \{x \in X \mid \mu_A(x) > 0\}$$

Example 2

Take Example 1. $S(A) = \{37.5, 38, 38.5, 39, 39.5\}$

Operations of Fuzzy Sets

The basic notions concerning operations on crisp sets will now be extended to fuzzy sets.

DEFINITION 4. Two fuzzy sets A and B in X are equal if $\mu_A(x) = \mu_B(x)$, $\forall x \in X$. We write $A = B$.

DEFINITION 5. A fuzzy set A in X is a subset of another fuzzy set B also in X if

$$\mu_A(x) \leq \mu_B(x), \forall x \in X.$$

The following definitions are concerned with the complement, the union, and intersection of fuzzy sets as defined by Zadeh. It should be stressed that these definitions, intuitively appealing as they may be, are by no means unique due to the nature of fuzzy sets. Others have proposed different definitions.

DEFINITION 6. The following membership functions are defined:

a) Complement $A;^{-}$ of a fuzzy set A in X

$$\mu_{A;^{-}} = 1 - \mu_A(x) \text{ , } x \in X$$

b) Union $A \cup B$ of two fuzzy sets in X

$\mu_{A\cup B} = \max[\mu_A(x),\ \mu_B(x)],\ x\in X$

c) Intersection $A\cap B$ of two fuzzy sets in X

$\mu_{A\cap B} = \min[\mu_A(x),\ \mu_B(x)],\ x\in X$

Example 3

In the context of Example 1 let us define a new fuzzy set B = “Dangerous temperature” as $B = \{0/37.5, 0.1/38, 0.2/38.5, 0.5/39, 0.8/39.5, 1/40\}$. According to Definition 6 we have

$A\cup B$ = “High or dangerous temperature”
= 0/36.5 + 0/37 + 0.1/37.5 + 0.5/38 + 0.8/38.5 + 1/39 + 1/39.5 + 1/40

$A\cap B$ = “High and dangerous temperature”
= 0/36.5 + 0/37 + 0/37.5 + 0.1/38 + 0.2/38.5 + 0.5/39 + 0.8/39.5 + 1/40

A;¯ = “Not high temperature”
= 1/36.5 + 1/37 + 0.9/37.5 + 0.5/38 + 0.2/38.5 + 0/39 + 0/39.5

The definitions of an intersection and union can be developed from a more general point of view. An intersection may be defined via a *t-norm*.

DEFINITION 7. A t-norm is a bivariate function t: $[0, 1]\times[0, 1]\rightarrow[0, 1]$ satisfying:

a) $t(0, 0) = 0$

b) $t(x, 1) = x$

c) $t(x, y) \le t(w, z)$ if $x \le w$ and $y \le z$ (monotonicity)

d) $t(x, y) = t(y, x)$ (symmetry)

e) $t[x, t(y, z)] = t[t(x, y), z]$ (associativity)

This definition provides the tools of combining two membership functions to find the membership function of $A\cap B$. For the union $A\cup B$ we have correspondingly the definition of the *t-conorm* or *s-norm*.

DEFINITION 8. A t-conorm is a bivariate function c: $[0, 1]\times[0, 1]\rightarrow[0, 1]$ satisfying:

a) $c(1, 1) = 1$

b) $c(x, 0) = x$

c) $c(x, y) \leq c(w, z)$ if $x \leq w$ and $y \leq z$ (monotonicity)

d) $c(x, y) = c(y, x)$ (symmetry)

e) $c[x, c(y, z)] = c[c(x, y), z]$ (associativity)

From these definitions, for two fuzzy sets $\mu_A(x)$ and $\mu_B(x)$, we obtain $\mu_{A\cap B}(x) = t[\mu_A(x), \mu_B(x)]$ and $\mu_{A\cup B}(x) = c[\mu_A(x), \mu_B(x)]$.

Example 4

The following are examples of t-norms and t-conorms.

Name	**$t(x, y)$ (intersection)**	**$c(x, y)$ (union)**
Algebraic product-sum	$x\,y$	$x + y - x\,y$
Hamacher product-sum	$\dfrac{xy}{x + y - xy}$	$\dfrac{x + y - 2xy}{1 - xy}$
Einstein product-sum	$\dfrac{xy}{1 + (1 - x)(1 - y)}$	$\dfrac{x + y}{1 + xy}$
Bounded difference product-sum	$\max(0, x + y - 1)$	$\min(1, x + y)$
Dubois-Prade ($0 \leq p \leq 1$)	$\dfrac{xy}{\max(x, y, p)}$	$1 - \dfrac{(1 - x)(1 - y)}{\max[(1 - x), (1 - y), p]}$
Minimum-maximum (min-max)	$\min(x, y)$	$\max(x, y)$

It is worth noting that, contrary to what holds in set theory, when A is a fuzzy set in X, then $A\cup\bar{A} \neq X$ and $A\cap\bar{A} \neq \varnothing$ because it is not certain where A ends and $\bar{A}$ begins. This is the fundamental reason that places probability and fuzzy sets apart, although both handle uncertainty. Probability is suitable for a different kind of uncertainty than fuzzy sets and, in our opinion, the debate about which discipline is "better" or "correct" is rather besides the point. Each of them performs its own scientific function successfully within its capabilities and limitations.

LINGUISTIC VARIABLES

Loosely speaking a *linguistic variable* is a variable "whose values are words or sentences in a natural or artificial language", as Zadeh has put it. Take for example the concept "Height" which can be seen as a linguistic variable with values "very tall", "tall", "not tall", "average", "short", "very short", etc. To each of these values we may assign a membership function. Let the height range over a region [0, 230 cm] and assume that the linguistic terms are governed by a given set of rules. Then we define formally a linguistic variable.

DEFINITION 9. A linguistic variable is a 4-tuple (T, X, G, M) where

T is a set of natural language terms called *linguistic values*;
X is a universe of discourse;
G is a context free grammar used to generate elements of T; and
M is a mapping from T to the fuzzy subsets of X.

Example 5

In the example above
T = {very tall, tall, …}, X = [0, 230]
and M for tall:

$$\mu_{\text{tall}}(x) = \begin{cases} 0 & x \le 170; \\ \dfrac{x-170}{15} & 170 < x \le 185; \\ 1 & 185 < x. \end{cases}$$

Linguistic variables are fundamental when we want to represent knowledge in approximate reasoning. Often the meaning of a term needs to be modified. Examples of modifiers are the following:

$$\mu_{\text{VERY}} = [\mu_A(x)]^2$$
$$\mu_{\text{MORE OR LESS}} = \sqrt{\mu_A(x)}$$
$$\mu_{\text{INDEED}} = \begin{cases} 2[\mu_A(x)]^2 & 0 \le \mu_A(x) < 0.5; \\ 1 - 2[1-\mu_A(x)]^2 & 0.5 < \mu_A(x) \le 1. \end{cases}$$

Fuzzification

The fuzzification interface functions as follows:

a) Identifies and measures the input variables.
b) Performs a scale transformation of the physical domain into a normalized or *standard* universe of discourse. This transformation is not always necessary. There are, however, cases where the physical domain is inconvenient and a transformation facilitates the fuzzy operations significantly. The most common standard domains are [–6, 6], [0, 6], [–4, 4], [0, 4], [–1, 1], and [0, 1].
c) Fuzzifies the crisp input data, whether normalized or not. Fuzzification is a way of dealing with data which, subjective or objective, might be fraught with vagueness and imprecision. The fuzzifier transforms crisp data into suitable linguistic values, corresponding to fuzzy sets, so that these data become compatible with the fuzzy

antecedent-consequent mechanism. Thus, when the crisp value $x = 175$ for the height of an individual is fuzzified, we obtain the degrees to which 175 belongs to the fuzzy sets “very tall,” “tall,” and so on. For the membership function given in Example 5, we have $\mu_{\text{tall}}(175) = 0.333$.

Fuzzification is closely related to knowledge since the membership functions are the result of deep system knowledge, mathematical or experiential.

Fuzzy Reasoning

Height is observed and the conclusion “height is positive small” is derived. This conclusion may be formally written by choosing a symbol *h* for height and a symbol PS for “positive small”, “*h* is PS”. The proposition “*h* is PS” is called *atomic* and assumes a certain membership grade, say $\mu_{\text{PS}} = 0.4$. Atomic propositions together with *connectives* such as "and," "or," "not," or "if-then" form *compound* propositions. For example the expressions

if *X* is *A*, **then** *X* is *B*
X is *A* **or** *B*

and so on are compound propositions.

The connective AND corresponds to logical *conjunction* “X is $A \cap B$” where *A* and *B* are fuzzy sets and the appropriate membership function is $\mu_{A \cap B}$. Similarly "or" corresponds to *disjunction* “X is $A \cup B$” and $\mu_{A \cup B}$ and "not" corresponds to “X is ¯;A” and $\mu^{-}_{;A}$.

Now consider a sustainability model with two components, ECOS (ecosystem) and HUMS (human system). An experienced operator decides in terms of natural language “if ECOS is G (good) and HUMS is also G, then OSUS (overall sustainability) is G.” This statement can be written

if ECOS is G **and** HUMS is G, **then** OSUS is G.

This proposition has the form

if (antecedents) **then** (consequents)

and is called a *fuzzy conditional* or *fuzzy if-then* production rule.

KNOWLEDGE BASE

The knowledge base contains the knowledge related to a particular problem. It consists of a *data base* and a *rule base*.

1) Data Base

The data base provides information needed to devise linguistic rules and the fuzzification/defuzzification procedures. Thus the fundamental function of the data base is twofold:

Selection of membership functions to define the meaning of pertinent input/output variables. This selection may be straightforward or rather involved. We shall develop in detail all the ideas concerning membership functions as we proceed. Needless to state that engineering judgment and expert knowledge play an important role.

Definition of the physical and normalized domains which boils down to selecting proper normalization/denormalization coefficients.

The number of inputs and corresponding fuzzy sets define the size of the rule base and thus the dimension of the system. As we shall see, this dimension grows geometrically with the number of fuzzy sets. Therefore, the choice of membership functions should be as economical as possible. Such a choice, on the other hand, may provide for an inaccurate system model. The final number of fuzzy sets is a tradeoff between computational speed and accuracy which is the result of trial-and-error as well as experience.

2) Rule Base

The rule base summarizes the actions of an expert in the form

if (input variables) **then** (output evaluation)

In other words, a linguistic description based on expert knowledge provides the "best" policy. We then have an antecedent or "if" side which incorporates the linguistic values of the inputs, expressed as linguistic variables and a consequent or "then" side which includes the output evaluation also in a linguistic form.

Example 6

For a 3-input, 2-output system, the fuzzy assessment rules have the form:

Rule 1: **if** x_1 is $A_1;^{(1)}$ **and** x_2 is $A_2;^{(1)}$ **and** x_3 is $A_3;^{(1)}$, **then** u_1 is $B_1;^{(1)}$ **and** u_2 is $B_2;^{(1)}$
Rule 2: **if** x_1 is $A_1;^{(2)}$ **and** x_2 is $A_2;^{(2)}$ **and** x_3 is $A_3;^{(2)}$, **then** u_1 is $B_1;^{(2)}$ **and** u_2 is $B_2;^{(2)}$ $\vdots$
Rule n: **if** x_1 is $A_1;^{(n)}$ **and** x_2 is $A_2;^{(n)}$ **and** x_3 is $A_3;^{(n)}$, **then** u_1 is $B_1;^{(n)}$ **and** u_2 is $B_2;^{(n)}$.

The following are needed to construct a rule base:

- *Input/output variables*. A proper choice of input/output variables is crucial in the description and performance of the system. This choice determines the structure of the model and relies on experience and engineering knowledge. Typical inputs in sustainability assessment are values of indicators such as threatened species, concentrations of pollutants, students per teacher, gross domestic product, etc.

Typical output variables are status, pressure, and response indicators or ecological sustainability etc.

- *Range of linguistic values*. The choice of the linguistic values is closely tied to the choice of membership functions as we have already seen. Their range is a matter of achieving the best performance which here is [0, 1], since this range can easily be transformed to a percentage which is readily understood by decision makers.
- *Derivation of fuzzy rules*. A large amount of information and concomitant reasoning in everyday life is linguistic. In a sense we operate as fuzzy decision makers in an impressive number of ways: when we drive, open a faucet, tune in on a radio station, play soccer or tennis, give a shot, squeeze an orange and so on. Fuzzy rules, consciously or subconsciously, are ubiquitous and may enable experts express their knowledge in convenient ways. Loosely speaking we have the following ways of building a rule base:
- Experience and engineering knowledge: We have already spoken about daily tasks requiring experience. Such experience and engineering knowledge expressed linguistically are the core of a rule base. Devising a rule may be aided by properly constructed questionnaires directed to specialists or operators.
- Fuzzy model: The linguistic description of a system comprises a fuzzy model of the system. The rule base is then constructed from this model.
- Mathematical model: If a mathematical model of the system exists, it may be used to develop a fuzzy rule base and assessment algorithm.
- A number of issues arise in the context of building a rule base. These are:
- *Consistency*: A rule base should be designed in such a way that no contradictions ensue. A rule base is *consistent* if it contains no rules with the same antecedents and different consequents.

Take for example a rule base where the following rules are encountered:

if PRESSURE is WEAK **and** STATUS is STRONG **and** RESPONSE is STRONG, **then** KNOW is VERY GOOD

if PRESSURE is WEAK **and** STATUS is STRONG **and** RESPONSE IS STRONG, **then** KNOW is AVERAGE.

This rule base is inconsistent. It should be stressed that all the rule bases in this chapter are formulated so that no two rules have the same antecedents and thus they are consistent.

Completeness. A rule base is a matrix of linguistic values for an output given linguistic combinations of the inputs. There might be combinations of inputs which produce a null output. Then we say that the rule base is not complete. A rule base is *complete* if all combinations of inputs produce nonnull outputs. Incompleteness of rule bases is common since not all input combinations are of interest.

DEFUZZIFICATION

Fuzzy reasoning combines fuzzified inputs and the antecedents of each fuzzy rule to provide fuzzy consequences for the output variables. Defuzzification is the final operation assigning a numerical value to each output variable. The most popular defuzzification methods in the literature are center-of-gravity, bisector-of-area, and centroid (height) defuzzification. In this paper we use the latter because it has similar properties with the other two methods (see [8] for a comparison of various defuzzification methods), but it is simpler and, most importantly, it guarantees a monotonicity property which is desirable in measuring sustainability as we shall later see.

Consider a rule base with n rules and a single output u. Suppose rule i, which has the form

if x_1 is $A_1;^{(i)}$ **and** x_2 is $A_2;^{(i)}$ **and** …, **then** u is L_i,

assigns the linguistic value L_i to u with membership grade μ_i. Centroid defuzzification is done as follows. First, we choose a characteristic value y_i for each fuzzy set L_i. The simplest choice for y_i is the *peak value* of L_i, that is, the value u for which the membership function $\mu_{Li}(u)$ of L_i is maximized. Then, the crisp value of u is computed from

$$\text{defuzz}(u) = \frac{\sum_{i=1}^{n} y_i \mu_i}{\sum_{i=1}^{n} \mu_i}.$$

SUSTAINABILITY COMPONENTS AND INDICATORS

Sustainability assessments rely on indicators to evaluate human development and environmental conditions. An indicator provides information regarding the state or trend of some component of the human system or the ecosystem. Sustainability is an inherently vague concept whose scientific definition and measurement still lack wide acceptance. We have developed a model called SAFE (Sustainability Assessment by Fuzzy Evaluation) [9] which uses basic indicators of environmental integrity, economic efficiency, and social welfare as inputs and employs “if-then” linguistic rules and fuzzy logic reasoning to provide sustainability measures on the local, regional, or national levels. We have also developed an approach to sustainable decision-making [10] which uses sensitivity analysis of the sustainability measure with respect to basic indicators to reveal the most important factors contributing to a sustainable society.

Fuzzy logic was developed to handle complex and uncertain systems. Sustainability and sustainable development exhibit exactly these qualities, as different people have different ideas about these concepts. In this work we develop a *monotonic* SAFE model for the assessment of sustainability. Monotonicity ensures that the assessment is consistent in the sense that whenever a component of sustainability improves, the overall sustainability increases. About eighty basic indicators of environmental integrity and human welfare over the period from 1990 to 2005 are combined to provide a sustainability ranking of various countries together with the most critical indicators.

SAFE uses indicators of environmental integrity, economic efficiency, and social welfare as inputs and employs fuzzy logic reasoning to provide national sustainability measures. The overall sustainability (OSUS) of a country is assumed to be a function of two major components: ecological sustainability (ECOS) and human sustainability (HUMS). Each major component is decomposed into four composite variables, as shown in Fig. 1. These are: water quality, land integrity, air quality, and biodiversity for ECOS; and political aspects, economic welfare, health, and education for HUMS. Finally, each composite variable is evaluated by means of three types of indicator: Pressure, State, and Response indicator [11]. State is the present state of a component such as the size of forested land. Pressure is a force tending to change status such as the deforestation rate. Response is a reaction that brings Pressure to a level that will guarantee a better State as, for example, protecting a given area.

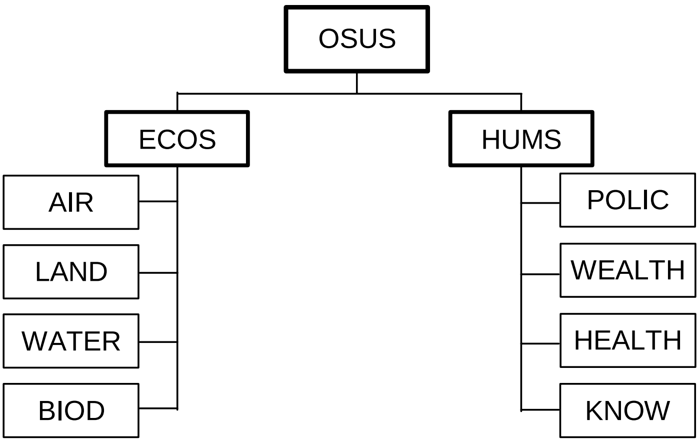

Figure 1. Dependencies of Sustainability Components.

SAFE provides flexibility in the choice of indicators and, although the structure shown in Fig. 1 remains the same, indicators can be changed if necessary, to capture the rapid changes in the global natural and socio-economical environments. The basic indicators we use in this chapter are shown in Table 1.

if DOMESTICATED LAND is *medium* **and** CURRENT FOREST is *weak*, **then** STATUS(LAND) is *bad*;

if PRESSURE(LAND) is *average* **and** STATE(LAND) is *good* **and** RESPONSE(LAND) is *bad*, **then** LAND is *average*;

if LAND is *very bad* **or** WATER is *very bad* **or** BIOD is *very bad* **or** AIR is *very bad*, **then** ECOS is *very bad*;

if HUMS is *good* **and** ECOS is *bad*, **then** OSUS is *moderate*.

The original SAFE model uses the min-max inference method to compute the outputs of each knowledge base. Accordingly, "and" is expressed by the min-operator while "or" is expressed by the max-operator. For example, in the first rule given above, the truth value of the composite proposition "DOMESTICATED LAND is *medium* with grade 0.2 **and**

CURRENT FOREST is *weak* with grade 0.717" is the minimum of 0.2 and 0.7 and, therefore, its conclusion is "STATUS(LAND) is *bad* with grade 0.2."

In this chapter we use product-sum inference because this method ensures that whenever an indicator is improved, the overall sustainability increases, i.e., the model is monotonic (a proof of this result can be found in [12]). For the example given above, the inference engine yields "STATUS(LAND) is *bad* with grade 0.2 × 0.717 = 0.1434."

If several rules assign the same fuzzy set, say, L to a composite indicator, then the overall membership grade μ_L of this indicator to L is the sum of the membership grades. Finally, a crisp value for the composite indicator is computed using centroid defuzzification:

$$\text{composite indicator} = \frac{\sum_L y_L \mu_L}{\sum_L \mu_L},$$

where y_L is chosen as the peak value of the corresponding fuzzy set L. For example, from Fig. 3b we see that $y_{\mathrm{VB}} = 0$, $y_{\mathrm{B}} = 0.3$, $y_{\mathrm{A}} = 0.5$, etc.

Table 1. Basic Indicators Used in the SAFE Model

Secondary component	**PRESSURE**	**STATE**	**RESPONSE**
LAND	(1) Municipal waste per capita (2) Nuclear waste (3) Hazardous waste (4) Population density (5) Population growth rate (6) Desertification of land	(7) Domesticated land (8) Forest area	(9) Participation in international environmental agreements (10) Forest change (11) Protected area (12) Glass recycling (13) Paper recycling
WATER	(14) Water withdrawals per capita (15) Pesticide consumption (16) Fertilizer consumption	(17-19) Quality of water resources: BOD, phosphorus, metals	(20) Public wastewater treatment plants (9) Participation in international environmental agreements
BIOD	(21-26) Threatened bird, mammal, plant, fish, amphibian, and reptile species	(7) Forest area	(9) Participation in international environmental agreements (10) Forest change (11) Protected area

Table 1. continued

Secondary component	**PRESSURE**	**STATE**	**RESPONSE**
AIR	(27) Ozone depleting substances (28) Greenhouse gas emissions per capita (29) Passenger cars (30) Fossil fuel emissions per capita	(31) Mortality from respiratory diseases (32-34) Atmospheric concentrations of NO_2, SO_2, total suspended particulates	(35) Renewable energy production (% total energy consumption) (36) Railway passengers (9) Participation in international environmental agreements
POLIC	(37) Military spending (38) Refugees to total population for country of origin	(39) Political rights (40) Civil liberty (41) GINI index	(42) Official development assistance and aid (%GNI) (43) Environmental laws and enforcement (44) Tax revenue
WEALTH	(45) GDP implicit deflator (46) Imports (47) Unemployment	(48) Central external debt (49) GNI per capita (50) Standard & Poor's Investable index (51) Poverty	(52) Average annual GDP growth (53) Exports (54) Foreign direct investment
HEALTH	(55) Infant mortality rate (56) Maternal mortality rate (57) AIDS HIV prevalence (58) Tuberculosis prevalence per 100.000 population (59) Number of confirmed polio cases	(60) Life expectancy (61, 62) Percent of one-year-old infants immunized against severe diseases: measles, DPT (63) Daily per capita calorie supply	(64) Number of doctors per 1000 persons (65) Hospital beds per 100 persons (66) Public health expenditure (67) Access to improved water sources (68) Access to improved sanitation
KNOW	(69) Patent applications by non residents (70-72) Ratio of students to teaching staff (primary, secondary, and tertiary education)	(73, 74) Expected years of schooling: male and female (75, 76) Net school enrollment: primary and secondary education (77) Illiteracy rate	(78) Public expenditure on R&D (79) Public expenditure on education (80) Patent applications by residents (81) Personal computers (82) Internet users (83) Expenditure on information and communication technology

SENSITIVITY ANALYSIS

Sustainable decision-making entails the choice of appropriate tools, such as, policies, actions, technologies, and resources to be used in order to improve sustainability. Due to restrictions in time, money, and natural and human resources, decision makers are often confined to just a small subset of indicators to be improved. The critical subset of indicators is the one that will maximize the overall sustainability.

A simple way to identify the major factors influencing sustainability is to rank the basic indicators according to the size of the gradient of OSUS with respect to each indicator. Indicators having large gradients are the most important ones when decision makers formulate policies for sustainable development. Gradient information can be obtained as follows.

(1) *Calculation of OSUS:* Compute the membership grades of composite indicators to the corresponding fuzzy sets. Start from the inference engines that use only basic indicators as inputs and proceed successively to the ones that use more composite indicators. Having computed the membership grades of OSUS, compute a crisp value for OSUS.

(2) *Introduction of perturbation:* For some basic indicator $x \in [0, 1]$, increase its normalized value by some fixed amount δ (for example, 0.1 or 10%). If the result is greater than one, then truncate it to one to avoid overshooting permissible regions of indicators.

(3) *Sensitivity analysis:* Assess the overall sustainability using the same set of data as in step (1) except for indicator x whose value is now $x + \delta$. Denote the new assessment OSUS($x + \delta$). The gradient of OSUS with respect to x is defined by the difference Δ_x = OSUS($x + \delta$) – OSUS.

Reset the basic indicator to its original value x.

Steps (2) and (3) are executed repeatedly for all basic indicators. By changing several indicators simultaneously in step (2) we can compute gradients of higher orders. For example, the second-order gradient of OSUS with respect to indicators x and y is $\Delta_{x,y}$ = OSUS($x + \delta, y + \delta$) – OSUS.

Sensitivity analysis reveals the most important factors contributing to a sustainable society. This is a nontrivial task and a necessary step towards using the full decision-making potential of the model. There are indicators whose values are good but they tend towards deterioration. The sensitivity analysis spots such indicators and often provides counterintuitive results necessary to form the full picture of sustainability.

RESULTS

Sustainability assessments for 24 EU countries are shown in Table 2.

Table 2. Sustainability Ranking of EU Countries

Ranking	Country	OSUS	Ranking	Country	OSUS
1	FINLAND	0.800	13	BULGARIA	0.660
2	AUSTRIA	0.768	14	PORTUGAL	0.658
3	NETHERLANDS	0.700	15	HUNGARY	0.658
4	UNITED KINGDOM	0.693	16	POLAND	0.655
5	LITHUANIA	0.689	17	SLOVENIA	0.636
6	GERMANY	0.681	18	GREECE	0.600
7	DENMARK	0.680	19	SPAIN	0.600
8	FRANCE	0.676	20	CZECH REPUBLIC	0.600
9	SLOVAKIA	0.672	21	ESTONIA	0.593
10	SWEDEN	0.668	22	BELARUS	0.564
11	ITALY	0.666	23	LATVIA	0.559
12	IRELAND	0.665	24	ROMANIA	0.500

The results of sensitivity analysis of the SAFE model are shown in Table 3. Sensitivity analysis identifies the most important factors contributing to sustainable development and provides valuable results concerning the factors that affect sustainability. It turns out that the critical factors for 10 out of the 24 EU countries are exclusively environmental (forestry, air quality and biodiversity). For nine EU countries, sustainable development seems to depend mainly on economic and other human sustainability indicators.

A second-order sensitivity analysis of the SAFE model pinpoints important pairs of indicators. In some cases, these indicators cannot be identified by first-order analysis. For example, the most important pair of indicators for Finland is *Protected area* combined with *Patents by non residents*, although the former is not among the most important indicators shown in Table 3. For the United Kingdom, the results are even more remarkable: the most critical pair is *Forest area* combined with *Passenger cars*. None of these two indicators is among the most important ones according to first-order sensitivity analysis. The reason for this discrepancy between first and second-order gradients is that OSUS is not a linear function of its inputs. Thus, a simultaneous improvement of two basic inputs is not equivalent to the sum of the individual improvements. In general, second and higher-order sensitivity analysis can be used to formulate more comprehensive environmental policies.

Table 3. Most Important Basic Indicators (First-Order Sensitivity Analysis)

Country	Indicator; (± change needed)	Country	Indicator (± change needed)
AUSTRIA	Threatened mammals (–) Threatened birds (–) Threatened plant (–) Threatened fish (–)	ITALY	Exports (+) GDP implicit deflator (–) Unemployment (–)
BELGIUM	Immunization against measles (+) Passenger cars (–)	LATVIA	Participation in international environmental agreements (+) Number of doctors per 1000 people (+) Hospital beds (+)
BULGARIA	Railway passengers (+) Threatened fish (–)	LITHUANIA	Threatened mammals (–) Threatened birds (–) Threatened fish (–)
CZECH REPUBLIC	Threatened fish (–) Renewable energy supply (+)	NETHERLANDS	Forest area (+) Tax revenue (+)
DENMARK	Forest area (+) Patents by non residents (–) Expenditure on information and communication (+)	POLAND	Urban TSP concentration (–) Renewable energy supply (+)
ESTONIA	Participation in international environmental agreements (+) Poverty headcount ratio at $2 a day (PPP) (–)	PORTUGAL	Tax revenue (+) Military spending (–)
FINLAND	Patents by non residents (–) Public expenditure on R&D (+) Patents by residents (+)	ROMANIA	Imports (–) Unemployment (–)
FRANCE	Urban NO2 concentration (–) Renewable energy supply (+)	SLOVAKIA	Threatened fish (–) Threatened mammals (–)
GERMANY	Exports (+) Mortality from respiratory infections (–)	SLOVENIA	GNI per capita (+) Public expenditure on R&D (+)
GREECE	Protected area (+) Threatened fish (–)	SPAIN	Passenger cars (–) Metals concentration in water (–)
HUNGARY	Tax revenue (+) Military spending (–)	SWEDEN	Forest area (+) Passenger cars (–)
IRELAND	Tax revenue (+) GDP implicit deflator (–) Foreign direct investment (+)	UNITED KINGDOM	Exports (+) Patents by non residents (–)

Another observation is that, although some indicators seem to prevail, improving them does not have the same effect on the overall sustainability for each country. Usually, the less affected countries are those that have either very high or very low sustainability index, and this shows a kind of difficulty to "escape" from a stable situation. Despite the fact that many countries can be grouped together according to their potential improvement, the extent of this improvement varies. Thus, it becomes clear that there is no unique sustainable path and, accordingly, policy makers should choose different criteria and strategies to make efficient sustainable decisions for each country.

CONCLUSION

We have developed a model called SAFE for the measurement of sustainability, which encompasses two broad components, ecological sustainability and human sustainability. The SAFE approach provides new insights of sustainable development and it may serve as a practical tool for decision-making and policy design at the local or regional levels. Such approaches are urgently needed nowadays if we want to attack the problem of sustainable development systematically.

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REFERENCES

[1] PRB (Population Reference Bureau) (2006). World Population Data Sheet. *http://www.prb.org*.

[2] Prugh, T., Daly, H., & Costanza, R. (2000). *The Local Politics of Global Sustainability*. Washington, D.C.: Island Press.

[3] Power, T. M. (1996). *Lost Landscapes and Failed Economies: The Search for a Value of Place*. Washington, D.C.: Island Press.

[4] Costanza, R., d'Agre, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Ñame, S., O'Neil, R. V., Raruelo, J., Raskin, R. J., Sutton, P., & van den Belt, M. (1997). The value of the world's ecosystem services and natural capital. *Nature*, *vol. 387*, 253-260.

[5] Venetoulis, J., & Cobb, C. (2004). The Genuine Progress Indicator 1950–2002 (2004 Update), Redefining Progress. *http://www.rprogress.org*.

[6] Lélé, S. M. (1991). Sustainable development: a critical review. *World Development*, *vol. 19, no. 6*, 607-621.

[7] WCED (World Commission on Environment and Development) (1987). *Our Common Future*, Oxford: Oxford University Press.

[8] Driankov, D., Hellendoorn., H., & Reinfrank, M. (1996). *An Introduction to Fuzzy Control*. New York: Springer-Verlag,.

[9] Phillis, Y. A., & Andriantiatsaholiniaina, L. A. (2001). Sustainability: an ill-defined concept and its assessment using fuzzy logic. *Ecological Economics*, *vol. 37*, 435-456.

[10] Andriantiatsaholiniaina, L. A., Kouikoglou, V. S., & Phillis, Y. A. (2004). Evaluating strategies for sustainable development: fuzzy logic reasoning and sensitivity analysis. *Ecological Economics*, *vol. 48*, 149-172.

[11] OECD (Organization for Economic Co-operation and Development) (1991). *Environmental Indicators, a Preliminary Set*. Paris: OECD.

[12] Kouikoglou, V. S., & Phillis, Y. A., 2007. *A monotonic hierarchical fuzzy system with sum-product logic*. Working paper.

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Chapter 8

BEYOND MONETARY VALUATION: UNDERSTANDING THE SOCIAL PROPERTIES OF PROPERTY

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ABSTRACT

Over the last decade, Ecological Economics has provided key contributions to the human understanding of the importance and value of ecosystem services. While there has been much debate about the validity and limits of the various existing monetary valuation tools, in this chapter we actually wish to draw attention to the extension of the concept of valuation. The 'economistic' language of valuation has gained dominance in the policy domain, yet the literature on policy evaluation is more pluralistic, and tends to stress the importance of developing and using other quantitative and qualitative measures of value to complement monetary estimates of costs and benefits of policy interventions. Indeed 'beyond monetary valuation' lies a rich and partially unchartered territory. In this chapter we draw attention to the possibility to extend the notion of valuation, and its observation, beyond the individual and towards the social, cultural and institutional. We focus on the wider notion of property in relation to value and contrast the simple economic typology of property rights with the anthropological concept of property as social relations that govern the conduct of people with respect to the use of certain material objects. We argue that these recent anthropological observations, whilst mostly derived from studies of non-western groups, are (a) likely to be compatible with many observations of publics and stakeholders in developed countries and (b) can provide a framework to engage with values in human society in a more holistic way. We conclude that a more critical and sensitive look at enacted property relations can benefit ecological economics by complementing the insights gained through the use of more established environmental economics approaches and by potentially reaching beyond some of the limitations of these approaches to inform more contextualized and locally appropriate policy interventions.

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INTRODUCTION

It is now widely accepted that valuing natural resources only at their observable market value, is an undervaluation which often results in the unsustainable management, degradation and ultimately the destruction of these resources. This realisation has given impetus to the development of monetary valuation methods for non-market goods and services provided by these natural resources, so that a wider range of values (such as non-consumptive use values, existence values, option and bequest values) or even the aggregate 'total economic value' (TEV) of these resources can be estimated. Monetary valuation of natural resources clearly has certain limits and limitations, though where these boundaries are to be drawn is widely and often hotly debated by economists and non-economists alike. The debate is further complicated by the multidimensionality of the possible concerns about valuation, from strictly methodological, to ethical, epistemological or even to end-use orientation. This chapter will not seek to further the particular debate on *monetary* valuation. In stead, we open up an exploration of the broader meaning and relevance of the concept of valuation. We draw attention to the analytical potential of anthropological studies of the role of property in society; studies which can further ecological economics by revealing new insights into how, why and when different individuals and groups in society value natural resources or specific aspects of these.

The chapter is structured as follows. First we place the 'doing' of valuation studies within the political process of policy evaluation and ask why and when valuation may be deemed important. We then highlight the potentially heterogeneous and interdisciplinary nature of the concept of valuation as a tool to inform policy making and policy interventions, by exploring several aspects of valuation beyond the usual boundaries set by the academic discipline of economics. These concern the usefulness and feasibility of observing or eliciting preferences from non-individual actors and in non-monetary (including non-quantitative) terms. Looking at the concept of valuation beyond the limits of the monetary and the individual subsequently allows us to contrast the established economic typology of property rights with several anthropological typologies of property relations. We conclude that a stronger research focus on the social context of enacted property relations can yield new and crucial information for policy interventions by identifying socio-cultural barriers and opportunities for more sustainable natural resource management.

VALUATION AND EVALUATION

The funding and carrying out of monetary valuation studies may be motivated by a number of rationales. For non-profit organisations such state agencies and NGOs involved in conservation or management of natural resources, monetary valuation can be a very useful tool to underpin requests for the increase or continuation of state support or other forms of external funding. This is especially relevant for the governance culture that is currently dominant in most developed countries, which stresses the 'accountability' of 'evidence based policies' that are pursued through 'best value' approaches. The language of economics has thus grown more dominant in the policy debate, suggesting perhaps that so has the influence of economists. However the adoption of economistic language is no automatic proof of the adoption of economics as the disciplinary framework of policy analysis. In pluralist democratic societies, public policy making is often a 'communicative incremental' process that involves negotiations between different stakeholders and the various representative structures of public authority, resulting in a compromise between various actors' interests (Buttoud, 2000). Whilst it may appear that the dominance business and economics jargon is indicative of the strong influence neo-liberal economics has on governments and policy makers, many mainstream economists will readily admit that policy makers have often embraced their language much more readily than their actual economic advice. Economics could simply be seen as a currently popular language to justify policies, whereas the development and adoption of policies is a complex process that is far from transparent and (at best) only partially informed by economic analysis.

Monetary valuation can play both a strategic and an applied role in informing or justifying policies. Efforts to estimate the total economic value (TEV) may fall within the remit of the strategic. Estimates of TEV (e.g. Gren et al. 1995; Guo et al 2001; Costanza et al., 1998) have little direct or practical relevance apart from conveying the message that ecosystems are incredibly valuable and that their value has often been (vastly) underestimated by policy makers. These estimates can be very useful for stakeholders such as environmental and conservation GNOs and state agencies to underline the importance of their remit and 'make the business case' for a continuation or increase of state funding. In terms of actually informing policy interventions, TEV is of very little relevance. At this applied level, marginal values are much more important (e.g. Turner et al., 2002) and at this level monetary valuation is just one tool in the toolkit of project and policy evaluation.

POLICY EVALUATION; TOWARDS METHODOLOGICAL PLURALISM

The performance of any deliberate human activity can be judged on the basis of a number of different criteria. These include the effort it took, the extent to which its original aims have been achieved and the occurrence of unforeseen positive or negative side-effects. In that sense, a rigorous approach to inform policy would seek to identify and describe, and if possible also quantify, the effort (cost), extent (effectiveness), efficiency (effectiveness per unit cost) side-effects (other costs and benefits, and to whom they befall) and uncertainties or risks. These different measures are all important elements of policy evaluation. In this chapter we assume evaluation to include *ex ante* (often called Appraisal), intermediate or mid-term, and *ex post* (i.e. when a project or a policy scheme is completed). Evaluation is considered an essential part of modern public sector management practice (EC, 1997) as it increases the transparency and accountability of policy decisions, both of which are seen as key concepts of sound financial management (Levy, 1996). The UK government alone has issued a raft of different publications to promote better evaluation practices (DoE, 1991; HM Treasury 1988, 1995, 1997).

The primacy of the efficiency criterion in evaluation is evident from this grey literature: according to HMSO (1991), Cost Benefit Analysis (CBA) is the UK Government's preferred tool of project evaluation. However sometimes costs and/or benefits cannot be estimated in monetary units, and CBA can not be applied. In such a situation, two partial measures of efficiency may still be of use. If the policy has set a clear target (e.g. x ha of nature reserves created), the policy maker can carry out a Cost Effectiveness Analysis (CEA) in order to establish how that target can be (or has been) achieved at the lowest possible cost. When a target is not set and the aim of the policy is simply to maximise some social good, then a Value-for-Money (VFM) analysis can be used to calculate how much of this good can be (or has been) delivered with the given budget. These are thus partial efficiency measures which have to assume that either cost (VFM) or benefit (CEA) are fixed. However as stated earlier, (economic) efficiency is not the only criterion of successful policy intervention. HM Treasury (1997, p. 3) states that *"any substantial appraisal and evaluation will always benefit from a multi-disciplinary approach"*.

In the post Rio 1992 era, it is widely recognised that there is no single comprehensive tool to analyse the nature of the various trade-offs associated with the different choice options for sustainable natural resource management. In ecological economic text books, measures of efficiency can be found sitting next to a variety of other measures. For example Tacconi (2000, p. 80) states that "*…applications of CBA do not discuss the place of efficiency in the decision making process. Economics, and CBA certainly have a role in ecosystem distribution and allocation processes, but it is a complementary role rather than an overriding one, as is often argued in neo-classical economics"*. When it comes to the evaluation of environmental and conservation projects, the simultaneous use of different, complementing methods is advised (OECD, 1995). More well-known examples of formal methods include Acceptable Risk Analysis which is focused on uncertainty and risk, (Social) Impact Analysis which seeks to map any distributional effects, and Multi-Criteria Analysis which explores the trade-offs between different objectives. However many more evaluation tools have been used or can be envisaged. A broad range of deliberative methods have been used to explore issues around natural resource management (e.g. Tacconi, 2000; Stagl, 2006). The successful use of these methods hinges on their ability to elicit unimpeded responses and constructive discussions from and amongst participants. These methods may not always be rigidly structured as they must be adaptable to the situation. Experienced facilitators will tailor the format of the meetings to fit in with the individuals participating, the topics that are being discussed or the physical settings and communication tools that are available to support the meetings.

Environmental economists have developed a useful range of non-market valuation methods to assess how much money ecosystem services are worth to *individuals* (for a full overview of these methods, see text books such as Markandya and Richardson, 1992; Hanley et al., 1997; van Kooten and Bulte, 2000). What really sets deliberative methods aside from most traditional non-market valuation methods, is that values are debated and elicited in an interactive social setting. In short it puts the choice of the individual into social context. A next level up would be to explore the expressions of value through the lens of the *social* and the *institutional*. In the light of our interest in the wider applicability of the concept of valuation in policy evaluation, we explore in the coming section some of the perhaps more obvious extensions to economists' focus on the capturing of *individual* values and preferences.

OBSERVING VALUES AND PREFERENCES – BEYOND THE INDIVIDUAL

Preferences can be inferred from actual behaviour e.g. where people go in their spare time, what they spend time, money or physical effort on. This category is consistent with what economists call 'revealed preference': *"a revealed preference method would look at revealed behaviour and infer a valuation from such data through a pattern of actual use"* (Dore, 1999, p. 31). Because actual behaviour has cost implications for the actor, these types of data can be seen as relatively robust measures of the values of individuals, groups and institutions. When we look at revealed preference beyond monetisable individual behaviour, a wide range of indicators can be of use to aid our understanding how different actors in society may relate to the management of the natural environment. One example of such indicator is the construction of physical capital by actors. 'Physical capital' is one of the five types of capital recognised in the Sustainable Livelihoods approach adopted by DFID (1999). Within this category we could distinguish between *cultural* physical capital, e.g. monuments, locations of archeological importance, and *infrastructural* physical capital, e.g. footpaths, pick-nick spots, car parks etc. which are designed to encourage and enable physical access so that more people can benefit from specific ecosystem services. Infrastructure may also be built to protect the natural environment or the provision of ecosystem services (e.g. fencing to protect against overgrazing, wildlife viaducts to limit the disturbance caused by a new road etc.).

Whilst the preferences of an actor are often most reliably studied through the observation of the actual activities of that actor, it is also possible to note potential preferences by analyzing the views expressed by that actor. It is possible to elicit preferences through interviews, focus groups and surveys which ask people hypothetical questions about what they would like to see, do etc. This category is consistent with what economists call 'stated preference' data: this method generates a forward looking valuation based on people's expectations (Dore, 1999, p. 31). The preferences expressed by organisations can be analysed from their various communications (websites, annual reports, press releases etc.).

The distinction between preferences that are observed in behaviour and those that are expressed in communications is not always clear-cut. For example different actors may have different levels of ability and freedom to behave in full accordance to their preferences. Similarly, some forms of behaviour may be excellent indicators of preference but could be more difficult for an outsider to observe, e.g. because they are carried out in privacy. Different 'framings' can hamber interpretation. Preferences expressed in communications may be tangential to the purpose of the actual question so that it may not be clear to the observer to what extent the question actually triggered the expressing of preference, or merely provided the social context in which the individual or group that is being observed, saw it fit to express a strongly held preference. An example of the potentially blurred lines is that of designated values. These may include official state designations such as national parks, nature reserves, recreational areas etc.; the 'branding' of regions (e.g. for tourism); or the literary or historic 'readings' of the landscape at a more informal level (e.g. linked to individual and group identity). Some of these designations, notably national parks and other state designations, are actively managed. Such management regimes are expressed through activities that require human and financial resources and these activities can therefore be considered as observed preferences by state institutions. Literary or (art) historic readings of the landscape on the other hand are not always accompanied by such costly activities. Un der these circumstances the researcher wishing to capture preferences may have little more than spoken or written words, photographs or paintings to go by.

It is beyond the scope of this chapter to fully explore the possible differences between individual preferences and those of other actors (state institutions, NGOs, etc.), but it is useful to reflect on the possible nature of these differences. In a democratic and consensual society that enjoys the fullest active participation of its citizens, these differences can be expected to be much smaller than in societies under authoritarian rule. However in all cases, it can be questioned to what extent the power and influence wielded by non-individual actors is in balance with the number of individuals supporting these views. There is always a diversity of human values represented in society and the ruling party, coalition, class etc. can never be fully representative for the wider population. This takes us to the question of power. Structures of governance and formal law making tend to be shaped by the more powerful groups in society and these groups are inclined to use these structures and laws to protect or enhance their power and ownership rights. In general, economists view the pressure on biodiversity rich tropical ecosystems as often being related to the lack of clear or enforceable property rights (e.g. van Kooten and Bulte, 2000, p. 86). However clarity and enforceability of property rights are two very different concepts. Enforceability depends on the strength and willingness of the authorities, usually the state, to intervene. The notion of clarity, in this context, is more readily contested. It would be a dubious assumption to think that local groups and communities would lack an understanding of what their day-to-day relationship is with regards to land and resources. Sufficient consensus within local groups, communities and settlements yields locally accepted relationships with regards to natural resources. Individual and communal property rights are an integral part of such property relations. When local groups are politically weak, which is often the case with traditional communities in areas of high biodiversity value, their laws are unlikely to be codified in written legislation or endorsed by the state. 'Clarification' of the existing property relations of such subaltern groups is indeed an important task for academic researchers, and this is the focus for the following sections.

DEFINITIONS OF PROPERTY

Although the theoretical meaning of property has been researched at great length, there is still not a conclusive and universally accepted definition of what property is (Hann, 1998). One reason why this has proven so difficult to define, is that property rights are surrounded by rules to define and enforce them and by opposing ideologies to justify and legitimise them (Ingold *et al.,* 1997). Another problem is that property has been examined from different disciplinary perspectives. Von Benda-Beckmann *et al.* (2006) offer a good overview of four of these perspectives:

Political theorists like, for example, Locke, Rousseau, Engels and Marx have mainly examined the sources of legitimate property rights, emphasising in their analysis issues like the role of the state, social justice and equity, the relationship between power and property and the balance between the rights and freedom of individuals versus the needs of the collective of which the individuals are part.

Legal scholars have also focused on the distinction between private and public property, but have also addressed the question of what sort of social actors should hold property rights and the relationship that exists or should exist between the multiple holders of rights in a single good.

Anthropologists, on the other hand, have mainly dealt with the problems of comparison across different cultures and societies focusing, for example, on the role of kinship in property management, on situations of legal pluralism, and generally deconstructing many Western assumptions about property and its social meaning.

Finally, *economists*, have mainly focused on the theoretical significance of property in managing the social and economic effects of the utilization of scarce resources in order to satisfy human needs optimally. Most economists maintain the theoretical distinction between the following types of property: open access, common property, private property and state property.

Whilst there is thus little synthesis across the different disciplinary perspectives, Pipes (2000) argues that in general terms, property can be defined in two radically different ways. In the first definition, consistent with economics, property refers to the right of the owner(s) to exploit assets to the exclusion of everyone else and to dispose of the assets by sale or otherwise. In the second definition, property is defined not as a right over things, but as *relations* in respect to things. In this definition property rights refer not to the physical possession or the relation between the owner and a thing, but to the relationship between the owner and other individuals in reference to things. The second definition that describes property relations as social relations between people is almost like a textbook anthropological definition: *"The essential nature of property is to be found in social relations rather than in any inherent attributes of the thing or object that we call property. Property, in other words, is not a thing, but a network of social relations that governs the conduct of people with respect to the use and disposition of things"* (Hoebel, 1966: 424 in Hann, 1998: 4). However, as Hann cautions, the strict demarcation between things and social relations in the anthropological definition may be too restrictive. Therefore, he argues, property relations are social relations whereby *"[t]he word property is best seen as directing attention to a vast field of cultural as well as social relations, to the symbolic as well as material contexts within which things are recognised and personal as well as collective identities made"* (Hann, 1998: 5).

Without an indepth understanding of social property relations on the ground, the causes for unsustainable resource use cannot be fully grasped, and appropriate policy interventions that work with local communities cannot be developed. The overexploitation of renewable natural resources is often still explained as the 'tragedy of the commons' in reference to Hardin's eponymous book (1968), even though it is now understood in economics that this tragedy only applies to open access resources as opposed to common property. In many cases however, resources may be held in overlapping combinations of these property regimes (Feeney et al., 1996) and under pressure from developments in society, property regimes may shift. Upheaval may cause private, state and especially common property to shift into open access. This failure (or 'tragedy') of the community or state is thus the underlying cause of the problem. However in order to better understand the nature of these shifts in the countries and areas where these resources are being overexploited, it is important to know the property systems that have been in place before these shifts occurred. In many cases there were previously traditional property systems of local groups in place which had been developed over a long period to achieve a (relatively) sustained flow of ecosystem goods and services. Generally speaking, the most environmentally sustainable traditional cultures are those of hunter-gatherers. Except in circumstances of perturbation caused for example by technological innovation of hunting tools, or migration into areas not previously inhabited by humans, hunter-gatherers tend to live in a dynamic equilibrium with their environment, without utilising natural resources beyond the point of sustainable yield. The (semi-)nomadic nature of hunter-gatherer societies limits expressions of material culture and the accumulation of material wealth. The absence of accumulated wealth tends to go hand in hand with a more egalitarian society so that there may appear to be a relationship between 'people being equal' and 'people living in harmony with nature'. Yet it is not the limited level of private ownership over material objects, which explains the (relative) environmental sustainability of these societies. The misconception that hunter-gatherers are somehow more sustainable because they lack private property stems perhaps from the absence of codification of their (traditional) law through writing and from the cultural loss brought about by encroachment or conquest by more technologically advanced agricultural societies. Contrary to popular belief, i.e. the interpretations by people raised in these dominant societies, hunter gatherers do have an extensive property system, but it's focus is on duties as opposed to (exclusive) use rights. In the following sections, several recent studies of the property systems of hunter-gatherers will be used as exemplars to gain some insights into the characteristics of more sustainable property systems.

TYPES OF PROPERTY AMONGST HUNTER GATHERERS

In discussing property rights in hunter-gatherer societies, Barnard and Woodburn (1997) distinguish between five different property categories. Each of these categories will be briefly discussed, focussing specifically on the rights that are linked to each category and how these rights are organised.

The first category consists of the rights over land, water sources and ungarnered resources (like fixed assets, ritual sites, dwelling sites, hunting sites and so forth). Access to this type of property in immediate-return societies is, in principle, equal to all, but can become restricted when there is fierce competition over use of scarce resources. In general terms, people are naturally endowed with unconditioned rights of personal access to land and ungarnered resources. When access is sought to another person's area, permission must be sought but this is in most cases easily obtained. Although land is divided in separate territorial categories belonging (usually) to one extended family, the land cannot be alienated and it is rare to refuse access to the land to outsiders. Property rights in this first category should not be confused with exclusive possessive rights; instead, possession should be more defined as custodial rights in the sense of having a duty to look after the land and the (ungarnered) resources on behalf of the collectivity (Ingold, 1986). In delayed-return hunter-gatherer societies, on the other hand, access to land is more complicated and restricted because in these systems it is stressed that land and people are not separate entities.

The second type includes rights over movable property such as weapons, clothing, cooking pots, beads, and so forth. This kind of property is personally owned, but is constrained by custom. People make their own weapons and other tools, and property rights are allocated to these things on the basis that individuals are entitled to hold property over the things they have produced with their labour. Another important rule is that people are not allowed to accumulate movable property beyond what they need; anything they posses in excess must be shared with others - this is a moral obligation.

A third form of property rights applies to rights over food, such as meat, vegetables, seeds and nuts and other harvested food. In immediate return societies, food belongs initially to the person who has worked for it, but food must be shared when more is harvested than is needed for immediate consumption or when somebody else in the community is hungry. Because it is obligatory to share food, the person who provided it (e.g. a successful hunter) receives hardly any social recognition. Sharing the food and in particular large animals is not frequently practiced in delayed-return societies; they usually have methods to store the meat and the hunters are usually under less social pressure from the other community members to share the meat. When sharing takes place this is usually limited to people with whom there is a kinship relationship and tends to create reciprocal indebtedness. Furthermore, transactions in these societies, like sharing a large animal with kinship, are linked with gaining a higher status position.

The fourth category of property rights applies over certain capacities and functions of specific people (e.g. rights over hunting labour, sexual capacity, reproductive capacity, and so forth). Such rights are not common in immediate-return systems, where kinship is not a mechanism used to control rights over other peoples; for example, men do not have any special authority over their wives. In delayed-return societies, on the other hand, there are frequent examples where women are treated as jural minors which allows men to have certain rights over the women and the products of their labour.

The final form of property rights is the right over intangibles such as knowledge and intellectual property (dances, songs, sacred knowledge, medicinal knowledge, ritual designs etc.). Individual rights over songs and dances are very common in delayed-return societies. Sometimes such individual rights are held by a person on behalf of a clan or lineage and can only be transferred to another member of the same clan. However, there are also examples when individuals hold property rights over, for example, songs and rituals on their own behalf (see Morphy (1997) and Keen (1997) for examples of individually owned intellectual property rights and Simet (2000) for an example of group-based intellectual property rights). In immediate-return systems, on the other hand, knowledge is more often freely shared. Even though some people might possess special healing powers, these powers are also used for the benefit of the entire community. While knowledge can be individually owned, similar to the principles that apply to meat, the individual is obliged to utilise this knowledge for the shared benefit of the community.

In short, it can be argued that the concept of individual property rights over movable property (both tangible and intangible) does exist in (former) hunter-gatherer societies on the basis that labour is recognised as a principle that attributes property rights to individuals. However, other principles, like the obligations of sharing and gift-giving, override this basic one which makes the distinction between individual and community-based property rights very blurred. Both movable and immovable property can be individually owned (usually linked to a specific function) but possession is more defined as fulfilling a custodian role for the benefit of the community.

Although this very brief description of the property rights system in hunter-gatherer societies does not do justice to the diversity of different systems in different communities and the complexity (e.g. the range and depth of different property principles) of the principles that guide the development of property rights in any one society, the above illustrates that property in hunter-gatherer societies is based on a broad analytical concept that extends far beyond the dichotomy between individual and community-based property rights. The above examples show that people can have a bundle of rights with regard to different objects. This bundle of rights can vary according to the relationship between different property holders and different property objects and extends beyond having rights only; each member has individual rights and collective responsibilities and both are inextricably linked.

PROPERTY RIGHTS AND DUTIES IN WESTERN SOCIETIES

The understanding of traditional property systems is of obvious relevance for the development of community based natural resource management in areas inhabited by indigenous, minority or traditional groups. However the above exemplar may also be of relevance in more mainstream cultural settings. The critical legal scholar, Joseph Singer, has argued in *The Edges of the Field* (2000a) and *Entitlement* (2000b) that even Euro-American property law has never given owners rights without also giving them responsibilities. However, in what Singer (2000b) calls the *ownership model*, more emphasis is given to the rights owners have over the things they own than to the obligations or limits that come with the property rights of the owners. In other words, while at first sight both indigenous and Euro-American property models are based on the principles of rights and obligations, it can be argued that one way in which the indigenous property model distinguishes itself from the Euro-American one is in the stronger emphasis it places on obligations and duties rather than ownership rights. This difference can be partially explained by the fact that the property rights of indigenous peoples are usually user rights. This concept of *usufruct* implies automatically more a duty of care than ownership rights.

It can be argued that while formal legal codes such as individual and communal property rights may play an important role in Western societies, a broader analytical concept must be applied to understand the concept of property in other societies (Hann, 1998; Hirsch, 2002). One way of extending the scope of analysis is to include the institutional and cultural contexts within which property codes operate. Nevertheless, as argued earlier in this chapter, the dichotomy between individual and communal property rights has become the dominant framework to analyse property relations both in the context of the dominant Western liberal property model and in the context of ‘alternative’ property models. The discourse of both the proponents of the liberal paradigm as well as those who criticise the liberal property model is embedded in the dominant view of individual versus communal property. Exporting this rhetorical model to non-Western cultural settings becomes even more problematic when it is considered that the Western rhetorical property model has, even in the Western context, been questioned for its empirical validity. Hann (1998) and von Benda-Beckmann *et al.* (2006) argue that the Western property model idealises individual ownership and ignores different kinds of rights and obligations linked to different property objects and property holders. As a result, in the Western property model individual property holders have been given far stronger formal rights than they used to have in social and societal contexts where rights are more bounded by association with responsibilities and obligations. Nevertheless, the supremacy of individual ownership is considered to exemplify the ideal property situation to which all property systems, both Western and non-Western, should aspire (*ibid.*).

TOWARDS A 'BUNDLE OF RIGHTS' APROACH

Classifying property into the four categories of open access, private property, communal property and state property and specifically emphasising the dichotomy between individual and communal property are, according to von Benda-Beckmann and von Benda Beckmann (2006), the most misleading concepts in the interpretation of property systems. For a long time, communal property was interpreted as having very negative economic connotations; it was dismissed as backwards in terms of social and legal evolution and it was perceived as an obstacle to economic and commercial development. More recently, the merits of communal property have become more recognised again and it is now often seen as useful for the protection of natural resources and for sustainable resource management.

However, von Benda-Beckmann and von Benda Beckmann (2006) argue that, in general, most policies based on communal property rights have had limited success because they have, so far, not given appropriate attention to the nature and distribution of different types of property relations that connect actual property objects and holders. In other words, interpreting communal rights as a homogeneous category distorts the complexity of property relations between different groups and individuals. In 'reality' it is more likely that each category of tenure meets the needs of specific community members. As such 'community property' can be described as a landscape divided into areas of land used for various purposes and managed under different tenures. Each area represents a particular tenure niche, a space in which use is governed by a common set of rules. Different niches can be identified within a single area, ranging from open access (e.g. grazing areas), through common property (e.g. medicinal field plants) to individual property (small agricultural plots): *"each person in a community has rights of access to the land depending on the specific needs of the person at the time; for example, in any given community, a number of persons could each hold a right or a bundle of rights expressing a specific range of functions; a village could claim grazing rights over a parcel, subject to hunting rights of another, the transit rights of a third and cultivation rights of the fourth"* (Nzioki, 2002: 229). This example highlights that in order to identify value it is crucial to ask who uses what resources/land and on what terms.

When used appropriately, a bundle of rights approach can assist in highlighting that property relations are expressed in various ways in different societies and in different periods of history. Such relations are expressed at three levels that include: first, the social units (e.g. individuals, groups, lineages, corporations and states); second, the construction of valuables as property objects; and third, the different sets of rights and obligations social units have with respect to such objects.

In this respect a bundle of rights approach compares with the biographical story of things (Appadurai, 2005; Kopytoff, 2005), to argue that things and knowledge can have multiple cultural meanings and functions depending on time and place. Von Benda-Beckmann *et al.* use the metaphor of a bundle of rights to conceptualise: *"first […] the totality of property* *rights and duties as conceptualised in any one society and second, […] any specific form such as ownership, which by itself can be thought of as a bundle"* (2006: 15). They also use the concept to *"first […] characterise the specific rights bundled in one property object and second, to characterise the different kinds of property held by one society"* (2006: 15).

von Benda-Beckmann *et al.* (2006) also argue that in order to grasp the full meaning of property, it has to be analysed at four different levels or layers at which it may manifests itself – i.e. ideologies, legal systems, actual social relationships and social practices (see Figure 1). Property has a different meaning in each of these layers. In previous theoretical frameworks these layers were often reduced to one or two - viz. legal and ideological – while the other two layers – viz. the actual social relationships and social practices - were simply ignored or misinterpreted. Furthermore, they argue, it is also important to acknowledge that, within the legal and ideological framework, property can have different sources of legitimacy – i.e. local or traditional law, state law, international law and religious law. All these factors and layers must be taken into account when examining property relations from an empirical, descriptive and analytical approach.

All the layers, as identified above, interact and interrelate in various ways, but a broad distinction can be made between three general types of property relations. The first are what von Benda-Beckmann *et al.* (2006) call *concrete* property objects, relationships and rights (first and second layer in their analytical framework – see Figure 1) which occur when people use, transfer, inherit or dispute a relationship with a property object. Concretised property relations are expressed through actual social relationships between actual property-holders with respect to concrete valuables. The next type of property relations is that of *categorical* property objects, relationships and rights (third layer in Figure 1). These manifest themselves in laws and rights that are reproduced and changed and in which the nature of property law is explained, discussed or disputed in settings such as courts, parliaments, mass media, academia and local fora. In other words, categorical property relations are linked to a legal-institutional level and include categories of property relations that are constructed by specific property-holders, property objects, and the rights and obligations attached to these.

The last type of property relations, *ideological* property relations, falls according to von Benda-Beckmann *et al.* (2006), outside the remit of both categorical and concrete property objects and therefore must be treated as a separate phenomenon. Ideological property relations express themselves through general cultural ideals, ideologies and philosophies. Competing ideologies (e.g. capitalism versus communism or welfare state ideologies versus neoliberalism) differ in their representation and justification of both legal-institutional property relations (categorical) and existing social, quotidian (concrete) property relations.

A bundle of rights approach thus allows a detailed identification of who has what rights and under which terms. So far most property scholars have focused mainly on legal-institutional (categorical) and ideological property relations while ignoring actual, concretised property relations. This has often meant that only property relations that are formally organized through legal institutions have been recognised or noted in evaluations of natural resource management policies and property relations that exist on the ground through (changing) social practices have often been ignored in the process to inform policy interventions despite the fact that the latter might be more important to local people because it reflects much more accurately the actual situation in which they live their daily lives. However it is important that both categorical and concrete property relations are examined (von Benda-Beckmann *et al.*, 2006); while concretised property relations are to a large extent shaped by categorical property relations, they are indeed different social phenomena and are constrained and enabled by different social interactions (categorical e.g. by changes in land law; concrete e.g. by influx of new immigrants as a result of drought). While von Benda-Beckmann *et al.* (2006) acknowledge that property ideologies and legal rules are certainly important sources through which people can rationalise and contest their property relations, it is rather the social relationships and daily social practices that will have a greater bearing on people's dealing with property. For the evaluation of any policy on sustainable natural resources management, this means that rules that are embedded in a formal, institutionalized framework should never be taken at face value without a critical and in-depth examination of the actual practices and feelings of different sections of (different) local communities.

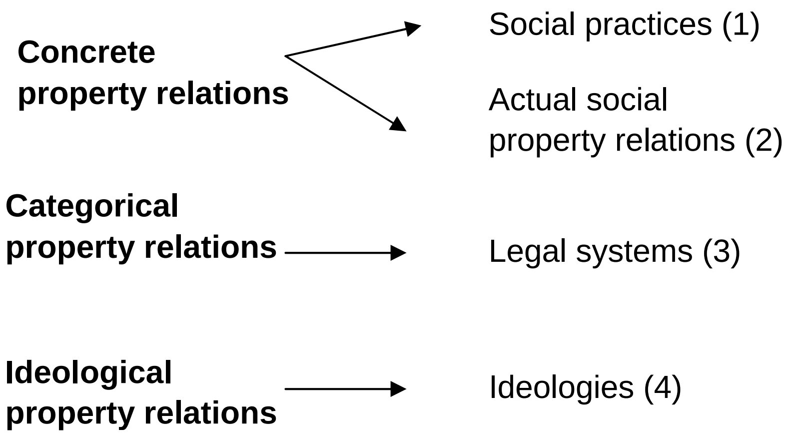

Figure 1: Four different layers at which property can manifest itself (adapted from von Benda-Beckman et al., 2006).

Nuijten and Lorenzo (2006) underwrite this more contextualised approach to examining property relations and argue that, indeed, many studies on property rights (such as studies on community-based resource management) assume that in most societies an institution exists that sets out in a formal way the rules with respect to property relations. However, they confirm the earlier findings in this chapter, that property relations are not so direct and linear and are often more complex; so they argue that in order to understand property relations on the ground, a more sophisticated framework of analysis is required. Nuijten (2005) acknowledges the fact that property rights are closely linked to other socio-economic and political conditions and that property relations are embedded in the wider field of social relations. More concretely, Nuijten suggests that examination of property relations on the ground requires mapping and analyzing of the power structures between different networks (e.g. different groups claiming rights over the same resource), the influence of formal and informal law and procedures, the role of formal and informal organisational structures and the role of various organisational structures and different positions of power on the ground between institutions and different groups of people. These suggestions can clearly be highly relevant for policy evaluation as it will yield insights that will allow the development of much more locally appropriate policy interventions.

Conclusion

This chapter has sought to widen the notion of valuation beyond the monetary and the individual, whilst retaining its relevance in informing policy interventions. A critical look at the context and stakeholders of valuation suggests that valuation as an activity can be characterised by different institutional arrangements and social framings. We have subsequently examined the values of nature through the lens of (social) property relations. This approach is clearly relevant in western settings, where notions of stewardship, duty of care or moral responsibility towards other people, other living organisms or future generations are widely felt and shared, even though at the formal legal level our notion of property does not extend much beyond a simple four-way classification of individual, communal, state or open access. The irony is then that we must draw on our (anthropological) observations of 'the other', such as hunter-gatherers and other indigenous peoples, to recognise the existence of more depth and complexity in property relations.

Whilst the dominant theoretical frameworks of property, notably exemplified by typology of property embraced by neo-liberal economics, have disembedded property relations from other social relations, it is shown in this chapter that the embeddedness of property in other socio-economic and political frameworks must be recognised in order to understand the full meaning of property relations in different (sub-)cultures. The dominant ideology in western society stresses the dichotomy between individual and communal property rights. In this chapter it is argued that this bifurcated view of property relations hampers the unpacking of more diverse and contextualized rules in existing (social) property relations. Therefore it is argued that new approaches to property theory, like the bundle of rights approach by von Benda-Beckmann *et al.* (2006), are needed and must be utilised in order to unravel property relations.

The analysis of formal law and the powers vested in state structures can give us a better understanding of what is valued and under-valued by the dominant stakeholders in society. The analysis of informal and customary law which is reflected in 'lived' property relations, can give us an understanding of values held by different less powerful groups in civil society, including minorities and subaltern groups. Some of these values can be much more conservation oriented than what is found in formal law, and may thus present opportunities for more stringent and pro-active policies on sustainable natural resource management. Other values may be less conservation oriented and may thus form a barrier to the successful implementation of even modestly ambitious policies for sustainable natural resource management.

Mapping of these values or informal laws can take place through a range of different and complementary methods which can support the analysis of observed preferences or stated preferences. Examples of the former include the observation of actual behaviour and the efforts of stakeholders (individuals, groups, corporations, etc.) to maintain the provision of ecosystem goods and services. The latter can be analysed by paying close attention to rethoric, story telling, popularity of certain media messages etc.

This research agenda is challenging because it is less likely to yield comfortable metrics for the current governance culture. However further examination of the commonalities and differences between typologies of economic values and layers or characteristics of different property rights, and between total economic value and bundle of rights, may yield new insights into the limitations of economic valuation approaches to sustainability, and provide important pointers for future developments in ecological economics.

References

Appadurai, A. (2005a) *The Social Life of Things: Commodities in Cultural Perpsective*. Cambridge: Cambridge University Press (first published in 1986).

Barnard, A. and Woodburn, J. (1997) 'Property, Power and Ideology in Hunter-Gathering Societies: an Introduction'. In: Ingold, T., Riches, D. and Woodburn, J. (eds.) *Hunter and Gatherers. Volume 2: Property, Power and Ideology* pp. 4-31. Oxford: Berg

Buttoud G. 2000. How can policy take into consideration the 'full value' of forests? Land Use Policy 17, 169-175.

Costanza R., d'Arge R., Groot R. de, Farber S., Grasso M., Hannon B., Limburg K., Naeem S., O'Neil R., Paruelo J., Raskin R.G., Sutton P., Belt M. van den 1998. The value of the world's ecosystem services and natural capital. Ecological Economics, 25(1), 3-15.

DFID, 1999. Sustainable livelihoods guidance sheets. Department for International Development, UK. http://www.livelihoods.org/info/guidance_sheets_pdfs/section2.pdf

DoE 1991. *Policy Appraisal and the Environment*. Department of the Environment, HMSO, London.

Dore M.H.I. 1999. The economics of well-being: a review of post welfarist economics. In: Dore M.H.I and Mount T.D. (eds) Global Environmental Economics. Blackwell, Oxford, UK.

EC 1997. *Evaluating EU expenditure programmes, A Guide, Ex post and Intermediate evaluation*. Office for Official Publications of the European Communities, Luxembourg.

Feeney D., Hanna S. and McEvoy A.F. 1996. questioning the assumptions behind the 'tragedy of the commons' model of fisheries. *Land Economics* 72, 187-205.

Gren I., Groth K. and Sylven M. 1995. Economic values of Danube Floodplain. Journal of Environmental Management, vol. 45, p. 333-345.

Guo Z., Xiao X., Gan Y. and Zheng Y. 2001. Ecosystem functions, services and their values – a case study in Xingshan County of China. Ecological Economics, 38, 141-154.

Hanley N., Shogren J.F. and White B. 1997. Environmental economics in theory and practice. Macmillan press, London.

Hann, C.M. (1998) 'Introduction: the Embeddedness of Property'. In: Hann, C.M. (ed.) *Property Relations. Renewing the Antrhopological Traditions* pp. 1-47. Cambridge: Cambridge University Press.

Hirsch, E. (2002) 'Malinowski's Intellectual Property'. *Anthropology Today* **18**, 1-2.

HM Treasury 1988. *Policy Evaluation: A guide for managers*. Her Majesty's Stationary Office, London.

HM Treasury 1995. *A Framework For the Evaluation Of Regeneration Projects And Programmes*. Her Majesty's Stationary Office, Norwich.

HM Treasury 1997. *Appraisal and evaluation in central government*. Second edition. Her Majesty's Stationary Office, Norwich.

Ingold, T. (1986) *The Appropriation of Nature: Essays on Human Ecology and Social Relations*. Manchester: Manchester University Press.

Ingold, T.; Riches, D. and Woodburn, J. (1997) *Hunters and Gatherers. Volume 2: Property, Power and Ideology*. Oxford: Berg.

Kopytoff, I. (2005) 'The Cultural Biography of Things: Commoditisation as Process'. In: Appadurai, A., (ed.) *The Social Life of Things: Commodities in Cultural Perspective* pp. 64-91. Cambridge: Cambidge University Press.

Keen, I. (1997) 'Yolngu Religious Property'. In: Ingold, T., Riches, D. and Woodburn, J. (eds.) *Hunters and Gatherers. Volume 2: Property, Power and Ideology* pp. 272-291. Oxford: Berg.

Levy R. 1996. Managing Value-for-Money Audit in the European Union: The Challenge of Diversity. *Journal of Common Market Studies*, 34(4), 507-529.

Markandya A. and Richardson J. (1992). Environmental Economics. Earthscan, London.

Morphy, H. (1997) 'Maintaining Cosmic Unity: ideology and the reproduction of Yolngu Clans'. In: Ingold, T.; Riches, D. and Woodburn, J. (eds.) *Hunters and Gatherers. Volume 2: Property, Power and Ideology* pp. 249-271.Oxford: Berg

Nuijten, M. (2005) 'Power in Practice: A Force Field Approach to Natural Resource Management'. *The Journal of Transdisciplinary Environmental Studies* **4**, 1-14.

Nuijten, M. and Lorenzo, D. (2006) 'Moving Borders and Invisible Boundaries'. In: von Benda Beckmann, F., von Benda-Beckmann, K. and Wiber, M. (eds.) *Changing properties of Property* pp. 218-242. New York: Berghahn Books.

Nzioki, A. (2002) 'The Effects of Land Tenure on Women's Access and Control of Land in Kenya'. In: An-Na'im Abdullahi A. (ed.) *Cultural Transformation and Human Rights in Africa* pp. 218-260. London: Zed Books Ltd.

OECD 1995. *The economic appraisal of environmental projects and policies. A practical guide*. Organisation for Economic Co-operation and Development, Paris.

Pipes, R. (2000) *Property and Freedom*. New York: Vintage Books.

Simet, J. L. (2000) 'Copyrighting Traditional Tolai Knowledge'. In: Whimp, K. and Busse, M. (eds.) *Protection of Intellectual, Biological and Cultural Property in Papua New Guinea* pp. 62-80. Canberra: Asia Pacific Press.

Singer, J.W. (2000a) *The Edges of the Field. Lessons on the Obligations of Ownership*. Boston: Beacon Press.

Singer, J.W. (2000b) *Entitlement. The Paradoxes of Property*. New Haven: Yale University Press

Stagl S., 2006. Multicriteria evaluation and public participation: the case of UK energy policy. *Land Use Policy* 23, 53-62.

Tacconi L. 2000. Biodiversity and Ecological Economics. Earthscan, London.

Turner R.K., Paavola J., Cooper P., Farber S., Jessamy V. and Georgiou S. 2002. Valuing nature: lessons learned and future research directions. CSERGE working paper EDM 02-05. University of East Anglia, UK.

van Kooten G.U. and Bulte E.H. 2000. The economics of nature. Blackwell, Oxford, UK

von Benda-Beckmann, F., von Benda-Beckmann, K. and Wiber, M. (2006) 'The Properties of Property'. In: von Benda-Beckmann, F., von Benda-Beckmann, K. and Wiber, M. (eds.) *Changing Properties of Property* pp. 1-39. New York: Berghahn Books.

von Benda-Beckmann, K. and von Benda Beckmann, F. (2006) 'How Communal is Communal and Whose Communal Is It? Lessons from Minangkabau'. In: von Benda Beckmann, F., von Benda-Beckmann, K. and Wiber, M. (eds.) *Changing Properties of Property* pp. 194-217. New York: Berghahn Books.

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Chapter 9

FORESTRY AND RURAL DEVELOPMENT: EXPLORING THE CONTEXT AS WELL AS THE PRODUCT

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ABSTRACT

The nature of rural development has undergone considerable change in the last 30 years. It is now recognised to include the construction of new networks, the combination of resources, and the renewed use of social, cultural and ecological capital. It often involves the reconfiguration of rural resources, many of which have previously been considered without value. Recent theories about rural development pay more attention to causes of local variations in development capacities and outcomes. Ecological economics as a discipline should attempt to equip itself with the approaches and tools to assess the complex and multi-faceted nature of rural development. This chapter discusses how various works in the field of Ecological Economics have started to build theories and approaches to do this. It goes on to review contributions from other disciplines, the Sustainable Livelihoods Approach and selected works in the fields of institutional and new institutional economics. Insight gained is used to inform the development of an approach to guide an appraisal of the rural development process and factors important in generating outcomes. The approach developed is used to assess several woodland related initiatives in Scotland. The assessment reveals the importance of factors such as culture, informal arrangements, payments in kind and networks in achieving environmental, economic and social outcomes and making the initiatives viable. It also brings to light constraints to development outcomes. It proposes that quantitative assessments of outcomes should be embedded in a broader analysis such as this in order to provide understanding and to inform policy. Also that further research is needed to test and refine tools and frameworks to guide the appraisal approach.

1. INTRODUCTION

As a multidisciplinary field of academic research that addresses the dynamic and spatial interdependence between human economies and natural ecosystems, ecological economics is fundamentally concerned with rural development. Ecological economics presents a pluralistic approach to the study of environmental issues and natural resource management and policy solutions. As a discipline, ecological economics needs to be equipped with approaches and tools to understand the multi-faceted and complex nature of rural development in the twenty first century.

1.1 Rural Development

In Britain the countryside has undergone considerable change since the 1980's, challenging the concept of rural development. The widespread adoption of neo-liberal approaches to macro-economic management during the 1980's encouraged a greater mobility of capital and the adoption of more diverse production methods (Lowe *et al*., 1993). Rural areas are no longer viewed primarily as the location of food production and new economic opportunities have opened up with rural areas increasingly seen as a collection of resources also available for amenity and leisure pursuits, provision of environmental features, forestry and industrial crops. There has been increased competition for rural resources from a variety of economic actors (Thomson, 1995; Lowe, 1993).

Puglieses (2000) sees sustainable rural development as combining the 1980's theories on sustainability with new strands of thought on rural development resulting from criticism of the modernisation of agriculture occurring during the twentieth century. Ray (1999) adds that sustainable rural development is widely thought to encompass the endogenous approach to socio-economic development, focusing on localities and their resources, and including the principles of participation, as a more effective means to robust development than its sectoral exogenous counterpart. At the same time, although the focus is increasingly local, this is combined with the extra-local in terms of resources, networks and partnerships (Lowe, 1996).

Van der Ploeg *et al*. (2000) ascertain that there is not yet a comprehensive definition of the new emerging type of rural development. 'It is about the construction of new networks, the revalorization and recombination of resources, the coordination and (re-)moulding of the social and the material, and the (re-newed) use of social, cultural and ecological capital. … . It involves the reconfiguration of rural resources, many of which have previously been considered without value.'

As recent theories about rural development pay more attention to causes of local variations in development capacities and outcomes, there is a need for analyses which reflect this. Analysis needs to look at a range of social and cultural factors which mediate development processes and provide a holistic understanding of the complex of interrelated processes which constitute the rural economy (Allanson; 1996; University of Aberdeen *et al*., 2002). In section 2 of this chapter we develop an appraisal approach to guide such an analysis.

1.2 Forestry and Rural Development

Changes in the rural economy and the new rural development agenda have resulted in new questions emerging about the relationship between forestry and rural development. These questions are complicated by the concern for both environment and rural development in current policy. Environment may be both a contributor to rural development through nature based tourism or other amenities or a constraint on rural development when, for example, species or habitat protection obligations curtail timber extraction possibilities. Current policy in Scotland (Scottish Executive, 2006) recognises the many roles that forestry has (such as supporting economic development, helping achieve rural diversification; securing community engagement) and the importance of maintaining a diverse resource to adapt to future uncertainties.

It is notable that forests appear to have been one of the main channels through which communities have expressed their desires for local control, amenity provision and other socio-economic benefits. This is particularly in evidence in Scotland. 'The forestry sector in Scotland is at the forefront of community engagement in land use' (Hodge and Maxwell, 2004). The first community-owned and managed woodland was established in 1987. By 2002, there were 64 operational and 19 planned community woodlands (MacIntyre and Marshall, 2003). According to the Community Woodland Association, 'there are now over 200 groups across Scotland, involved in or responsible for the management of thousands of hectares of woodland and open space' (http://www.community-woods.org.uk/).

In addition to the general emphasis on social objectives and participation embodied in sustainable development, the origins of the community woodland movement can be traced to two other areas. Firstly, to work in developing countries where, since the 1970's, there has been an emphasis on involving local people in management of forests in order to address problems of deforestation and livelihood needs of rural households.

Secondly, to land reform in Scotland. Land ownership in Scotland is characterised by a large portion of the country owned by very few people and a high proportion of absentee landowners. After long-standing unease about the iniquitous land rights patterns in Scotland, a Land Reform Policy Group was established in October 1997 and the resulting Land Reform Scotland Act (2003) introduced rights of responsible access and community rights to buy.

The potential of community woodlands to contribute to rural development is highlighted by MacIntyre and Marshall (2003): '*There are clear signs that it (the community woodland movement) could play a crucial role in helping reverse the economic and social decline prevalent across much of rural and post –industrial Scotland today. It is quite feasible that, within 10 years .., there could be several hundred community woodland groups .., working to develop their woodlands as a long term, renewable, local resource, using them as a central focus for community based social and educational activities, and as a catalyst for a range of new, locally based, diversified activities*.'(pg 5). A range of social benefits are claimed – development of social networks (Pickering, 2001; Slee *et al.*, 2004); cultural capital (Evans, 2001); acquiring and using local knowledge (Evans, 2001); social inclusion (ERM and Willis, 2004); identity and sense of belonging (Slee *et al.*, 2004; Burgess and O'Brien, 2001; Hunter *et al.*, 2001; Evans, 2001; Bejot-Seeboth, 2003).

Despite the claims above, there is a shortage of evidence on the benefits of community involvement and how they are delivered. This is highlighted in the new Scottish Forestry

Strategy (Scottish Executive, 2006a) which calls to 'improve the evidence base on ways to secure maximum benefit from woods in and around communities' (p. 36).

The remainder of this chapter is devoted to developing a framework through which to appraise rural development initiatives and, secondly, using the framework to guide an analysis of several community forestry initiatives in the Scottish Borders.

2. DEVELOPING AN APPRAISAL APPROACH

In order to develop an appraisal framework, literature is reviewed from the fields of development studies and (new) institutional economics. Insight from this review is combined with information from research in the forestry sector in Scotland.

2.1 Background

Drawing on selected works in the fields of institutional and new institutional economics and the Institutional Analysis and Development Approach, provides useful insight into factors important in the development process.

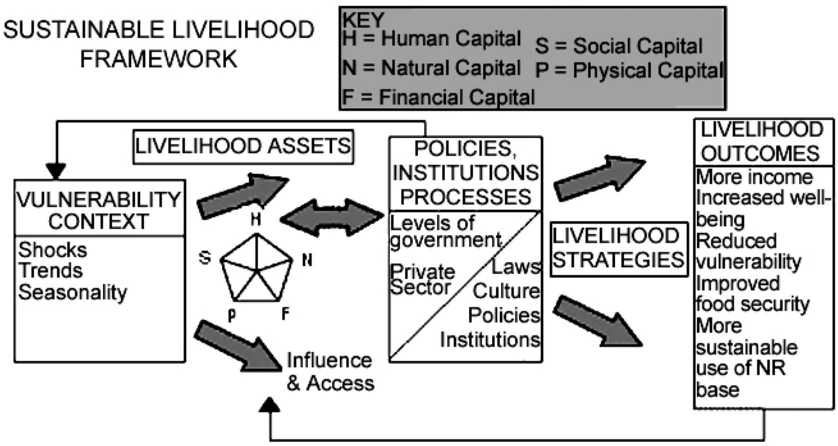
North (1990) develops an analytical framework for understanding the role of institutions in economic development. Institutions are defined to include any form of constraint that human beings devise to shape human interaction. They may be either formal, such as rules, or informal, such as conventions or norms of behaviour. They may be created or evolve over time and are the framework within which human interaction takes place. The major role of institutions in a society is to reduce uncertainty by establishing a stable structure to human interaction. North (1990) pays particular attention to the transaction costs involved in trade and the need for an institutional environment that reduces or minimises transaction costs. North identifies path dependency, the history of the development of an institution as being of importance and, in relation to this, power. In considering how institutions are formed or changed, Morrison *et al*. (2000) suggests we should look to how and by whom institutions are created. It is usually those in power who are responsible for creating institutions and changes tend to be incremental and influenced by existing or preceding institutions. The fact that institutions interrelate, that they are often embedded in 'higher level' institutions and that an analysis needs to consider the multiplicity of institutions and their interrelations is highlighted by Leach *et al*., (1999). See also Bingen, (2000) and Tompkins *et al*., (2002).

Morrison *et al*. (2000) highlights some of the contributions that New Institutional Economics (NIE) can make. He suggests how institutional arrangements can be viewed as supporting asset exchange between transacting parties and asset coordination between those holding or buying or selling similar assets. Under asset exchange, transactions range from 'spot transactions', the impersonal free market ideal, to non-market transactions, embedded in personalised, social or organisational relations. Asset coordination is beneficial where people obtain relatively small individual gains from holding, buying or selling assets, due to low unit value or small scale activity, and the associated transaction costs have a high fixed cost element incurred irrespective of the scale of the holding transaction. Savings can then be made in the cost of holding or exchanging an asset if the scale can be increased through coordination or consolidation. NIE emphasises the importance of access to assets which depend on institutional arrangements, information flows, asset characteristics and the vulnerability/power of different actors. The physical and economic characteristics of assets should not be examined without reference to the institutional arrangements which constrain or promote their use. Likewise the value of physical and natural assets can be better understood if an understanding of how different institutional arrangements can affect asset values to different users. Leach *et al*. (1999) make a similar distinction. They develop a framework which 'seeks to elucidate how ecological and social dynamics influence the natural-resource management activities of diverse groups of people, and how these activities in turn help to produce and to shape particular kinds of environment.' (Leach et al, 1999, p 226). They make a distinction between rights and resources and the ability to derive wellbeing from them. Analysis concentrates on the effect of micro, meso and macro level institutions on this process.

The Institutional Analysis and Development Approach (IAD) was developed by Elinor Ostrom and colleagues in the early 1980's as a multidisciplinary tool to frame policy research (Ostrom, 1999). Institutions are defined as 'the shared concepts used by humans in repetitive situations organized by rules, norms and strategies' (p 37). Rules are understood to mean shared prescriptions (must, must not or may) that are mutually understood and predictably enforced; norms are shared prescriptions that tend to be enforced by the participants themselves through internally and externally imposed costs and inducements, and strategies are the regularized plans that individuals make within the structure of incentives produced by rules, norms and expectations. The IAD elaborates on transaction costs. It breaks down transaction costs into information costs, coordination costs and strategic costs. Information costs occur as a result of searching for and organising information. They also include the cost of errors resulting from an ineffective blend of information. Coordination costs are the sum of the costs invested in negotiating, monitoring and enforcing agreements. Strategic costs result from asymmetries in information, power or other resources such that some obtain benefits at the expense of others (Imperial, 1999). Institutions are often established to reduce transaction costs and their ability to do this forms a useful focus of investigation (Morrison *et al.* 2000). The IAD approach draws attention to the contextual conditions, including physical and material conditions and attributes of community (culture), which affect how institutions are designed and operate.

The IAD framework also suggests examining overall performance using four criteria. It suggests that the effect of institutions on outcomes should include an evaluation of efficiency, equity, accountability and adaptability (Ostrom, 1998). Efficiency can be viewed from two perspectives. Firstly in terms of the market – what effect does the institutional arrangement have on wealth generation or productivity. Secondly, in terms of administrative efficiency and the costs of administering the regulatory framework. Equity also has two aspects. Firstly, in terms of 'fiscal equivalence' those that benefit from a service should bear the brunt of the associated costs. Secondly, redistributional equity concerns differential abilities to pay. Adaptability assesses the ability of institutional arrangements to adapt to changing environments and accountability refers to the need for sanctions or other mechanisms to hold organisations or individuals to account (Imperial, 1999).

The sustainable livelihoods approach is another useful tool to review. It originated in the work of Chambers and Conway - it is an integrating concept, incorporating social, economic and ecological dimensions (Chambers and Conway, 1992). The approach, or framework, is designed to assess situations and guide development interventions. It can be used at a variety of scales and levels, from local to national and community to policy. It was adopted by several development agencies in the 1990's, notably DFID's Natural Resources division, FAO, Oxfam, CARE and UNDP; and since then has been widely taken up by smaller development organisations. It 'provides a way to improve identification, appraisal, implementation and evaluation of development programmes' (DFID, 2000, p 5). While remaining flexible in its application it is centred on six core concepts aiming to be people centred; holistic; dynamic; to build on strengths; bridge the gap between macro and micro; and be sustainable.



Source: Sustainable Livelihoods Guidance sheet no. 1. (www.livelihoods.org/info/guidance_sheets)

Figure 1: The Sustainable Livelihoods Approach

Vulnerability Context

The 'vulnerability context' sets out the exogenous environment and trends and events over which people in the unit of analysis have little or no control. It includes trends (such as population, governance, economic or technological) shocks (such as health or economic) and seasonality (such as prices, employment, production) (DFID, 2000).

Livelihood Assets

Livelihood assets are one of the main building blocks available to the unit of analysis for development. The SLA divides assets into the following forms of capital.

Human capital: the skills, knowledge, ability to labour and good health. It is required to make use of the other forms of capital.

Social capital: in the context of the livelihood framework, social capital is taken to mean the 'social resources upon which people draw in pursuit of their livelihood objectives. These are developed through networks and connectedness..; membership of more formalised groups..; and relationships of trust, reciprocity and exchanges.' (DFID, 2000, p 10).

Natural capital: the natural resource stocks from which resource flows and services (e.g. nutrient cycling, erosion protection) are derived. There is a wide variation in the resources that make up natural capital, from intangible public goods such as the atmosphere and biodiversity to divisible assets used directly for production such as trees and land.

Physical capital: the basic infrastructure and producer goods needed to support activities, such as transport, buildings, water supply, energy, communications, tools and equipment.

Financial capital: the availability of financial resources. It includes flows as well as stocks.

Transforming Structures and Processes

The transforming structures and processes are the institutions, organisations, policies and legislation that shape livelihoods. They operate at all levels, from the household to the international. They effectively determine: access to capital, livelihood strategies and decision making bodies; the terms of exchange between types of capital and returns to any livelihood strategy (DFID, 2000). In a more recent DFID document, this area of the framework is described as embracing ‘a complex range of issues associated with participation, power, authority, governance, laws, policies, public service delivery, social relations (gender, ethnicity), institutions (laws, markets, land tenure arrangements) and organisations (NGOs, government agencies, private sector). It contains the macro-micro linkages and the relationships between the state, private sector, civil society and citizens’ (DFID, 2002).

Livelihood Strategies

These are the range and combination of activities and choices that people make/undertake in order to achieve their livelihood goals, such as production activities, investment strategies, reproductive choices (DFID, 2000).

In summary, the SLA framework shows people operating in a context of vulnerability, having access to various assets which gain their meaning and value through the prevailing social, institutional and organizational environment. These factors influence the strategies available to people to improve their livelihoods (DFID, 2000). See http://www.livelihoods.org for a comprehensive set of guidelines and more information on the livelihoods approach.

Although widely adopted, there have been criticisms of the SLA. The breadth of the framework and number of concepts that are encompassed by it make criticism easy. On the one hand, practitioners have expected it to explicitly include everything relevant in the development process, and it has been criticised for leaving out various concepts. DFID and FAO suggest that there should be more recognition of socio-economic, historical and cultural factors and Biswas, (2002) found that it paid inadequate attention to inequality. Bingen (2000) suggests that the framework broadens institutional analysis to include familial and community structures. Satchwell-Smith (2004) found that power is not adequately addressed. Also, the lack of attention to peoples’ rights has been a criticism and a fusion between the idea of rights and the SLA to produce a ‘livelihoods rights approach’ has been suggested (Moser and Norton, 2001 in Toner, 2002; Satchwell-Smith, N., 2004). On the other hand it has been criticised for being too broad and encompassing to be meaningful for understanding key components and processes in specific locations (Farington *et al*., 1999; Longley and Maxwell, 2003). It has also been criticised for not providing enough detail on how to use it: DFID and FAO (2000) find it offers no guidance on how to analyse and measure the capital assets or on linking micro-macro levels or policy analysis. Hobley (2000), Marzetti (2001) and DFID (2002) conclude that there needs to be considerable more work to determine the best way to analyse the ‘policies, institutions and processes’ part of the framework.

More generally, in a summary of experiences by DFID and FAO, it is suggested that that the approach is insufficiently flexible and that the overall concept is ethnocentric and not easily translatable (DFID/FAO, 2000). Beall (2002) argues that conceptualising people's assets as different forms of capital, reduces them to neo-classical economic concepts and tells us nothing of the relationship between assets or how they change over time. Kamuzora and Toner (2002) (in Toner, 2002) argue that participatory methods lead to an appraisal of all relevant elements of the SLA framework and, as such, the SLA is merely an extension of participatory methods.

Authors in field of Ecological Economics, such as Plummer and Armitage (2007) and Rudd (2004), have drawn on some of the above approaches and theories in order to develop their own approaches to appraise natural resource management. Plummer and Armitage (2007) argue for the need for contextually specific criteria to evaluate 'adaptive co-management' and propose three generic parameters for the components of ecological sustainability, livelihoods and process. Rudd (2004) proposes a modified IAD framework with an emphasis on multilevel causal linkages and flexibility for designing and monitoring ecosystem-based fisheries management policies. In this chapter, based on field work in the Scottish Borders, an approach is developed through which to analyse the process by which forestry initiatives contribute to rural development.

2.2 A New Appraisal Approach Developed

The diagram below represents an approach suggested as being useful to appraise rural development initiatives. The framework developed follows the SLA in its inclusion of external factors (vulnerability context in SLA), capital assets and governance/institutions. When considering governance and institutions, the importance of different levels and types and their interrelations and the history of how they have been formed are noted as points to explore. The framework shows governance and institutions interacting with capital, under the influence of external factors, to give outputs which are divided into four main types: creation of assets, access to assets, coordination of assets and asset exchange. Outputs have the potential to give rise to outcomes, economic, environmental and social and the framework points to this being an important stage to assess. Outcomes, in turn, often feed back into changed levels of capital. Evaluation occurs at the output and outcome levels, and focuses on the associated transaction costs, the adaptability of the institutions and institutional arrangements (or governance), whether equitable results are obtained, the efficiency with which outputs and outcomes are obtained and sustainability over time.

As with any such framework, its development was a 'balancing act', attempting to incorporate the important elements without making it too unwieldy. As it is, it would require a large study to incorporate an analysis of each element it covers, and this is not necessarily how it should be used. Rather, it should be used as a guide and tailored to different areas of research. It should remind the researcher of issues they should be aware of as bearing influence on the development process, even if they are not going to be fully incorporated into the study.

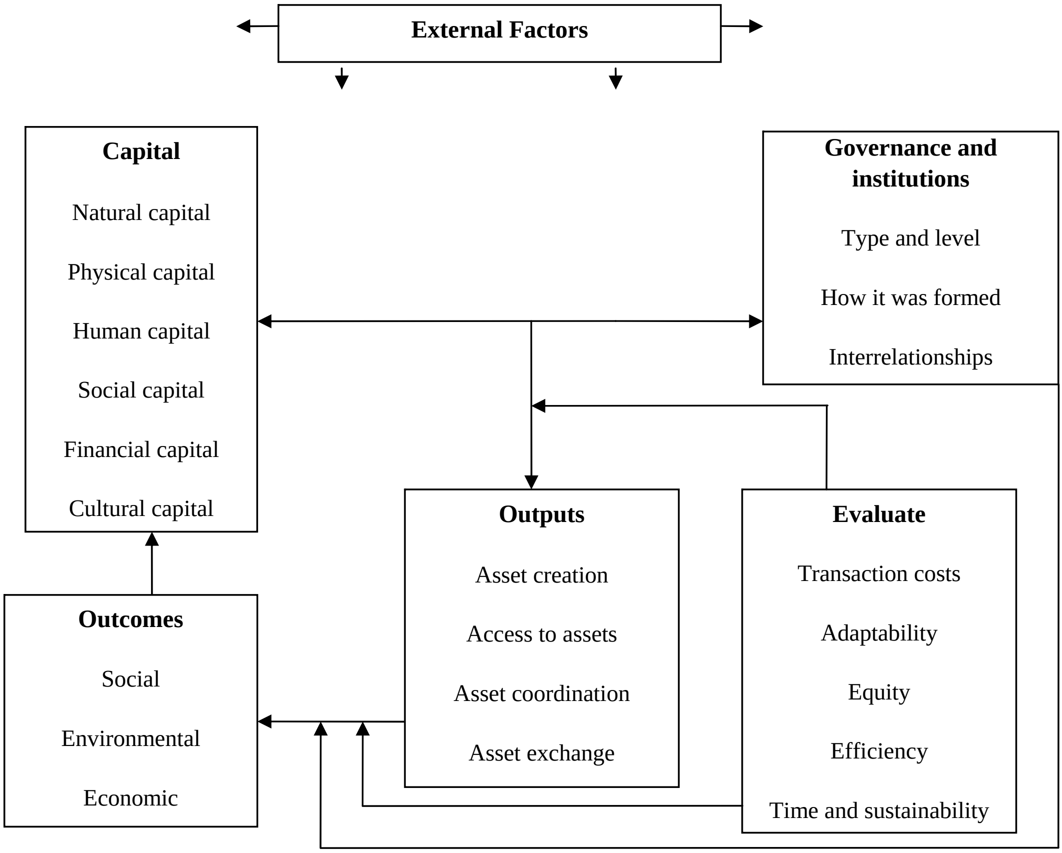


Fig. 2: An analysis framework

3. An Analytical Discussion

In this section we demonstrate the use of the framework through an analytical discussion of several forestry initiatives in the Scottish Borders.

3.1 External Factors

The potential of forestry to contribute to rural development was found to be affected by ‘external factors’ such as:

a)Global Market

The world market affects how attractive timber and other forest products are as an economic venture. The prices of timber and timber substitutes were mentioned by several interviewees as being important in management decisions and planned activities.

b)Agriculture

The recent and upcoming reforms to the Common Agricultural Policy (CAP) are having an effect on land use decisions. Recent reforms of the CAP have eroded subsidies related to production and are likely to have a marked effect on land management strategies. At present there is a degree of uncertainty over the effects and implications of the reforms which was said to be contributing to low levels of new planting. It is hoped that, in the future, forestry and agriculture will be more integrated and there will be increased planting by farmers.

c)Global Environmental Concern

Growing concern over environmental issues during the 1980’s culminated in the Rio Earth Summit in 1992. Resulting from that summit have been several international treaties/conventions and a general increased awareness of the wider (social and environmental) benefits of native species and forestry. This has lead to a favourable climate for planting conservation oriented and multi-purpose woods, resulting in funds being available for such activities. Several of the initiatives interviewed had benefited from such funds.

d)Access to Countryside

Several trends have contributed to a climate whereby there is a greater demand and expectation for access to assets.

e)Production Cycle

Inconsistency of quantities of timber coming into production creates difficulties for the industry. High levels of planting in the 1950’s and 60’s mean that production is going to increase by about 50% over the next 10 years in the Scottish Borders (Scottish Borders Council, 2005).

f)Work and Markets

Seasonality of markets, such as firewood, was mentioned by interviewees in relation to challenges for small enterprises in the Borders.

3.2 Levels of Capital

Levels of capital need to be considered both in terms of whether they exist and also whether they are available.

a)Natural Capital.

Land quality in the Borders is relatively good compared to the rest of Scotland and soils and climate offer some of the most favourable growing conditions for commercial conifer crops in Britain. It is also good agricultural land and incentives for planting are often not adequate to make forestry an attractive alternative to agriculture. Existing levels of woodland are low as are proportions of native woodland, offering plenty of potential for increased planting. Much of the coniferous resource is going to come ‘in to production’ (be ready for felling) in the next 10 – 15 years. This opens up considerable opportunities for restructuring and changing the species composition of these areas.

b)Human Capital.

Human capital was said to affect the ability to add value to products in the forestry sector in the Borders where there are relatively low levels of skills and knowledge relating to wood processing and products. The total population is increasing slightly, but young people tend to leave the Borders and in-migration tends to be by older people, generally retired. On the other hand, the relatively well educated population has been successful in managing projects and accessing funds for woodland initiatives.

c)Physical Capital.

Wood processing capacity is limited to small scale operations. Ninety percent of timber is processed in larger sawmills outwith the Scottish Borders. Of the existing sawmills, few have kilning capacity which limits their potential product range (and shortly, with new legislation coming in, will limit it further). Further education facilities offering training in woodland management or processing are absent, apart from at Woodschool which has very limited places. The road network is not suitable for transporting large volumes of timber.

d)Social Capital.

Social indicators suggest above average rates in terms of ‘community participation’. The inclination and ability of community groups to form and function requires a certain level of social capital and the community initiatives encountered suggest that some social capital exists. This is perhaps countered by the observation by one interviewee about the increasing new housing in the area for the ‘itinerate’ commuter population.

e)Financial Capital.

The initiatives and enterprises encountered all had to be resourceful in accessing financial capital to support the woodlands. However, funds exist and many initiatives had been very successful in fundraising. Also the ability to raise funds from the public (local communities) by some projects reflects a population that is able and willing to support local initiatives financially.

f)Cultural Capital.

Amongst the public, cultural capital in terms of a wood using, wood connecting culture was said to be low. The wood processing sector was said to be relatively old fashioned and lack a ‘business’ or ‘marketing’ culture or culture of innovation (with exceptions). It was mentioned that there is a lack of ‘wood culture’ whereby people fail to see the whole product or the value of the resource. Also a culture of ‘distrust’ in the supply chain was mentioned. The culture amongst the Forestry Commission and policy makers was said to have been a focus on supply with little attention to demand.

3.3 Governance and Institutions

This section discusses some of the governance arrangements and institutions encountered.

<table>
<tr><th colspan="6">Community woodlands</th></tr>
<tr><th>Glenkinnon</th><th>Janet’s Brae</th><th>Eshields</th><th>Darnick</th><th>Lindean</th><th>Gordon</th></tr>
<tr><td colspan="3" rowspan="2">Public estate. Joint management agreements between Borders Forest Trust (BFT) and Forestry Commission (FC).</td><td>Leased by BFT from council</td><td>Owned by BFT</td><td>Owned by community</td></tr>
<tr><td colspan="3">Community woodland associations</td></tr>
<tr><td colspan="6">Informal partnership between BFT and voluntary sector groups/special interest groups</td></tr>
<tr><td colspan="6">Use of payments in kind</td></tr>
<tr><td colspan="6">Borders Community Woodland Forum</td></tr>
</table>

It includes comments on the following points where appropriate:

1. The level and type of governance structure or institution, how it was formed and interrelationships.
2. How the governance structure or institution is used to create outputs:

   - Asset creation – planting new, or restructuring existing, woodland;
   - Access to assets – access to management/ownership rights and use of woodland for recreation, education or other amenity;
   - Asset coordination – coordination of assets to benefit from economies of scale;
   - Asset exchange – exchange of woodland products.

3. The effect of external factors and capital on its functionality.
4. Comments on transaction costs, adaptability, equity, efficiency and sustainability.

Community Management Through Leasing Arrangements and Joint Management Agreements

Community involvement and management as a mechanism for increasing access to assets is gaining in popularity. There are opportunities for communities to enjoy varying rights, from complete ownership to involvement in management. Only one of the community woodland groups interviewed owned their woods (said to be largely due to the legal responsibilities and level of administrative work that accompany ownership). In the Borders, the BFT often act as an intermediary body between the land owner and community. They were leaseholders for four of the community woodlands encountered, three of which were managed by a joint BFT/FC management board. These arrangements between organisations were found to be embedded in good relations, trust and respect for the partners. Origins of individual initiatives vary but amongst the initiatives interviewed all were instigated by BFT or the FC / BFT.

Opportunities for community involvement have tended to arise in woodlands which are not being primarily managed for timber production, or which are due to undergo restructuring or on other areas of unproductive land.

Community members have to know of the possibility of such an arrangement. In the Borders, the BFT who work closely with the FC and other land owners, are very proactive in seeking out such arrangements on behalf of communities. They significantly reduce information and coordination costs.

Such arrangements tend to be equitable in that they facilitate input from the community who also derive extra benefit. Adaptability depends on the nature of the arrangement between the land owner and community and any other agencies involved. Amongst the initiatives encountered, the arrangements were said to be relatively flexible. Administrative efficiency depends on the costs of administering the joint agreements. Joint management agreements run very smoothly in the Borders largely due to the presence of BFT. However, the presence of the BFT also adds to the public costs of these arrangements which in other parts of the country are held directly between the FC and community (voluntary) groups. The cost/benefit of BFT involvement and effects on administrative efficiency would need to be investigated. In terms of productivity, these arrangements add value to the woodland by increasing the amenity and social value, especially in cases, where the wood was previously unmanaged and/or inaccessible. However, there was also a case where some of the nearby population are not aware of the amenity that has been created for their benefit. Also, the reported degree of apathy regarding communities in part of the study area not seeking more involvement in management of local resources means that that the organisations involved have to ‘work hard’ for outcomes. There are claims that community management can have benefits for the local economy (especially in more remote areas) through stimulating interest in woodland products and entrepreneurial action. Little evidence of this was found amongst community woodlands in the Borders. On the contrary, one community woodland individual expressed concern that, though voluntarily carrying out planting and maintenance work in their woodland, they were not providing employment or supporting the local economy. However, it can be said that, although maybe some woodlands do not provide direct employment, in general community involvement leads to increased activity in the woodland with potential to have positive knock on benefits. BFT do make products from thinnings from the community woodlands, such as small household items, and kindling and firewood, demonstrating the opportunities for value added and economic activities. Some of the institutions to facilitate community involvement, such as the links between BFT and communities, are seen (by the agencies) to be short/medium term measures until the community takes over complete responsibility of the woodland. As such they are not meant to be sustainable in themselves, but to lead to a sustainable outcome of a community managed resource delivering a range of benefits. Other arrangements, such as communities leasing woodlands are intended to have the potential to be longer term, if demanded by the community.

Management by Bodies of Charitable Status

The community woodland association encountered were registered as charitable bodies. This formalises the public benefit nature of the initiatives at the community level and enables them to have formal members and apply for funds from a variety of sources, accessing financial capital usually for asset creation and asset exchange. The process of forming an association is also a step further in taking responsibility for the community woodlands and interviewees indicated that it resulted in benefits relating to empowerment.

A degree of commitment and agreement is required from a number of people in order to establish a charitable body. There need to be people willing to be trustees and office holders.

Social and human capital is advantageous. In one of the community woodland projects investigated, initial steps towards establishing an association had failed due to lack of agreement.

A condition of being registered as a charitable association is that the resource has to be managed for the public at large in theory resulting in equitable outcomes. In the community woodlands interviewed, the development of an association has lead to the group taking on more management, relying less on BFT. This makes the operation more efficient in terms of public cost. The community woodland associations were too young to comment on sustainability. The ability of one to raise money locally through small annual membership fee suggests that this initiative has prospects to be sustainable.

Informal Partnerships with other Civil Society Groups

Informal partnerships and links between the managing body and other civil society groups are used to add value to resources and make ventures viable. They result in asset creation and access to assets. BFT have a number of informal arrangements with groups such as Community Service, and Scottish Wildlife Trust. In general such groups come to the woodlands for particular activities. They use the woods for training or development purposes and in return assist in management. Other groups, such as museums or archaeological groups, come for educational trips. The FC are also keen to increase access for voluntary sector groups and work with the Scottish Association for Mental Health (SAMH), cubs, guides and others. Several of the community woodlands work with local schools. These arrangements are embedded in strong networks amongst voluntary sector organisations and establishing links.

Information costs are involved in making initial contacts and coordination costs in coordinating activities, but once links are established these costs are small. Such arrangements generally address redistributional equity, offering opportunities for access to disadvantaged groups. Fiscal equivalence is also addressed to a certain extent in that those carrying out the work do benefit, although their work is usually to improve facilities for other user groups and the wider public. Such arrangements appear to be very efficient as both parties to the partnership, through sharing their resources, generally achieve outcomes at very little cost.

Payments in Kind

BFT have a number of informal partnerships with local enterprises which exist on a basis of 50/50 (or similar) split of the product. These range from partnerships with firewood or charcoal enterprises to using storage facilities on neighbouring estates. They make asset exchange viable from the community woods.

4. THE SOCIAL BENEFITS OF COMMUNITY INVOLVEMENT

Taking the analysis one step further, the framework was used to explore the factors important in the process by which community involvement in forestry initiatives result in social benefit. Surveys were carried out amongst local communities which investigated whether and how the initiatives had led to social capital building, acquisition of knowledge and skills, social inclusion, cultural capital and an increased sense of belonging or connection to the area.

Social Capital and Knowledge and Skills

The study revealed that community involvement led to social capital building and acquiring knowledge and skills. These benefits were found to be associated with involvement in management groups or in the running of projects and taking part in work days and events. Existing levels of community cohesion was found to be important in determining whether communities became officially involved in management of the projects. Also human capital and networks were required for successful organisation of events. In our cases, where communities lacked capacity in such areas, BFT assisted in management and organisation of events which meant that benefits were still derived. As well as facilitating community involvement in management and the organisation of work days and events, the close involvement of agencies directly led to increase in trust in the organisations.

The projects had led a few people on to involvement in other initiatives. At the individual level, such links were often facilitated by agencies involved which also affected the nature of these links. At the longer established community woodland, with its own strong ethos and identity, links had tended to be formed in the areas being pursued by that project.

At the community level, there was evidence that the success of one of the cases, a recent community woodland project purchased and managed by the community, has increased community confidence and impetus and led to other projects in the village. At the district level, the involvement of BFT, a very active local agency, has facilitated links and networks at the district level to assist in furthering the community woodland movement in the Borders.

Acquisition of knowledge and skills was also found to be associated with visiting the wood. All the cases had good access. For larger and more complicated access constructions, out with the capabilities of the manpower of the community, grants had to be applied for and the community group (+ agency) needed the capacity (knowledge and time) to apply for grants and manage the construction. The availability and quality of alternative local countryside walks also affected the extent to which the community woodlands were visited. We also observed in one case that the local community were very inclined to take countryside walks and many had chosen to live in that area because of the options for such walks. Various ‘interest factors’ were also found to be important and the potential of woods to be a space in which people could develop and realise interests in aspects such as wildlife and local history, was found to be valuable.

Connection and Sense of Belonging

Analysis showed that most people felt differently about their area as a result of the woodland initiative, even if they had not been involved in it in any way. Findings suggested that the existence of a community wood where the local community is new and unformed had a significant impact on how people felt about their area, with many people feeling an increased connection or sense of belonging, even though there was no community group and the community wood was initiated and largely managed by an agency. In one case the community had purchased and was managing a wood in a village which lacks similar amenities. This was seen as a huge achievement and has changed the way that many people feel about their community with many people reporting increased pride, sense of community and that they felt that they had a stake in their surroundings.

Cultural Capital

The study suggested that being involved in a management group, very closely involved in the running of the project, is linked to change in values, as is going to events. Other forms of participation, less direct involvement in management or running of the initiative, taking part in work days and visiting the wood are not linked to change in values. The popular appeal of events, combined with their educational value, to people who may not otherwise visit the wood or have an existing ‘wood culture’ may explain why events are a better mechanism for cultural change than work days or some involvement in management. The initiative with a longstanding involvement from a group exhibited the highest rates of change in values and practices as would be expected.

Social Inclusion

To encourage public use was an objective of all the initiatives explored and all had made significant improvements to access to make visiting the wood easier and inviting. The fact that they appeal to all ages and people from all walks of life was commented on. Many people were first attracted to the woods for a specific event and have, since then, used it for walks. In this way, the community woods have encouraged people to use it who may not otherwise have been inclined to visit. Local schools were involved in all the initiatives. Other interest groups and voluntary groups also use the woods, either through BFT or, in the case of the long established community wood, through direct arrangements with the woodland group. BFT, with its overview of the portfolio of community woods in the Borders, are in a good position to match interest groups with appropriate sites and match the needs of specific woodlands with the skills or services of other voluntary groups.

Other Benefits

The framework reminds us that social benefits are one of three main benefit areas. It is important to situate the social benefits in the broader context of other benefits of the community involvement initiatives and explore any connections.

Environmental outcomes were a high priority of most of the community woodlands. Each of the initiatives appeared to be delivering environmental benefits, through management of the woods and educational events and activities. Economic activity was found to generally be of less interest amongst the community groups. One group does include it as an objective and sell woodland products and Christmas trees have been sold from another wood. Additionally, where work is contracted out, local contractors (where they exist) are favoured by all groups encountered and BFT.

There is visible evidence of the ‘overlapping’ nature of social, environmental and economic benefits. Examples of local socio-economic benefits being embedded in conservation outcomes were encountered. It was often people’s keen interest in the improved habitats and associated species in their woodlands which attracted them to become involved, participate in activities and visit the woods which, in turn, lead to social benefits. We can see a strong link between environmental and social benefits and it could be said that in these cases the social benefits experienced are embedded in the environmental / biodiversity related potential of woodlands and the popular appeal of wildlife.

5. CONCLUSION

Through using the appraisal approach developed, we took a wide and context encompassing appraisal of community forestry initiatives in the Scottish Borders. We appraised the governance structures and institutions which help overcome transaction costs and make the initiatives viable. We investigated the nature of the local capital stocks in terms of the opportunities and constraints they offer to the woodland initiatives. Existing social capital, natural capital and cultural capital were all found to be instrumental in determining levels of participation, use and social benefit derived. The approach was found to be very helpful as a guide to the appraisal, pointing to areas which should be considered, even if resource constraints mean that this can not be in detail, to provide a considered overview of the process involved in rural development. It is suggested that such an overview is necessary if analysis is going to attempt to understand the factors that are affecting development outcomes and make policy recommendations.

REFERENCES

Allanson, P. (1996). A sustainable rural economy? In The Rural Economy and the British Countryside. P. Allanson, and M. Whitby (Eds). London: Earthscan Publications.

Beall (2002). Living in the present, investing in the future: household security among the urban poor. In Urban Livelihoods: A People Centred Approach to Reducing Poverty. C. Rakodi and T. Lloyd-Jones (Eds). London: Earthscan.

Bejot-Seeboth, N. (2004). A Review of the First Six Years of the Wychwood Project. Oxon: The Wychwood Project.

Bingen, J. (2000). Institutions and Sustainable Livelihoods. Michigan State University.

Biwas, S. (2002). Does the Sustainable Livelihoods Framework Address Issues of Freedom, Choice, Capabilities of the Urban Poor: a Critique. MSc dissertation: London School of Economics and Political Science.

Burgess, J. and O'Brien, L. (2001). Trees, woods and forests: an exploration of personal and collective values. In Trees are Company. L. O'Brien and J. Claridge (Eds.). Social Science Research into Woodlands and the Natural Environment, Cardiff, Forest Research.

Chambers, R. and Conway, G. (1992) Sustainable Rural Livelihoods: Practical Concepts for the 21st century. IDS discussion paper no 296. Brighton: University of Sussex, Institute of Development Studies.

DFID (2000). Sustainable Livelihhoods Guidance Sheets. London: DFID.

DFID (2002). Sustainable Livelihhoods Guidance Sheets 5 - 7. London: DFID.

DFID/FAO (2000). Inter-agency Experience and Lessons from the Forum on Operationalising SLAs. London/Rome: DFID/FAO.

ERM and Willis, K. (2004). Making Woodland Count: its Contribution to our Quality of Life. Grantham Lincolnshire: Woodland Trust.

Evans, R. (2001). The virtuous forest: woodlands, community and identity in Scotland. In Trees are Company. L. O'Brien and J. Claridge (Eds.). Social Science Research into Woodlands and the Natural Environment, Cardiff, Forest Research.

Farrington, J., Carney, D. et al. (1999) Sustainable Livelihoods in Practice: Early Application of Concepts in Rural Areas. ODI Natural Resources Perspectives 42. London: ODI.

Hobley, M. (2001). Unpacking the PIP box. London: DFID.

Hodge, S. and Maxwell, C. (2004). Involving Communities in the Governance of Scotland's State Forest. Edinburgh: Forestry Commission Scotland.

Hunter, S., Pidgeon, N. *et al*. (2001) Forests, people and place: how individuals and communities perceive and relate to trees, woodland and forests in a Welsh context. In Trees are Company. L. O'Brien and J. Claridge (Eds.). Social Science Research into Woodlands and the Natural Environment, Cardiff, Forest Research.

Imperial, M. (1999). Institutional analysis and ecosystem-based management: the institutional analysis and development framework. Environmental Management 24 (4): 449 – 465.

Leach, M., Mearns, R. *et al*. (1999). Environmental entitlements: dynamics and institutions in community-based natural resource management. World Development 27 (2): 225 – 247.

Longley, C. and Maxwell, D. (2003). Livelihoods, Chronic Conflict and Humanitarian Response: A Review of Current Approaches. ODI Natural Resources Perspectives 89. London: ODI.

Lowe, P. (1996). Blueprint for a rural economy. In The Rural Economy and the British Countryside. P. Allanson and M. Whitby (Eds). London: Earthscan Publications.

Lowe, P., Murdoch, T. et al. (1993). Regulation of the new rural spaces: the uneven development of land. Journal of Rural Studies 9 (3):205 – 222.

MacIntyre, D. and Marshall, N. (2003). A Review of Community Woodlands in Scotland: Context, Status and Future Directions for the Community-led Woodland Movement. Edinburgh: Reforesting Scotland.

Marzetti, G. (2001) Brazilianising the SL Approach, Mimeo, October 2001. DFID, Brasilia**.**

Morrison, J., Dorwad, A., Kydd, J., Poulton, C., Smith, L. (2000). Sustainable livelihoods and new institutional economics. http://www.livelihoods.org/info/docs/nieweb2.doc

North, D. (1990). Institutions, Institutional Change and Economic Performance. Cambridge, New York: Cambridge University Press.

Ostrom, E. (1999). Institutional rational choice: an assessment of the institutiona analysis and development framework. In Theory of Policy Process. P. Sabatier (Eds). Oxford: Westview Press.

Van de Ploeg, J., Renting, H. *et al*. (2000). Rural development from practices and policies to theories. Sociologia Ruralis 40 (4): 391 – 409.

Pugliese, P. (2000). Organic farming and sustainable development: a multifaceted and promising convergence. Sociologia Ruralis 41 (21): 112 – 130.

Plummer, R. and Armitage, D. (2007). A resilience-based framework for evaluating adaptive co-management: linking ecology, economics and society in a complex world. Ecological Economics. 61 (2007): 62 – 74.

Ray, C. (1999). Endogenous development in t era of reflexive modernity. Journal of Rural Studies 15 (3): 257 – 267.

Rudd, M. (2004). An institutional framework for designing and monitoring ecosystem-based fisheries management and policy experiments. Ecological Economics 58 (2004): 109 – 124.

Satchwell-Smith, N. (2004). What are the potential synergies and disabling factors between the rights based approach and sustainable livelihoods approach to development. MPhil, IDS, University of Sussex.

Scottish Executive (2006). The Scottish Forestry Strategy. Edinburgh: The Scottish Executive.

Slee, B., Roberts, D. and Evans, R. (2004) Forestry and the rural economy: a new approach to assessing the impact of forestry on rural development. Forestry 77 (5): 441 – 453.

Tompkins, E., Adger, N. and Brown, K. (2002). Institutional networks for inclusive coastal management in Trinidad and Tobago. Environment and Planning 34: 1095 – 1111.

Thomson, J. (1995). Rural Futures: Issues for the forthcoming Rural White Paper. Exeter: University of Exeter Agricultural Economics Department.

Toner, A. (2002). Sustainable Livelihoods Approaches: Can they Transform Development? BCID Research Paper No 2. Bradford Centre for International Development.

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Chapter 10

NATURE RESERVE SELECTION FOR ENDANGERED SPECIES CONSIDERING HABITAT NEEDS: THE CASE OF THAILAND

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ABSTRACT

Creating or enlarging nature reserves to preserve key habitats and species living within those reserves is one of the important strategies to conserve biodiversity. This paper uses 0-1 programming models originating from the location science and termed SSCP (or Species Set Covering Problems), requiring representation of each and every species in the system within a minimum number of land parcels. The species under consideration in this study are the 68 mammals, reptiles and amphibians listed as threatened species in Thailand by the International Union for Conservation of Nature and Natural Reserves (IUCN), and the sites under consideration are known in Thailand as "amphoes", or small administrative districts. Since habitat requirements rarely have been introduced explicitly in reserve selection methods, this paper aims at identifying strategies to protect each threatened species while taking into account the habitat range of each particular terrestrial vertebrate of the data set. Results of the model are compared with the standard SSCP model and differences in outcomes are evaluated. Estimating the opportunity costs of converting countryside and forested areas for conservation purposes in terms of loss in economic output and incorporating them in the formulations further refines the model. Reserve networks that protect all threatened species and also that

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consider habitat needs are then selected at minimum opportunity costs. Results are compared with the former models to evaluate conservation policy options.

1. INTRODUCTION

The Convention on Biological Diversity, opened for signature at the United Nations Conference on Environment and Development, held in Rio de Janeiro in 1992, was ratified by a large number of countries, including Thailand. Its goal is to achieve a significant reduction in the current rate of biodiversity loss by 2010.

One of the most important strategies for conserving biodiversity is the creation or enlargement of nature reserves. Since Thailand ratified the Convention on Biological Diversity, it expanded, among other conservation actions, its national nature reserve system.

The aim of this study is to assemble a data set and build quantitative models to theoretically determine the minimum number of districts to be set aside as nature reserves or the minimal cost of the reserve so as to protect all threatened and endangered species. This is a theoretical exercise, applied on a real data set. Hence, it does not explicitly consider existing reserves in Thailand but proposes academic exercises in conservation policies.

The method of analysis for selecting nature reserves parallels that of location science aimed at selecting sites for locating economic activities (see ReVelle et al., 2002, for an extensive comparison between reserve selection and location covering models). The theoretical corpus used in this study is based on a location science problem termed LSCP, or Location Set Covering Problem. It was first proposed by Toregas et al. (1971) and Toregas and ReVelle (1973). The problem was to locate the least number of facilities (say, fire stations) so that all demand nodes (say, households) are covered by stations within a certain distance or time standard.

More than a decade later, Kirkpatrick (1983) and later Margules (1989) developed heuristic formulations, selecting sites by iterative methods, for protecting species (by attractivity of habitat for example). But it was only a decade again after the publication of Kirkpatrick's paper that Possingham et al. (1993) and Underhill (1994) independently recognized that the problem at stake was in fact a bioscience counterpart of the LSCP. This new problem of nature reserve selection, called SSCP, or Species Set Covering Problem, aims at selecting the smallest number of eligible land sites for a nature reserve so that all species are present in at least one selected site.

Over the years, models have been refined by including spatial characteristics (see Williams et al., 2005 for a review of such spatially-oriented models), probabilistic formulations (see for example, Polasky et al., 2001 and Haight et al., 2000), redundant coverage (see for example Malcolm and ReVelle, 2005 and Hamaide et al., 2006) or dynamic formulations (e.g. Costello and Polasky, 2004), among others. Very recently, Marianov et al. (2007) considered the importance of habitat needs in formulating a theoretical reserve selection problem.

This paper starts from the idea, like Marianov et al. (2007), that habitat range, differentially articulated for individual species, is important and should be taken into account when establishing a reserve network. New models are thus proposed here in which the aim is to identify methods that protect all threatened and endangered species either at minimum cost or with a minimal number of sites. They are applied to species presence-absence data constructed for the whole Kingdom of Thailand while requiring that all species enjoy at least their individual maximal home ranges.

The remainder of the paper is organized as follows. Section 2 details the theoretical models. The next section exercises the models on random data for determining their use and the potential differences between the proposed models. Section 4 reviews the data set for Thailand, that is, the presence-absence matrix, the list of eligible sites and the surface area of each site, the list of threatened and endangered species, the maximal individual home range for each of these species and the opportunity cost, for each district, if it is part of the reserve network. Section 5 presents the results of the models based on the elaborated data and compares the outputs of the models. The paper concludes with some summary observation and caveats in interpretation of the findings.

2. THE MODELS

The original SSCP requires representation of each species in at least one parcel in the system and seeks the minimum number of parcels in the reserve system to achieve this requirement.

Let there be n parcels of land eligible for selection and indexed j and represented by the set of sites J (J = {1, 2, .., n}) and m species that all need to be represented in the system, indexed i, and represented by the set of species I (I ={1, 2, .., m}). The problem is formulated as the following integer program:

$$\text{Min } Z_1 = \sum_{j=1}^{n} x_j \tag{1}$$

s.t.
$$\sum_{j=1}^{n} a_{ij} x_j \geq 1 \quad \forall i \in I \tag{2}$$

$$x_j \in \{0,1\},\, a_{ij} \in \{0,1\} \quad \forall j \in J,\, \forall i \in I \tag{3}$$

where $x_j = \begin{cases} 1 \text{ if site } j \text{ is selected for the reserve system} \\ 0 \text{ otherwise} \end{cases}$

and $a_{ij} = \begin{cases} 1 \text{ if site } j \text{ belongs to the set of sites that contains species } i \\ 0 \text{ otherwise} \end{cases}$

Objective (1) seeks to minimize the number of land parcels required for species protection. Constraint (2) requires that for each species i, the sum of parcels containing that species must be greater or equal to 1, that is, at least one parcel x_j where species i is present (a_{ij}=1) must be equal to 1, or equivalently, species i must be contained in the system.

The solution to this integer program gives a number Z_1 representing the number of sites necessary to represent all species. Replacing the objective by equation (4) and defining c_j as the opportunity cost of acquiring site j would result, after solving the integer program, in finding a number representing the minimum cost for preserving all species:

$$\text{Min} \sum_{j=1}^{n} c_j x_j \tag{4}$$

Among terrestrial vertebrate species represented in this study, some may be considered, in surface-area terms, as short range while others can be qualified as long range. This home range consists of a more or less restricted area within which an animal moves in the course of its daily activities (Harris et al., 1990). For example, a squirrel does not need a large territory to be able to sustain itself, find food and reproduce if it is living in a suitable habitat. On the contrary, carnivorous species such as tiger need a much larger area for its hunting, breeding and other daily activities.

As sites generally represent large patches of land, let us suppose that all species can sustain themselves in either one or two contiguous parcels; said differently, home-range can either be included in a single site or needs to spread over two sites. Let us further suppose that all sites have the same area. The second model can thus be written as such:

$$\text{Min } Z_2 = \sum_{j=1}^{n} x_j \tag{5}$$

$$\text{s.t.} \qquad \sum_{j=1}^{n} l_{ij} x_j \geq 1 \quad \forall i \in I_1 \tag{6}$$

$$\sum_{j=1}^{n} f_{ij} v_j \geq 2 \quad \forall i \in I_2 \tag{7}$$

$$v_j \leq \sum_{k \in H_j} f_{ik} x_k \quad \forall j \in J, \quad \forall i \in I_2 \tag{8}$$

$$v_j \leq x_j \quad \forall j \in J \tag{9}$$

$$x_j \in \{0,1\}, v_j \in \{0,1\}, l_{ij} \in \{0,1\}, \ f_{ij} \in \{0,1\} \quad \forall j \in J, \forall i \in I \tag{10}$$

where the set of new variables is defined as such:

$$l_{ij} = \begin{cases} 1 \text{ if site } j \text{ belongs to the set of sites that contains a short range species (species needing 1 site)} \\ 0 \text{ otherwise} \end{cases}$$

$$f_{ij} = \begin{cases} 1 \text{ if site j belongs to the set of sites that contains a long range species (species needing 2 sites)} \\ 0 \text{ otherwise} \end{cases}$$

$$v_j = \begin{cases} 1 \text{ if site } j \text{ is selected for the reserve system as one of the two adjacent sites in which } i \text{ breeds} \\ 0 \text{ otherwise} \end{cases}$$

H_j = set of adjacent sites to j and excluding j itself
I_1 = set of species whose home range requires one site, referred to as short range species
I_2 = set of species whose home range requires two sites, referred to as long range species

Objective (5) aims at finding the minimum number of sites while requiring, with equation (6), that, for each shorter range species i ($i \in I_1$), at least one parcel x_j where species i is present (l_{ij}=1) must be preserved, hence is greater or equal to 1. Constraint (7) requires that, for each longer-range species i ($i \in I_2$), at least two parcels v_j where species i is present (f_{ij}=1) must be part of the nature reserve as well. However, since these longer range species need two adjacent sites for being protected, constraint (8) ensures that, for each species i ($i \in I_2$), if v_j is selected (v_j=1), then at least one contiguous cell to j among which the species in question is present (f_{ik}=1 for species i) is also part of the reserve ($\sum_{k \in H_j} f_{ik} x_k \geq 1$). Additionally, constraint (9) makes sure that x_j is considered as part of the reserve once v_j is equal to 1 ($v_j \leq x_j$).

The solution to this integer program gives a number Z_2 representing the minimal quantity of sites that is necessary to protect all species taking home ranges into account.

In reality, sites rarely, if ever, have identical areas. Regular geometric areas can be specified when the species presence-absence data set is constructed for conservation purposes. A widely used example of such a data set is the Oregon terrestrial vertebrates data. Since the mid 1990’s, many selection models have been implemented with various versions of the Oregon data set (see for example, Csuti et al., 1997; Haight et al., 2000, Polasky et al., 2001, Arthur et al., 2004, Hamaide et al., 2008 and others). But when natural boundaries, like counties, provinces or districts are used to delineate sites, these will vary, often considerably, in surface area. Snyder et al. (2004) imposed limits on the total area of selected sites or minimized the surface of the reserved network while trying to protect as many species as possible.

When parcels do not have identical areas, it remains theoretically possible that a long-range species would need the equivalent of 4 or 5 sites if it were to select the smallest contiguous parcels. In order to avoid this more costly choice – it is often the case that when longer borders are included in a reserve network, the cost of managing the reserve is higher – the models described hereafter are formulated so as to choose the smallest number of parcels (for example two large sites for wide ranging species) and not the smallest surface (for example five small sites with the same surface as the previous two large sites). The third model outlined below does so while requiring that habitat range is met for all eligible species:

$$\text{Min } Z_3 = \sum_{j=1}^{n} x_j \tag{11}$$

$$\text{s.t.} \quad \sum_{j=1}^{n} s_j l_{ij} x_j \geq R_i \quad \forall i \in I_1 \tag{12}$$

$$\sum_{j=1}^{n} s_j f_{ij} v_j \geq R_i \quad \forall i \in I_2 \tag{13}$$

$$v_j \leq \sum_{k \in H_j} f_{ik} x_k \quad \forall j \in J, \quad \forall i \in I_2 \tag{14}$$

$$v_j \leq x_j \quad \forall j \in J \tag{15}$$

$$x_j \in \{0,1\}, v_j \in \{0,1\}, l_{ij} \in \{0,1\}, f_{ij} \in \{0,1\} \quad \forall j \in J, \forall i \in I \tag{16}$$

where the new variables are defined as such:

s_j = surface (in km^2) of each site j
R_i = home range (in km^2) for each species i

Taking the hypothesis that two contiguous sites of different areas can cover all longer range species[1], the problem is very similar to the previous one since it aims at minimizing the total number of sites (and not total area for preventing possible selection of many small sites) selected while taking habitat range into account. The only difference lies in equations (12) and (13). These equations stipulate that for each short and long-range species i, the surface of each parcel j selected, as part of the reserve network, must at least cover the species' home range (R_i). And for the longer-range species, when more than one site is selected, the model, via equations (14) and (15), ensures that selected sites are contiguous as site x_k is required to have a common border with with site v_j (since $k \in H_j$) and hence x_j (since $v_j \leq x_j$).

The outcome of the model is another integer Z_3 identifying the minimum number of sites in the reserve network with contiguous sites for longer-range species that need more than one parcel for performing their daily activities.

Another version of this third model would be to select those sites that are the cheapest (lowest opportunity cost) while respecting constraints (12) to (16). As stated above for the third model, objective (11) is then simply replaced by objective (4).

[1] The application on Thailand data, as detailed later in Sections 4 and 5, shows that all long range threatened or endangered terrestrial mammals may be preserved within two contiguous sites at the most, which is the reason why two adjacent sites are proposed here. The same analysis can neverthelelss be done with three or more contiguous sites. Equation (14) would then need to be modified accordingly. Note that it would even be possible to require that long range species be protected in one site only. However, as this would limit the site selection, mostly for the Asian elephant, this restrictive hypothesis is not considered

3. CLASSROOM EXAMPLE

Prior to the application of the models described above on a real data set, it may be interesting to test them on more limited, made-up data.

Suppose that an independent region can be divided up into 36 square sites (J=36). Suppose further that there are 20 species of interest (I=20). Presence and absence data of the 20 species in the 36 sites are generated randomly[2].

The models in this section and in Section 5 are coded with Mosel language and run on Xpress MP (Xpress MP, 2006).

The SSCP model is applied first (equations 1 to 3). One optimal scenario among the various alternate optima is illustrated in Figure 1 and shows a selection of 3 sites for protecting all species.

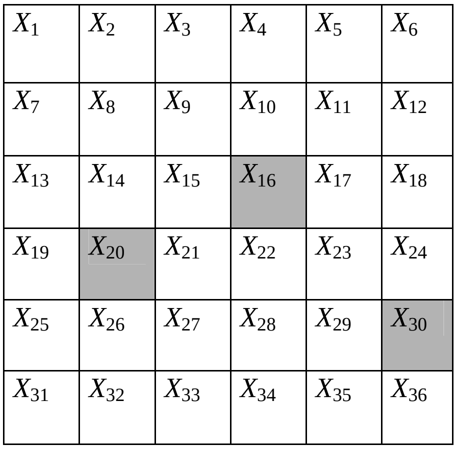

Figure 1: SSCP Model 1 – Output of the classroom example

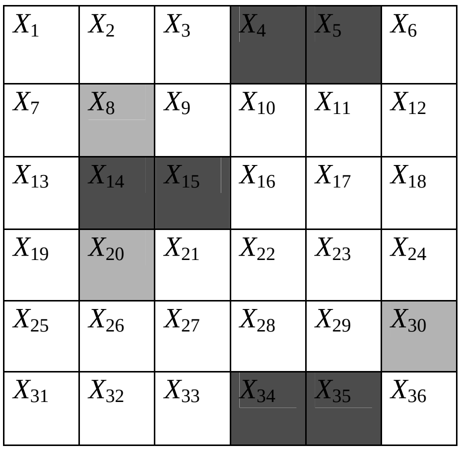

Figure 2: Model 2 – Output of the classroom example

[2] Data can be made available upon request

X_1	X_2	X_3	X_4	X_5	X_6
X_7	X_8	X_9	X_{10}	X_{11}	X_{12}
X_{13}	X_{14}	X_{15}	X_{16}	X_{17}	X_{18}
X_{19}	X_{20}	X_{21}	X_{22}	X_{23}	X_{24}
X_{25}	X_{26}	X_{27}	X_{28}	X_{29}	X_{30}
X_{31}	X_{32}	X_{33}	X_{34}	X_{35}	X_{36}

Figure 3: Model 3 – Output of the classroom example

The second model includes the importance of home range for selecting sites (equations 5 to 10). It takes the hypothesis that all sites have identical areas and that 3 of the 20 species considered are longer-range species (I_1=17 and I_2=3). Hence, they need at least two parcels of land for being considered as protected while the others are covered within one site. Nine sites are now selected. The contiguous sites aimed at protecting long-range species are depicted in dark grey while the sites protecting short-range species are in light grey (see Figure 2). This selection means that the three long-range species cannot be protected on similar sites (this is due to the random data set) and none of these contiguous sites are able to protect as many species as the three original sites from the SSCP. Hence, no economies in sites can be found and the model needs to choose six additional parcels, compared with the SSCP, for respecting the home range constraint of equations (7) to (9).

The last model lifts the strong hypothesis of identical site sizes. Suppose that the 12 northern sites have a surface of 2 km^2, the 12 central sites, of 4 km^2 and the remaining cells have a surface equivalent to 6 km^2. This is depicted in Figure 3. Running Model 3 (equations 11 to 16) with these additional data gives a minimum number of 10 sites. This is one more site than if all parcels are of equivalent size. Analyzing the results a bit more shows that two of the three long range species select the same sites as before (x_{14} and x_{15} for one x_{34} and x_{35} for the other) while protection of the last long range species is no longer possible on the previous sites (x_4 and x_5). The reason is that the combined surface of these two parcels is too small to be able to protect the species. Therefore, the model needs to select other contiguous sites, such as x_{24} and x_{30}, meeting the minimum surface (10) required for that long-range species. The two other long-range species keep the same sites as before as they respect their home range of 8 km^2 for the former and 12 km^2 for the latter.

The next section details data computation for the Kingdom of Thailand and the three models will then be re-run on those data.

4. DATA SET

4.1. Species, Sites and Presence-Absence Data

The 68 species included as candidates for the Thailand study, and detailed in Table I, are those 57 mammals, 9 reptile and 2 amphibian species that appears in the International Union for the Conservation of Nature and Natural Resources (IUCN) Red Lists for any year(s) from 1990 through 2006 (IUCN, 2006). Included are those taxa that are classified as any of the IUCN categories – rare, threatened and endangered. Some species about which the preponderance of evidence suggests that this animal has been extirpated in the wild in Thailand are excluded. Examples include Cervus eldi (Eld's deer) and Rhinoceros sondaicus (Javan rhinoceros). Also, note that all of these 68 species are considered to be terrestrial vertebrates. As a matter of fact, otters are considered terrestrial, although they occupy aquatic and shoreline habitat. Similarly, Macaca fascicularis (long-tailed macaque) is a mammal occupying a habitat such as lagoons, mangroves, etc. Cetaceans or marine mammals are likewise excluded.

Table I: IUCN Redlisted Species for Thailand

Scientific Name	Common Name	Taxon
Craseonycteris thonglongyai	Kitti's Hog-Nosed Bat	mammal
Macaca nemestrina	Pig-tailed Macaque	mammal
Macaca assamensis	Assamese Macaque	mammal
Macaca arctoides	Stump-tailed Macaque	mammal
Macaca mulatta	Rhesus Macaque	mammal
Macaca fascicularis	Long-tailed Macaque	mammal
Presbytis melalophus	Banded Langur	mammal
Presbytis obscura	Dusky Langur	mammal
Presbytis cristata	Silvered Langur	mammal
Presbytis phayrei	Phayre's Langur	mammal
Hylobates lar	White-handed Gibbon	mammal
Hylobates pileatus	Pileated Gibbon	mammal
Hylobates agilus	Agile Gibbon	mammal
Manis javanica	Malayan Pangolin	mammal
Manis pentadactyla	Chinese Pangolin	mammal
Ratufa affinus	Cream-colored giant squirrel	mammal
Ratufa bicolor	Black giant squirrel	mammal
Petuarista elegans	Lesser giant flying squirrel	mammal
Aeromys tephromelas	Large black flying squirrel	mammal
Petinomys setosus	White-bellied flying squirrel	mammal
Belomys pearsoni	Hairy-footed flying squirrel	mammal
Eothenomys melanogaster	Pere David's vole	mammal
Hapalomys longicaudatus	Marmoset rat	mammal
Rattus sillimensis remotus	Island rat	mammal
Rattus neilli	Neill's rat	mammal
Rattus hinpoon	Limestone rat	mammal
Canis aureus	Golden Jackal	mammal

Table I: IUCN Redlisted Species for Thailand (Continued)

Scientific Name	Common Name	Taxon
Cuon alpinus	Dhole	mammal
Ursus thibetanus	Asiatic black bear	mammal
Helarctos malayanus	Sun bear	mammal
Mustela strigidorsa	Black-striped weasel	mammal
Lutra lutra	Common Otter	mammal
Lutra (Lutrogale) perspicillata	Smooth-coated Otter	mammal
Lutra sumatrana	Hairy-nosed Otter	mammal
Amblonyx (Aonyx) cinera	Small-clawed otter	mammal
Prionodon pardicolor	Spotted linsang	mammal
Prionodon linsang	Banded linsang	mammal
Arctictus derbyanus	Binturong	mammal
Hemigalus derbyanus	Banded palm civet	mammal
Felis marmorata	Marbled cat	mammal
Prionailurus viverrinus	Fishing Cat	mammal
Felis bengalis	Leopard Cat	mammal
Felis chaus	Jungle Cat	mammal
Catopuma temmincki	Asian golden Cat	mammal
Neofelis nebulosa	Clouded Leopard	mammal
Panthera pardus	Leopard, Panther	mammal
Panthera tigris	Tiger	mammal
Elephas maximus	Asian Elephant	mammal
Tapirus indicus	Malayan Tapir	mammal
Dicerohinus sumatrensis	Sumatran Rhino	mammal
Muntiacus feae	Fea's Barking Deer	mammal
Cervus porcinus annamiticus	Indochina Hog Deer	mammal
Bubalus bubalis	Wild Water Buffalo	mammal
Bos javanicus	Banteng	mammal
Bos Gaurus	Guar	mammal
Capricornis sumatraensis	Serow	mammal
Naemorhedus goral	Goral	mammal
Tylototriton verrucosus	Crocodile salamander	amphibian
Rana fasciculispina	Spine-breasted giant frog	amphibian
Platysternon megecephalum	Chinese big-headed turtle	reptile
Batagur baska	River terrapin	reptile
Heosemys spinosa	Spiny terrapin	reptile
Testudo emys	Chinese land tortoise	reptile
Varanus bengalensis	Bengal monitor	reptile
Varanus rudicollis	Red-headed monitor	reptile
Varanus dumerilii	Black jungle monitor	reptile
Python molurus bivattatus	Burmese python	reptile
Python curtus	Blood python	reptile

Source: IUCN

Thailand, in the geographic configuration considered, is comprised of 76 provinces; each province is divided into "amphoes", the nation's smallest administrative districts. In this study, we consider 567 amphoes. All amphoes for the Bangkok metropolitan area have been excluded because of their highly urbanized and disturbed environments, as well as the presence of few of the species of interest there. Also, there have been recent boundary changes in the provinces in central and northeastern Thailand. Hence, for the purpose of our analysis, amphoes in that area have been classified back into their original set of amphoes so that all data remain comparable and at the same scale.

The species set for Thailand is elaborated in Sheerin (2007). The main compendium of recent information on endangered species is that of Humphrey and Bain (1990), which updates and extends Legakul and McNeely's seminal study (1977) on mammals in the country. Specific locational references to species occurrences are coded into a presence-absence matrix of all amphoes, followed by amphoe reference for species presence from distribution maps from agencies of the Royal Thai Government. Where multiple estimates of distribution are available, the more conservative of restrictive alternative is used. In the cases where more recent species-specific studies are available, that source is used, as in Sompoad and Varavudh (1995) for Bos javanicus (banteng) and Bos gaurus (guar). Also, constructed data can be crosschecked with the electronic Bioinventories of the World (2006), which applies to protected areas and can be readily mapped to amphoes. The correspondence between this source and the others (Humphrey and Bain, 1990 and Legakul and McNeely, 1977) is extremely close; in only one case is there a reference found in which the Bioinventories cited an occurrence in an amphoe which does not occur in the other sources.

With the passage of time since distribution estimates were made, and subsequent human development and encroachment, it seems clear that the ranges as coded to amphoes are best considered as historical ranges. This may be considered as a caveat in the analysis. However, these ranges, even if only accurate historically, offer conditions that provide suitable natural habitats, and have been used in this regard for species reintroduction. Hence, we feel that these presence-absence data may be helpful as a tool for conservation management in Thailand.

4.2. Species Frequency

After completing the elaboration of the presence-absence matrix, it is interesting to examine the difference in occurrences among all threatened and endangered species in Thailand. Indeed, a limited number of species is very frequent – communities of Malayan pangolin, smooth coated otter, leopard cat, clouded leopard and leopard were reported at some period in most of the parcels[3] – while others are very rare – the island rat, Pere David's vole, black-striped weasel and Bengal monitor were reported in two or three districts only. Three species are only present in one single amphoe: Neill's rat, limestone rat and Kitti's hog-nosed bat. Moreover, that last species is endemic to Thailand. The frequency distribution of species' areas of occupancy is shown in Figure 4.

[3] It is important to remember that the presence of these species in most of the parcels does not mean that these species are not in danger. Even though they are widely present, their numbers may be low and their reproductive habitat may be altered. So, these species remain at risk; they are either rare, threatened or endangered.

Church et al. (1996) and Storch and Sizling (2002) found out that, in general, most species are either relatively infrequent or very common in terms of the number of occupied sites, that is, the frequency distribution of species occurrences often is bimodal. It is confirmed by Hamaide et al. (2006) for the Oregon data set but it is not the case with the Thailand data set. This is not surprising because in this study, we have included only threatened and endangered species that is, the statistical tail of one mode of a general distribution of species occurrences by location. Therefore, it is expected that most of these species are present in a small number of parcels: about 45% of the species occupy less than 10% of the parcels of Thailand (the first bar of the graph) and about 75% of them (50 out of 68) are present in 30% of the parcels at the most (the first three bars of Figure 4). Said differently, many threatened species are present in few amphoes, as expected, and hence, quite a few sites need to be selected if all listed species are to be protected.

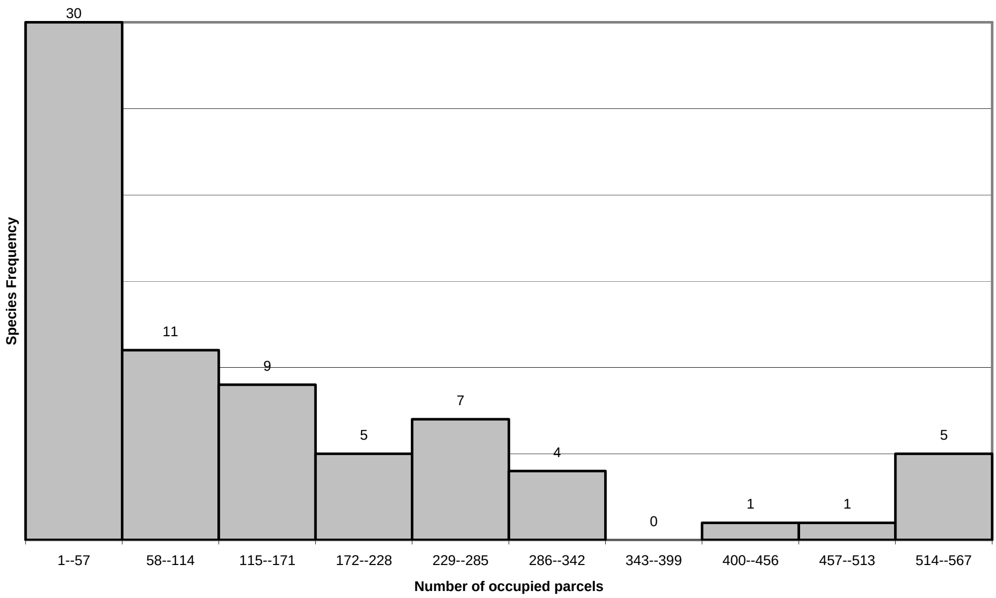

Figure 4: Frequency distribution of species' area of occupancy

4.3. Surface Data

The data on district surface area are collected from the websites of the Ministry of Interior, Royal Thai Government. The sites of the most relevance are (1) the listing by province (http://www.moi.go.th/province.htm#3) and (2) www.amphoe.com/menu.php. These websites allow the user to search for information for the districts in each province.

The current data source refers to districts as configured in 2007. In some cases, their geographic boundaries differ from boundaries used in assembling the species presence-absence data, whose amphoes are based on an earlier set of province and district boundaries. As mentioned above, this is the result of the creation of new districts from former sub-district area units. It is mainly relevant to provinces established in the recent periods, since the 1980s. In one case, the province of Sakaew was formerly part of the province of Prachinburi, and was created in the year 1993.

For the purpose of the paper, we have compiled the data in two steps. Firstly the current districts are listed by province, and the information on the surface area is obtained from the district information website (www.amphoe.com). The second step is to match these to the district list used for the species data. At this step, where it is found that the district names do not match, implying that new districts have been created for the province, we examine the "history" section of the district database, where usually there is information on how the new district has been formed from parts of a former district. Once this is done, we reclassify the new district back to the old district. The total number of such reclassification is less than 10% of the number of districts used in the analysis.

4.4. Home Ranges

The Thai Wildlife Research Division, Department of National Parks and Wildlife, in the Ministry of Natural resources and Environment of the Thai Government, have estimated home ranges for about 20 large mammals. These estimations come either from their own field research in Thailand or from other researchers' work such as Khan (1967) and Eisenberg (1997) who respectively studied the elephant and the tapir.

For the other species, home ranges are collected from existing literature. For example, Humphrey and Bain (1990) and Mason and McDonald (1986) provide estimates for various types of otters[4]. Habitat range for gibbons comes from Leighton (1987) and Rowe (1996). And Grassman et al. (2005), Grassman (2001), Odden et al. (2005), Nowak (1999) and De Lisle (1996) provide data for binturong, various cat species, the hog deer, the wild water buffalo and monitors respectively. Home range computations for various macaques and langur are also found in Nunn et al. (2004) and in Kaplan (2007). For all these species, when a range is proposed in the literature, the upper bound is used as a safety criterion. Also, when male and female data differ, the larger of the two – typically the male home range – is considered in the analysis.

When data are not available, range descriptions for the nearest taxonomic relative are used. Said differently, the problem-species parent Genus range, or failing that, taxonomic Family range is used. And as a further safety criterion, the inferred species range is set at the largest found in that family.

4.5. Opportunity Costs

Conservation of rare species entails costs, at least in the form of short-run costs. The scale of such costs might be approximated by estimates of the economic opportunities foregone in conversion to conservation uses: economists' measure known as opportunity cost.

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[4] Note that otters' home ranges are often computed in river length, that is, in kilometers, instead of km^2. The value is used as if it were a km^2 value for considering the complete length of the home range. Not doing so might underestimate the length of the otters' daily activities.

We estimate a worst-case scenario of opportunity cost for each site under consideration as the annual loss of economic output, Gross Product Originating (GPO), at the amphoe level[5]. New data on rural population and income at the amphoe (and smaller) levels recently has been derived by Healy and Jitsuchon (2007). Numerical extension of amphoe rural population by mean annual income per capita and expressing GPO as an index number permits the comparison of this measure of economic sacrifice among sites as well as the ability to compare it with regional and national aggregates. Source population and income data used are public data from the Thailand National Statistical Office (Royal Thai Government, 2000).

The Jitsuchon-Healy data sets are the result of an extensive econometric exercise in which the authors derive annual income and consumption data at the amphoe and tambon level from the 2000 Thai population census and Thai 2000 Socioeconomic Survey. While most of their paper consists of assessing the accuracy of the method by multiple means and the identification of small-area poverty-alleviation targets for domestic policy purposes, it also yields four very large data sets, one for income in urban places in each province, and one for consumption in urban places in each province, then two more for income and consumption for rural places (amphoe) in each province. At the authors' suggestion, the consumption series for rural places was used here, although the data for income and consumption were quite similar. The consumption series exhibited greater stability over time and location. Further, the two measures sum to a similar value over time.

5. Applying the Models on the Thailand Data Set

The set covering model seeks the minimum number of sites so that all species are represented at least in one of these selected sites. Running that first model (equations 1 to 3) on the Thailand data set with Xpress-MP (2006) gives a solution Z_1=11. This means that, under the hypothesis that each species can be considered as protected once it appears in one single site, all of them can be covered with the selection of 11 amphoes as nature reserves. Figure 5 shows the location of such parcels on a map of Thailand.

The outcome displayed in Figure 5 is one of the many alternate optima that can be found for this integer program. Said differently, other sites are able to protect all species as well but it is not possible to cover all species with a smaller number of sites.

For choosing the smallest number of sites, the model selects those parcels that are either species-rich (that is, they include many different species) as well as those that are the only natural habitat for the rarest of the species (as three species are only present in one single amphoe).

[5] If the purpose of the analysis is to determine absolute cost values, it might have been more appropriate to estimate the capital value of income streams over time to get a global discounted cost number for each site. However, in addition to the fact that no dynamics are used in this study, the purpose is not to work with precise cost figures but to have estimates for comparing the various alternatives. Since all data are computed with the same methodology, the comparison is therefore acceptable.

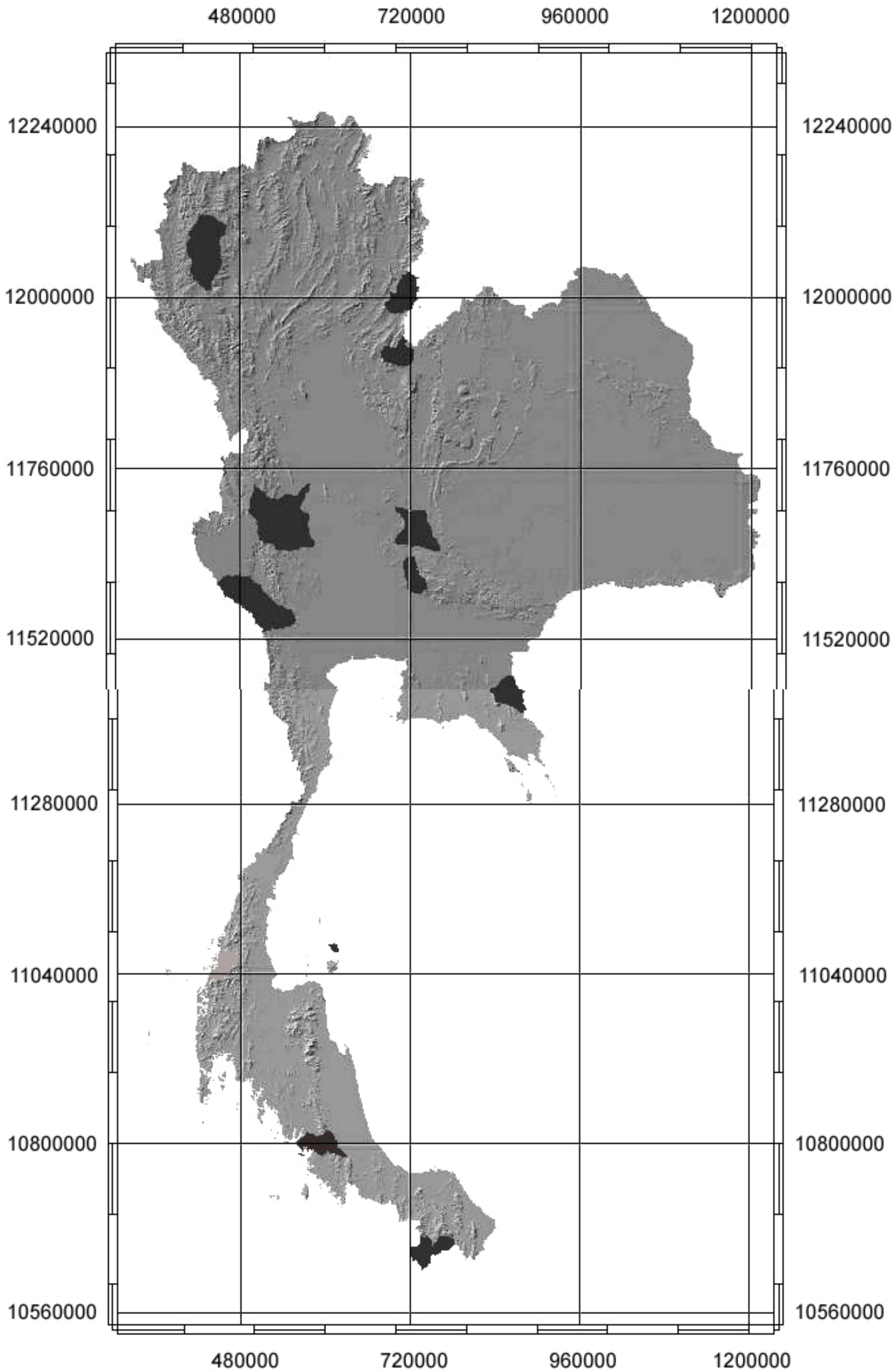

Figure 5[6]. Districts selected, SSCP solution

[6] Base map source for Figures 5 to 7: Souris (2007).

The second model described in section 2 (equations (5) to (10)) takes home ranges into consideration but does not account for differences in site areas – which means that the model implicitly assumes that all sites have the same surface areas. This is obviously not the case when applied to the Thailand data set. Hence, we should rather concentrate on Model 3, represented by equations (11) to (16), whose aim is to select the smallest number of sites while representing all species in the reserve network and requiring that each shorter range and longer range species enjoy a habitat whose area is at least equivalent to that of its respective maximal home range. Besides, the model also ensures that longer-range species benefit from contiguous sites when protected.

Most of the 68 vertebrate species used in this study are animals of small body size that typically have a very small home range (from less than 1 km^2 to a few km^2). Some species have a larger home range (from 5 to 60 km^2) and only two species, the Asian elephant and the tiger, are considered for this paper as longer-range species as they may respectively cover a maximum of 300 and 100 km^2 for their activities. Comparing these two home ranges with district surface areas may lead us to overcome the problem of selecting contiguous sites. As a matter of fact, many sites have a larger area than 300 km^2. Equation (14), requiring the selection of at least two contiguous sites for each longer-range species, might be considered as unnecessary. If that equation were deleted, the model would simply select a sufficiently large amphoe in which each longer range species is able to enjoy at least its own home range. The purpose for not deleting that equation and for requiring selection of at least two contiguous sites for longer-range species is to account for edge effects. As a matter of fact, it may be possible that the species occurs close to the border of one amphoe. If this is the case, it might be advisable to select its neighboring site as well as species do not know amphoes' borders[7]. Obviously, this is less true for shorter-range species, as by definition, their daily activities are much more limited in terms of area.

The third model (equations (11) to (16)) selects 13 sites, 11 of which protect shorter-range species and two contiguous sites protecting both longer-range species. The site selection is displayed in Figure 6.

Compared with the SSCP model's solution, two additional sites are selected. Because all shorter range species (that is all species but two) need only a single site for protection (since all their home ranges are smaller than the area of each parcel selected), the model would not find any of these species-rich area or area where the rarest of the species occur for selection as natural habitat for longer range species. Hence, two additional contiguous sites are selected for longer-range species – they are depicted in white while the other sites selected for shorter-range species are in dark grey. Also, because of the site minimization objective, the model has chosen only two additional sites instead of three or four sites because both Asian elephant and tiger occur in the same two sites that are shown in white[8].

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[7] One may oppose to this argument that the species in question may live at the other side of the amphoe and therefore, say if the second site selected is the one at the right border of the first site and if the species lives close to the left border, the contiguous site selection will be mis-placed. To overcome this problem, while running the model, we took this element into consideration and selected those contiguous sites for longer range species where either contiguous site could be selected, that is where the long range species is present in both contiguous sites.

[8] For comparison purposes, running model 2 (equation (5) to (10)) brings about the selection of 13 sites as well among which the same four sites as the SSCP and model 3 are selected and the same two contiguous sites for longer range species are selected as well.

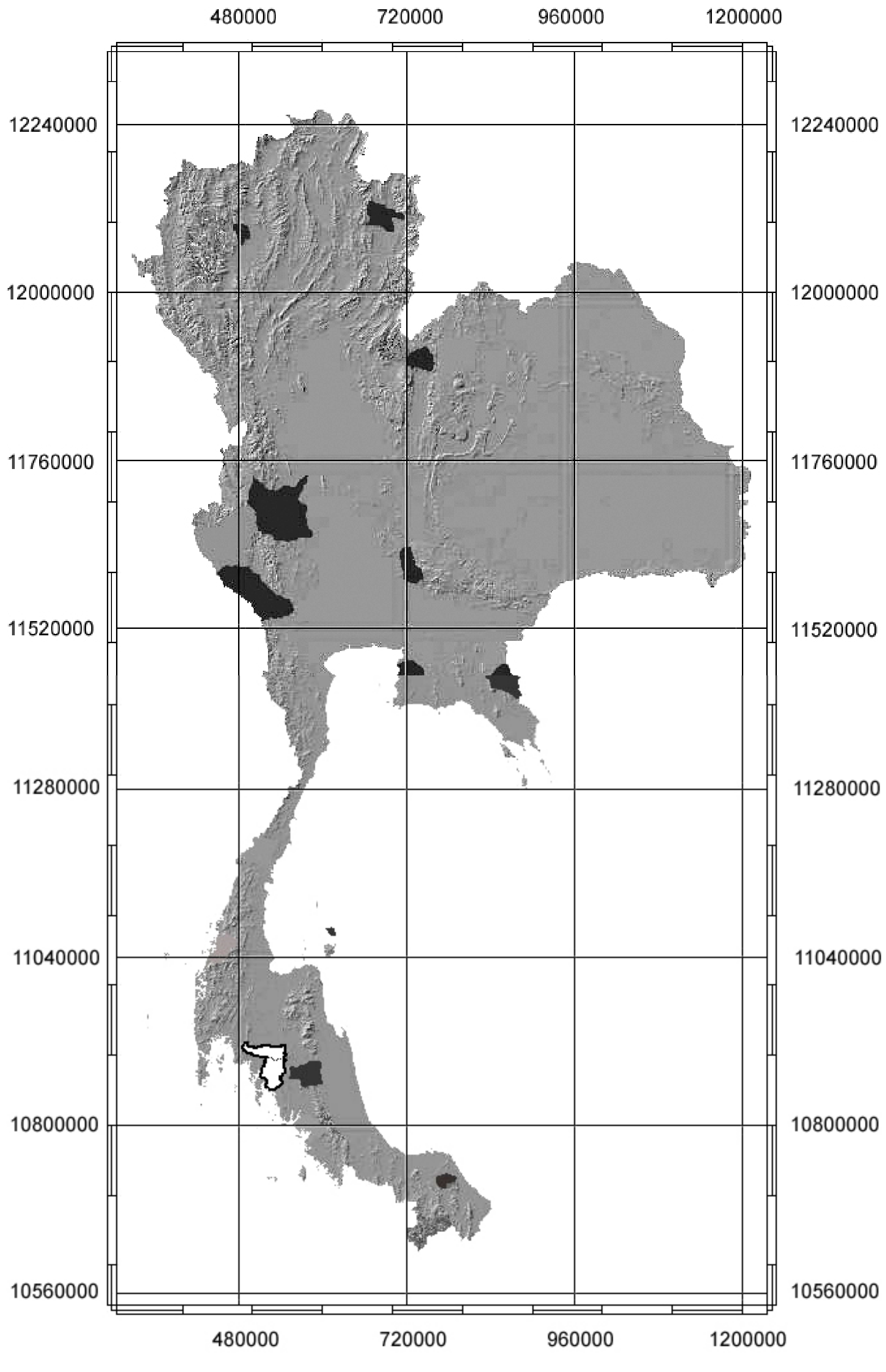

Figure 6. Districts selected, Model 3 solution without cost consideration

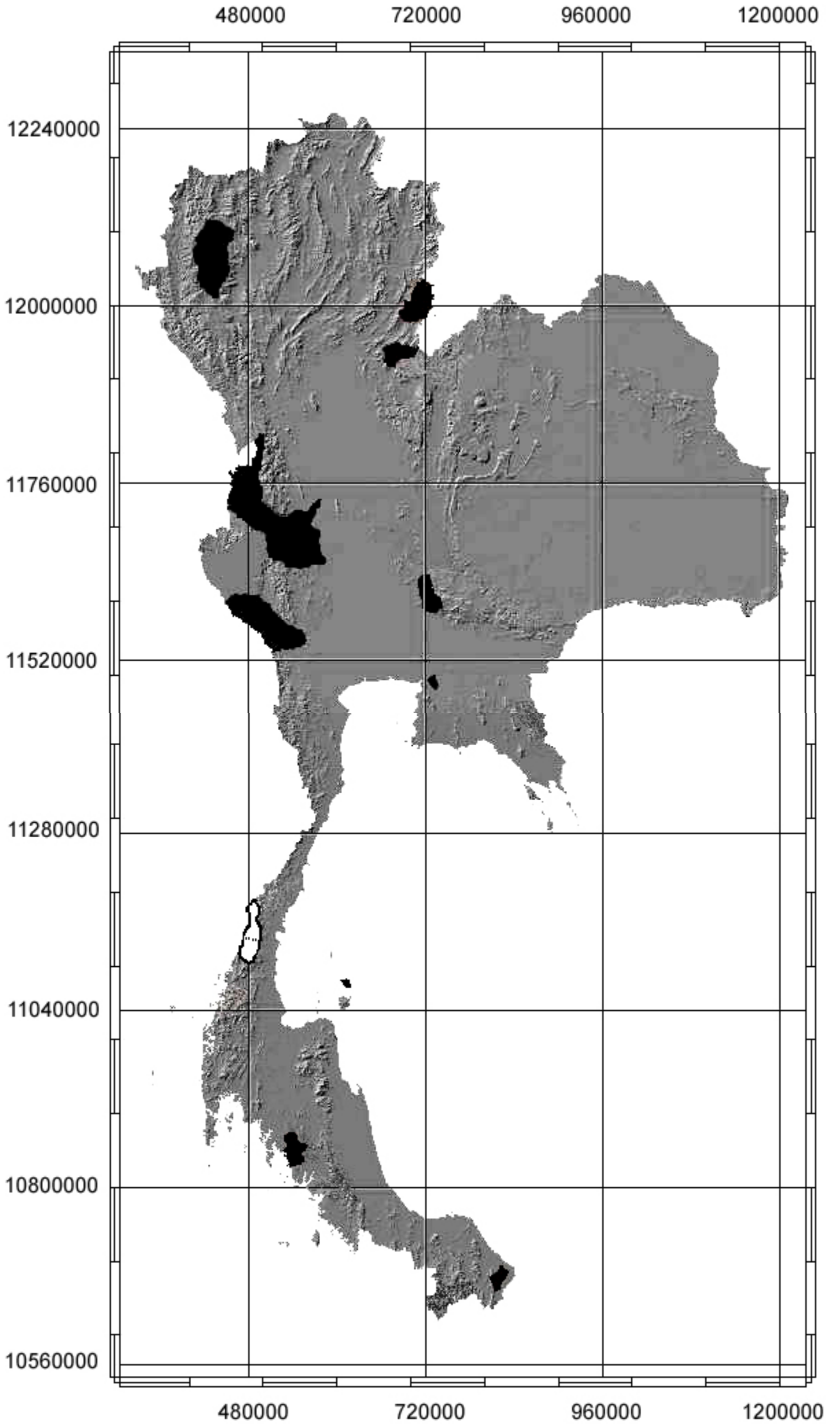

Figure 7: Districts selected, Model 3 solution with cost consideration

The model has selected areas ranging from 166 km^2 to 5126 km^2 for short-range species while the two contiguous sites have a combined surface area of 1832 km^2. Compared with the SSCP solution, five sites are identical in both models and two other ones are very close (selection of either a contiguous or a very nearby site – often with similar species frequency characteristics) while the others are different. Three sites must of course be identical as they are the single site available for the three rarest species but it is not so for the fourth and fifth sites. Furthermore, one of these sites is present in all runs and in all alternate optima that can be found for all of the three models detailed in Section 2. That very site might have the irreplaceability characteristic, as determined by Jacobi et al. (2007).

Finally, considering cost elements can refine the model. As a matter of fact, setting land aside for nature reserve entails an opportunity cost in terms of output foregone, as explained above in Section 4.5. Replacing objective (11) by equation (4) in Model 3 gives a minimum cost value of 606 Million Baht[9] for 13 sites selected, as depicted in Figure 7.

The model now chooses the least expensive sites for protecting all species within a suitable area range meeting habitat requirements for each and every species. The opportunity cost of defining an amphoe as a nature reserve varies from 21 to 489 for the complete data set and selected districts' costs range from 21 to 125 – the highest cost being reserved for selecting the only site in which one of the species is present. This is obviously a more cost efficient alternative as the optimal solution of the site-minimizing Model 3 (the same model without cost consideration), amounts to a cost value of 1285, which is more than twice that of the cost-minimizing solution of Model 3. The gain is very important even though the site selection was fairly limited for various reasons. First, 3 sites out of 13 must be selected whatever their cost because 3 species are reported in a single site in Thailand. Second, four additional species are present only in two or three districts, which also severely limits the choice for a lower-cost parcel selection. Third, the number of species in this study is limited to threatened and endangered species, which prevents the bi-modal species frequency pattern often observed in studies that involve a large number of both common and rare species. Notwithstanding these remarks, the cost-efficient model is by far less costly than the site-minimizing problem.

Only the three sites that contain the three rarest species are common to the site-minimizing Model 3 and the cost-minimizing Model 3. However, six sites are common to the cost-minimizing Model 3 and the SSCP. This difference has the effect of moderating the irreplaceability characteristic mentioned above. In fact, it seems that, aside from the three sites that must be selected, as they are the only option for the three "site-endemic" species, no other site is totally irreplaceable but some areas (sites that are close-by) are nevertheless selected in most of the computer runs.

Moreover, concerning the longer-range species, the two contiguous sites selected for the cost-minimization problem (white color in Figure 7) are different from those selected with the site-minimization problem (white color in Figure 6). The reason is of course the lower opportunity costs of having these two sites in a reserve network. In this modified model, the new costs for contiguous sites correspond to a total of 84 compared with the higher amount of 155 for the site-minimization solution.

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[9] This number should not be viewed as a cost figure for purchasing the site as it represents the annual income of that amphoe, as explained in Section 4. All subsequent opportunity cost data will be expressed in Million Baht as well.

6. CONCLUSION

A classical method for protecting species is to set aside pieces of land to be part of a nature reserve. This paper formulates various set covering models aimed at protecting all those vertebrate species considered as threatened and/or endangered by the IUCN in Thailand. Species presence/absence data are assembled from the results of previous studies, area of amphoes, the nations' smallest districts are considered, home-ranges of each of the 68 species under consideration are gathered from various sources, and opportunity costs in terms of output foregone when a district is selected as part of the reserve are computed.

Starting with the well-known set covering problem, protecting each and every species in at least one district, the model is further refined to account for home ranges and for opportunity costs of creating reserves.

This paper should be considered as a theoretical exercise since it does not incorporate existing nature reserves in Thailand. Rather, it starts from the theoretical hypothesis that no reserve exists in the country and hence, it selects those districts best suited to provide natural and large enough habitat for preserving all species at stake. The geographic sites selected, which are the various outcomes of the models, are displayed on maps in Figures 5 to 7.

Clearly, these results are not the only possible results. There are in general many alternate optima for such integer programs. In this particular case, there may be fewer optima than in other reserve site selection models applied on other large data such as the Oregon data set because three sites (that is more than 20 percent of all sites needed to protect all species with home range requirements) contain single species and must therefore be selected. Moreover, other species also occur in very few sites, which limits the selection procedure and the quantity of alternate optima.

Another important point to consider in the interpretation of the above results is that, as in Williams et al. (2003), a selected amphoe can be thought of, not as a site to acquire or a reserve, but as a general locale in which specific parcels for acquisition could be identified by local planners and decision makers. These smaller specific parcels are nevertheless expected to be large enough to respect the home-range constraints.

Finally, this paper does not consider competition among species and problems of co-location of habitat. It is well established in the biology and ecology literature that, for a variety of reasons, some species do not occur in the same area as other species. On the contrary, some species may need to live in the same surroundings as other species. These biological elements are not taken into account here and might somewhat modify the results.

REFERENCES

Arthur, J., Camm, J., Haight, R., Montgomery, C., Polasky, S. (2004). Weighing conservation objectives: maximum expected coverage versus endangered species protection. *Ecological Applications*, *14*, 1936-1945.

Bioinventories of the world (2006). *Biological inventories of the world's protected areas.* International center for the environment, University of California, Davis. www.ice.ucdavis.edu/bioinventory/bioinventory.html

Church, R., Stoms, D., Davis, F. (1996). Reserve selection as maximal covering location problem. *Biological Conservation 76*, 105-112.

Costello, C. and Polasky, S. (2004). Dynamic Reserve Site Selection. *Resource and Energy Economics, 26*, 157-174.

Csuti, B., Polasky, S., Williams, P., Pressey, R., Camm, J., Kershaw, M., Keisler, A., Downs, B.,

Hamilton, R., Huso, M., Sahr, K (1997). A comparison of reserve site selection algorithms using data on terrestrial vertebrates in Oregon. *Biological Conservation, 80,* 83-97.

De Lisle, H. (1996). *Natural History of Monitor Lizards.* Krieger Piblishing Company, Florida.

Eisenberg, F. (1997). Tapirs, status survey and conservation action plan. In Brooks, D. (Ed.). *IUCN Tapir Specialist Group.* IUCN. Switzerland.

Grassman, L., Tewes, M., Silvy, N. (2005). Ranging, habitat use and activity patterns of binturong 'arctictis binturong' and yellow-throated marten 'mates flavigula' in north-central Thailand. *Wildlife Biology, 11,* 49-57.

Grassman, L. (2001). *Spatial ecology and conservation of the felid community in Phu Khien wildlife sanctuary.* Progress report to CAT.

Haight, R., ReVelle, C., Snyder, S. (2000). An integer optimization approach to a probabilistic reserve site selection problem. *Operations Research 48*, 697-708.

Hamaide, B., ReVelle, C.S. and Malcolm, S (2006). Biological Reserves, Rare Species and the Trade-off between Species Abundance and Species Diversity. *Ecological Economics, 56*, 570-583.

Hamaide, B., Williams, C., ReVelle, C. (2008). Cost-efficient reserve site selection favoring persistence of threatened and endangered species. *Geographical Analysis,* accepted for publication.

Harris, S., Cresswell, W., Forde, P., Trewhella, W., Woollard, T., Wray, S. (1990). Home range analysis using radio tracking data: a review of problems and techniques particularly as applied to the study of mammals. *Mammal Review, 20,* 97-123.

Healy, A. J., & Jitsuchon, S., (2007). Finding the Poor in Thailand, *Journal of Asian Economics* In print.

Humphrey, S., Bain, J. (1990). *Endangered animals of Thailand.* Sandhill Crane Press, 468p.

IUCN (2006). *Summary statistics for globally threatened species.* 1990, 1994, 1995, 2002, 2006. Gland, Switzerland. www.iucnredlist.org

Jacobi, S., ReVelle, C., Pressey, R., Williams, J. (2007). Novel operations research methods for efficiently determining irreplaceable sites for conservation. *Environmental Modeling and Assessment, 12,* 91-103.

Kahn, M. (1967). Movement of a herd of elephants in the upper Perak area. *Malay Nat. Journal, 20*, 18-23.

Kaplan, H. (2007). Primate database. www.unm.edu/-hkaplan/pnas/primatedatabase.xls

Kirkpatrick, J. (1983). An iterative method for establishing priorities for the selection of natural reserves: an example from Tasmania. *Biological Conservation 25*, 127-134.

Legakul, ., McNeely, A. (1977). *Mammals of Thailand.* Association Conservation Wildlife, Bangkok, 758p.

Leighton, S. (1987). Gibbons: territoriality and monogamy. In Smuts, B., Cheney, D., Seyfarth, R., Wrangham, R. and Struhsaker, T. (Ed.). *Primate societies.* University of Chicago Press, Chicago, 135-145.

Malcolm, S., ReVelle, C., (2005). Preserving species diversity with backup coverage. *Environmental Modeling and Assessment, 10*, 99-105.

Margules, C., Nichols, A., Pressey, R. (1988). Selecting networks of reserves to maximize biological diversity. *Biological Conservation 43,* 63-76.

Marianov, V., ReVelle, C., Snyder, S. (2007). Selecting compact habitat reserves for species with differential habitat size needs. *Computers and Operations Research,* in print.

Mason, C., McDonald, S. (1986). *Otters: ecology and conservation.* Cambridge University Press, Cambridge.

Ministry of Interior. Royal Thai Government. www.moi.go.th www.amphoe.com

Novak, R. (1999). *Walker's mammals of the world.* 6th edition. The Johns Hopkins University Press, Baltimore.

Nunn, C., Altizer, S., Sechrest, W., Jones, K., Barton, R., Gittleman, J. (2004). Parasite species richness and evolutionary diversity of primates. *American Naturalist, 164,* S90-S103.

Odden, M., Wegge, P., Storaas, T. (2005). Hog-deer 'axis procinus' need threatened tallgrass floodplains. *Animal Conservation, 8,* 99-104.

Polasky, S., Camm, J., Garber-Yonts, B. (2001). Selecting biological reserves cost-effectively: an application to terrestrial vertebrate conservation in Oregon. *Land Economics, 77*, 68-78.

Possingham, H., Day, J., Goldfinch, M., Salzborn, F. (1993). The mathematics of designing a network of protected areas for conservation. In: Sutton, D., Cousins, E., Pierce, C. (Eds.), *Decision sciences, tolls for today,* Proceedings of the ASOR Conference, Adelaide, Australia.

ReVelle, C., Williams, J., Boland, J. (2002). Counterpart models in facility location science and reserve selection science. *Environmental modeling and assessment 7,* 71-80.

Rowe, N. (1996). *The pictorial guide to the living primates.* Pogoneas Press, East Hampton, NY.

Royal Thai Government (2000). *Socio-economic survey: 2000.* Thailand Population and housing census.

Sheerin, J. (2007). *Protecting Endangered Species in Thailand.* Manuscript.

Sompoad, S., Vadavudh, S. (1995). Populations of gaur and banteng and their management in Thailand. *Natural History Bulletin Siam Society, 43,* 55-83.

Souris, M. (2007). *Thailand DEMS. 400m Resolution Maps of Thailand.* IRD, Asian Institute of Technology, Pathum Thani, Thailand. www.star.ait.ac.th/~souris/thailand.htm.

Storch, D., Sizling, A. (2002). Patterns of commonness and rarity in Central European birds: reliability of the core-satellite hypothesis within a large scale. *Ecology 25,* 405-416.

Snyder, S., ReVelle, C., Haight, R. (2004). One and two objective approaches to an area-constrained habitat reserve site selection problem. *Biological Conservation, 119,* 565-574.

Toregas, C., Swain, R., ReVelle, C., Bergman, L. (1971). The location of emergency service facilities. *Operations Research 19,* 1363-1373.

Toregas, C., ReVelle, C. (1973). Binary logic solutions to a class of location problems. *Geographical Analysis, 5,* 145-155.

Underhill, L. (1994). Optimal and suboptimal reserve selection algorithms. *Biological Conservation 35*, 85-87.

Williams, J.C., ReVelle, C.S., Bain, D. (2003). A decision model for selecting protected habitat areas within migratory flyways, *Socio-Economic Planning Sciences 37*, 239-268.

Williams, J., ReVelle, C., Levin, S. (2005). Spatial attributes and reserve design models: a review. *Environmental Modeling and Assessment*, *10*, 163-181.

Xpress MP, (2006). Xpress MP software development, Dash optimization, Release 2006, Englewood Cliffs, NJ, USA.

In: Ecological Economics Research Trends
Editor: Carolyn C. Pertsova, pp. 231-250
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Chapter 11

THE PRICE FORMATION PROCESS IN TIMBER AUCTIONS AND THE FACTORS AFFECTING THE PRICE OF BEECH TIMBER IN TURKEY: A CASE STUDY

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ABSTRACT

Numerous factors affect the auction price of principal forest products of the state forest enterprises in Turkey. This study tries to determine the factors affecting the price of third-class normal-sized beech timber sale by auctions. It is carried out in two rival state forest enterprises (Bartın and Yenice) of Zonguldak Regional Forest Directorate in the West Blacksea Region of Turkey. The data in this study has been obtained from a total of 149 timber auctions in the period 1998-2002. The effects of seasonal and other factors on the auction price of beech timber are investigated by variance analysis, seasonal and monthly indices, correlation, regression, and principal component analyses respectively.

The analyses indicate that beech timber prices in Yenice differ significantly over the seasons, but there are no significant differences in beech timber prices in Bartın. Moreover, the month with the highest timber price is April in both Yenice and Bartın. A correlation analysis indicates that Yenice is more dependent on Bartın than the other way around. Furthermore, according to the regression analysis, the rival prices and price

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mark-ups significantly explain the variation in prices. Additionally, the quantity offered during an auction significantly explains the prices in Bartın, while the log size and time between auctions significantly explains the prices in Yenice. Finally, the principal component analysis indicates that the (1) price and timing of the auctions, (2) demand level, and (3) average volume per log are the main decision variables.

Keywords*:* beech timber; auction price; correlation analysis; regression analysis; principal component analysis.

1. INTRODUCTION

Rapid population growth and technological development will continuously increase the demand for both quantity and variety of forest products in almost every country of the world (Lyke and Brooks, 1995; Brooks, 1997). On the one hand, concerning increasing demand, the presence of imperfect competition conditions and inadequate supply into the forest product's markets lead to higher timber prices and do increase enterprise's incomes. On the other hand, if marketing decisions are based on customer satisfaction and provide responsibility for economic, social and environmental conditions, they will contribute to a sustainable management of forest resources (Juslin and Lintu, 1997). In this context, insights into the price formation process, factors affecting the price of forest products (Nautiyal, 1988; Duerr, 1993) and optimal marketing policies (Palo et al., 2001) are needed for sustainable forest management.

Price as one of the four Ps (Product, Price, Place and Promotion) in marketing has an effect on the total income of a firm, and it is also an easy to understand, quantitative and one-dimensional variable (Kotler, 1972). Generally, many factors affect the price of goods, such as the level of demand and supply of goods, its features, prices and quantities of available substitutes and complementary goods,[3] features and structures of markets, consumer tastes and preferences, rivals (or competitors), owners of production factors, governments and different interest groups (Mucuk, 1984). Providing insight into the role of these factors and to formulate policies of marketing and management are very important for the firm's profitability and sustainability.

1.1. Economic Theory of Price Formation under Perfect and Imperfect Competition

Economic theory of price formation under perfect and imperfect competition is reviewed to explain price formation in auctions. For that reason, Figure 1is drawn to explain the price formation as an equilibrium condition under perfect and imperfect competition (Nautiyal, 1988; Klemperer, 1996; Türkay, 1997; Geray, 1998; Dinler, 2003).

[3] On substitutes and complements see for instance Bulow et al. (1985).

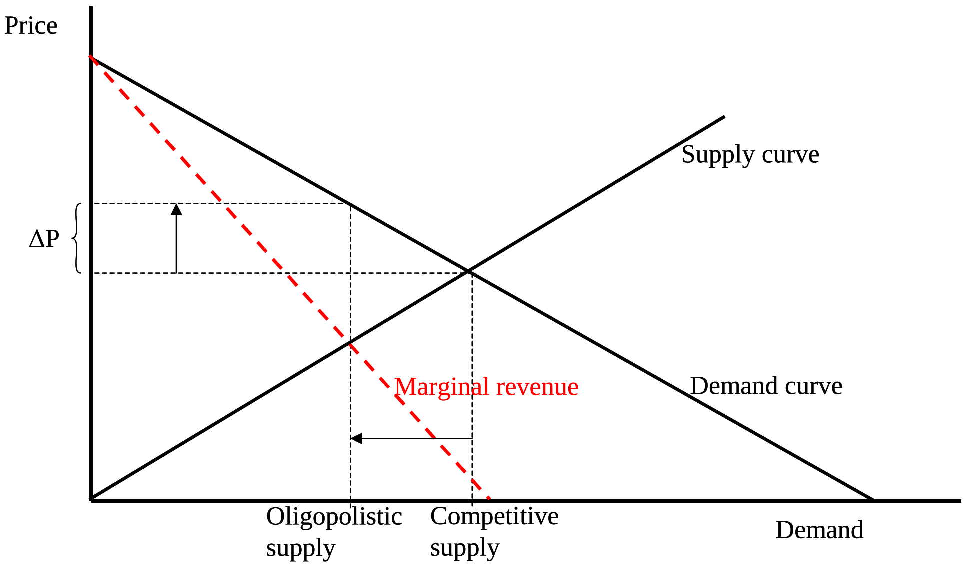

Figure 1. The equilibrium price under perfect and imperfect competition.

Under perfect competition, there are many sellers and buyers, firms produce a homogeneous commodity, firms and buyers possess perfect information and they are free to enter the market or leave it, where all resources are perfectly mobile. A typical outcome of perfect competition is that the marginal revenue is equal to the marginal cost resulting in zero profits under equilibrium. Moreover, the marginal revenue curve coincides with the demand curve under perfect competition. In equilibrium the supply and demand curve intersect in Figure .

Imperfect competition is more likely in real life situations, because the conditions of perfect competition do not always hold in the market due to different reasons *(market failures)*. Imperfect competition includes oligopolisitic (few firms) and monopolistic (one firm) competition. As shown in Figure 1, under imperfect competition firms have market power to change the price of their output. Moreover, prices can also be affected by other factors such as substitution, good quality, consumer income, etc. This can be explained by drawing the marginal revenue curve for oligopolistic firms as shown by the striped line in Figure 1. The supply reduction results in a demand reduction, where the oligopolistic firms see the market price increase by ΔP, as compared to perfect competition.

There is a close analogy between the theory of optimal auctions (that maximises the seller's expected revenue) and theory of price formation under imperfect competition (Bulow and Roberts, 1989; Klemperer, 1999). In practice, the price formation in standard auctions resemble imperfect competition, because there is a single seller who controls the trading mechanism, and they are designated with a certain number of buyers who submit bids and have downward-sloping demand curves. Therefore, it can be said that imperfect competition is likely to occur among sellers and buyers in auctions. Hence, the selling price in auctions is affected by features of the buyers and goods supplied, and sometimes by demand and other factors related to imperfect competition.

1.2. Fundamentals of Auction Theory

Auctions have been around for a long time, but they entered the economics literature relatively recently. There are four basic types of auctions in the literature, namely ascending, descending, first-price sealed bid, and second-price sealed bid (Vickrey 1961; McAfee and McMillan, 1987; Maskin and Riley, 1985; Klemperer, 1999). The rules of the ascending-bid auction (also called the open or oral auction) are further described here, because beech timber in Turkey is auctioned with this method without entry fee. In this auction the seller announces the reserve price, bidders enter the auction and the price increases continuously, while bidders gradually leave the auction, and the price increases until a single bidder remains, and that bidder acquires the auctioned good at the selling price.

All auctions have the same expected revenue, because of each bidder is making the same expected payment. This *Revenue Equivalence Theorem* is a fundamental result in auction theory (Rosenthal, 1980; Maskin and Riley, 1985; Klemperer, 1995, 1999). This theorem is based on three crucial assumptions: Risk-neutrality, independent private information, and private values or signals. Since the excepted payment of all bidders across the different possible types is the same, all auctions yield the same expected revenue (Klemperer, 1999).

On the other hand, Bulow and Roberts (1989), and also Bulow and Klemperer (1996) show that the expected revenue from an ascending auction equals the expected marginal revenue of the winning bidders (marginal revenue approach to auctions). Finally, if the bidders are risk-neutral and their information signals are independent, the expected revenue from any standard auction equals the expected marginal revenue of the winning bidder.

The auction situation can be explained by applying the usual logic where marginal revenue equals marginal cost. A seller should not sell below the price where marginal revenue equals marginal cost. The optimal selling price occurs where marginal revenue equals the seller's marginal cost. Hence, a majority of the sellers have marginal revenues higher than their marginal cost. So under the assumptions of the revenue equivalence theorem, all standard auctions are optimal, where the seller settles at the optimal reserve price (Klemperer, 1999).

1.3. Scope and Purpose

The forest area in Turkey consists of 20.7 million hectares, and 99.9% of this area is under state ownership (DPT, 2001). The General Directorate of Forestry (GDF) is not only a public institution, but also a public economic enterprise working under conditions of imperfect competition. In total, 241 state forest enterprises manage the whole forest area on behalf of the State. The pricing of the timber sales by auctions at the State Forest Enterprises (SFEs) follow similar marketing policies and consist of a process with four steps, according to the *mark-up model*, which is a well-known financial pricing method (Acun, 1977; Çağlar 1989; Türker, 1996; Daşdemir, 2003). *In the first step*, the cost price of the product is calculated as the sum of the direct and indirect production cost, stumpage price, marketing expenditure and a dividend portion. *In the second step*, the management determines the reserve price of the product. *In the third step*, the real selling price of the product is established by the interaction of demand and supply during the auction. *In the last step*, the consumer price of the product is determined after adding some legal payments to the real selling price.

The Bartın and Yenice forest enterprises in the Zonguldak Regional Forest Directorate (ZRFD) resemble each other with regard to capacity and scale; they have the same demand and supply features, produce similar products, address the same market, and have the same marketing system. Moreover, they are rival enterprises (Daşdemir, 2003). Therefore, the prices and quantities of timber sales of these two enterprises are affecting each other under imperfect competition. Consequently, investigating and comparing the timber auctions of these two enterprises and determining the factors affecting the timber prices is useful for gaining insight into effective marketing and economic sustainability.

To achieve this aim, the research analyses the seasonal effects on the auction price of beech timber (*Fagus orientalis* Lipsky.) harvested from natural forests, using indices for seasonal and monthly effects. The linkages among the variables are pre-investigated by a variance analysis. Furthermore, the effects of other factors are investigated by correlation, regression and principal component analysis analyses. By evaluating and discussing the results with regard to the current marketing mentality and policies, this paper contributes to enterprise management to make effective marketing decisions.

The paper is outlined as follows. Section 2 presents the research area, collected data, and used methods. The results are presented and discussed in Section 3, consisting of time variation in the price, correlation, regression and principal component analyses. The final section concludes.

2. MATERIALS AND METHODS

2.1. Research Area

The research is carried out in two rival state forest enterprises (Bartın and Yenice) of the ZRFD in the West Blacksea Region of Turkey (Figure 2), possessing an important share of beech forests, growing stock, production and sales. Auctions are investigated, because they reflect real pricing.[4] Furthermore, third class normal sized (3CNS) beech timber is taken into consideration, because the incomes from sales of 3CNS form the most important part (60%) in total incomes in Bartın and Yenice SFE.

2.2. Data and Definition of Variables

The data in this study has been obtained from a total of 149 timber auctions in the period 1998-2002, namely 78 and 71 in Bartın and Yenice SFEs respectively. Before applying statistical analyses, the variables that supposedly affect the price of beech timber are shown in Table 1. For both locations a set of 9 different variables are used in the analysis. After each auction a final sales price (PAB, PAY) is paid for the actual quantity sold (SAB, SAY), which

[4] On forest management in Turkey see Atmiş *et al*. (2007a,b).

is lower or equal to the quantity offered (DAB, DAY).[5] The actual sales price can be substantially above the reserve price, because of imperfect competition. This price mark-up (EAB, EAY) is measured as a percentage.[6] As a further characteristic of the timber sold, the average volume per log (VAB, VAY) and the stack sizes of timber (GAB, GAY) are available. Furthermore, the date of auction is given as well, and from these dates additional variables are derived, namely the time between two auctions (TAB, TAY), and the time difference between the current auction and the previous auction (CAB1, CAY1) and the time difference between the current auction and the next auction (CAB2, CAY2). The latter four variables will only be used in a principle component analysis.

Figure 2**.** Research area.

In order to study the impact of rival SFEs among each other, the two data sets are pooled where each observation of Bartın is matched with each nearest observation of Yenice later on in time and where each observation of Yenice is linked with each nearest observation in Bartın earlier on in time. In most cases the auctions are alternating, however in some instances there can be up to four auctions in the same SFE before a new auction takes place in the rival SFE. As a result the pooled data set consists of 92 observations.

5 By subtracting the actual quantity sold from the quantity offered, a residual remains. This variable is excluded from the analysis as this is typically fully correlated with SAB (SAY) and DAB (DAY).

6 The price mark-up can also be used to calculate the reserve price, namely RPB = PAB/(1+EAB), and RPY = PAY/(1+EAY).

Table 1. Names, Labels and Units of the Variables in the Research

No	Name and definition of variable	Label	Unit
1	Sales price in each auction of Bartın	PAB	US $/m^3
2	Price mark-up in each auction of Bartın ((selling price – reserve price)/(reserve price) x 100)	EAB	%
3	Total quantity offered in each auction of Bartın	DAB	1000 m^3
4	Total quantity sold in each auction of Bartın	SAB	1000 m^3
5	Average volume per log in each auction of Bartın	VAB	1000 m^3
6	Stack sizes of timber in each auction of Bartın (supply quantity/number of timber stacks)	GAB	1000 m^3
7	Time between two auctions following each other (last auction date-first auction date) in Bartın	TAB	days
8	Cross time between date of each auction in Bartın and date of the former auction in Yenice	CAB1	days
9	Cross time between date of each auction in Bartın and date of the following auction in Yenice	CAB2	days
10	Sales price in each auction of Yenice	PAY	US $/m^3
11	Price mark-up in each auction of Yenice ((selling price – reserve price)/(reserve price) x 100)	EAY	%
12	Total quantity offered in each auction of Yenice	DAY	1000 m^3
13	Total quantity sold in each auction of Yenice	SAY	1000 m^3
14	Average volume per log in each auction of Yenice	VAY	1000 m^3
15	Stack sizes of timber in each auction of Yenice (supply quantity/number of timber stacks)	GAY	1000 m^3
16	Time between two auctions following each other (last auction date-first auction date) in Yenice	TAY	days
17	Cross time between date of each auction in Yenice and date of the former auction in Bartın	CAY1	days
18	Cross time between date of each auction in Yenice and date of the following auction in Bartın	CAY2	days

2.3. Analysis Methods (Statistical Analyses)

To determine the effects of seasons and other factors on the auction price of beech timber, all obtained data are evaluated by a variance analysis, indices are derived for seasonal and monthly effects (SE and ME), and correlation, regression and principal component analyses are undertaken (Harman, 1967; Kalıpsız, 1988; Sokal and Rohlf, 1995).

In applying these statistical techniques, Version 9.0 of Statistical Package for Social Science (SPSS) is used. Also, the current prices of beech timber in the research period are adjusted by the Wholesale Prices Index of the State Institute of Statistics to the prices of 1998 base year, and the analyses and evaluations are done according to the adjusted (real) prices.

3. RESULTS AND DISCUSSION

3.1. Seasonality of Beech Timber Prices

Certain issues such as preparation of production, arrangement of stocks and adjustment of prices in a firm can be planned by having insight into the effects of seasonal fluctuations on sales, stocks and prices (Cillov, 1993). By using a variance analysis on the prices of beech timber auctions, it is found that the seasonal differences in timber price are significant at the 90% level in the Yenice SFE, while they are not significant in the Bartın SFE, using (Table 2).

Table 2. Results of Variance Analysis According to the Seasons

Enterprise	Sources of Variation	Sum of Squares (US $)	Degree of Freedom	Mean Squares (US $)	F Value
Bartın	Between Groups	707.117	3	235.706	0.115
	Within Groups	152.069.542	74	2.054.994	
	Total	152.776.660	77		
Yenice	Between Groups	15.808.754	3	5.269.585	**2.160**
	Within Groups	163.434.594	67	2.439.322	
	Total	179.243.348	70		

The Duncan Test[7] is applied to determine different seasonal prices and the results are given in Table 3. According to Table 3, the highest price in Yenice SFE occurred in *spring*; prices of *summer* and *autumn* followed this, and the lowest price occurred in *winter*. Booming markets, increasing the demands of the forest industry organizations in spring, caused the beech timber prices to rise. This price increase continued into the summer, albeit at a lower rate. However, the prices in both SFEs are relatively low in autumn and winter.

Table 3. Results of Duncan Test for Yenice SFE

Seasons	Subset for alpha = 0,05	
	Group 1 (US $)	**Group 2** (US $)
Winter	86,6	---
Autumn	122,9	122,9
Summer	140,3	140,3
Spring	---	172,2

Using the prices of beech timber sale by auctions in both SFEs, the seasonal effect (SE) can be defined as follows:

SE = (Average price per season /Average price per year) x 100 -100

The values of SE are calculated using these formulas, and the results for Bartın and Yenice SFE are shown in Table 4.

[7] It is known as Duncan's Multiple Range Test too. See also Duncan (1974) and Kalıpsız (1988).

Table 4. Timber Prices and SE Values as to the Seasons and Enterprises

Seasons	**Price** (US $/m^3)		**SE** (%)		**Number of observations**	
	Bartın	**Yenice**	**Bartın**	**Yenice**	**Bartın**	**Yenice**
Winter	121.3	86.6	-6.82	-33.62	14	7
Spring	**135.7**	**172.2**	**4.23**	**31.95**	**22**	**21**
Summer	134.4	140.3	3.19	7.48	23	28
Autumn	129.4	122.9	-0.60	-5.82	19	15
Average	130.2	130.5	0	0	78	71

The seasons with a positive SE have above average prices and seasons with a negative SE have below average prices. Hence, the beech timber prices are not statistically significantly different in the Bartın SFE. In other words, the price is relatively stable, because of a sufficient supply of fresh goods to the markets in all seasons. However, it can be said that spring and summer seasons have slightly higher prices than in others seasons.

Yet, in the Yenice SFE, the beech timber prices vary statistically significant over the seasons. The spring and summer seasons have relatively high beech timber and prices are lower in the other seasons. The price is 32% above average in spring and 34% below average in winter. An explanation for the result in Yenice SFE is that the demand during spring and summer is particularly high due to an increased seasonal demand for house construction, furniture, etc. Moreover, there is more oversupply ((supply - demand)/supply) in Yenice (33%) than in Bartın (19%), which also contributes to the more extreme swing in prices in Yenice.

The variance analysis based on prices of beech timber auctions shows that the prices are not statistically significantly different across months in both SFEs. However, since the results of the variance analysis on monthly price differences is generally suspect in this kind of studies (Acun, 1977), a monthly effect (ME) is defined, based on the monthly averages of five years prices of beech timber auctions in both SFEs as follows and the results are shown in Table 5.

ME = (Average price per month /Average price per year) x 100 -100

Table 5 shows that the average prices in March, April, May, June, July, September and October are above the yearly average, while they are below the yearly average in the other months in the Yenice SFE. The highest average price (+37%) and the lowest average price (–48%) occurred in April and December respectively. In the Bartın SFE, the average prices in April, May, June, August, October, November and December are above the yearly average and below the yearly average in other months. The highest average price (23%) and the lowest average price (–40%) occurred in April and January respectively. Hence, the month with the highest average price is *April* for both SFEs.

Table 5. Timber Prices and ME Values as to the Months and Enterprises

Months	**Price** (US $/m^3)		**ME** (%)	
	Bartın	**Yenice**	**Bartın**	**Yenice**
January	74.9	---	-40.00	----
February	113.9	98.8	-8.81	-26.36
March	85.3	172.7	-31.66	**28.65**
April	153.6	184.5	**23.03**	**37.48**
May	146.6	163.7	**17.36**	**21.96**
June	137.3	160.4	**9.96**	**19.51**
July	114.8	144.0	-8.10	**7.31**
August	145.1	124.0	**16.21**	-7.58
September	104.7	137.0	-16.15	**2.04**
October	146.3	137.7	**17.15**	**2.58**
November	136.1	83.2	**9.00**	-37.99
December	139.9	70.4	**12.03**	-47.58
Average	124.9	134.2	0	0

These results resemble other studies conducted in Turkey. In a study over the period 1970-1975 in the West Blacksea Region of Turkey (Acun, 1977), it was found that the prices in May, June, July, August, October, November and December were generally higher. Furthermore, another study over the period 1986-1996 in Gazipaşa SFE of Turkey (Ok, 1998), determined that the seasonal difference of auction prices of Calabrican pine (*Pinus brutia* Ten.) timber was statistically significant at the 90% level. Moreover, the prices in February, September, October, November and December were above average and below average in the other months.[8]

3.2. Correlations between the Variables Affecting Timber Prices

The results of a correlation analysis based on the 18 variables in Table 1 are given in Table 6. Actually, Table 6 consists of three sub-tables. The first table presents the correlations among the variables in the Bartin SFE, the third table shows the correlations among the variables in the Yenice SFE, while the second table consists of the cross-correlations between the two SFEs. Mainly the correlations with double stars in Table 6 are discussed and interpreted below in order to keep the number of relations to go through manageable and to focus on the most important correlations.

[8] The seasonal variation is not considered during the correlation, regression and factor analyses below, because of their largely low significance and to avoid technical difficulties. Moreover, seasonal dummies for months and years turned out to be insignificant during the regressions as well.

Table 6. Correlation Coefficients of Variables

	PAB	DAB	SAB	VAB	EAB	GAB	TAB	CAB1	CAB2
PAB	1.000	.048	.155	.223	.608**	-.245*	.328**	.220	.286*
DAB		1.000	.859**	.194	-.017	.492**	.031	-.237*	-.081
SAB			1.000	.081	.159	.347**	.088	-.186	.044
VAB				1.000	.085	-.044	-.074	.016	.041
EAB					1.000	-.265*	.297**	.337**	.457**
GAB						1.000	-.131	-.193	-.001
TAB							1.000	.321**	.208
CAB1								1.000	.344**
CAB2									1.000

	PAY	DAY	SAY	VAY	EAY	GAY	TAY	CAY1	CAY2
PAB	.930**	-.218*	.082	.005	.495**	-.160	.247*	.337**	.117
DAB	-.133	.122	.064	-.008	-.190	.114	-.204	-.189	-.089
SAB	-.020	-.050	.045	.018	-.059	.085	-.096	-.078	-.023
VAB	.169	-.277**	-.236*	-.029	.041	-.188	-.014	.066	.084
EAB	.467**	-.265*	.021	-.054	.541**	-.181	.462**	.353**	.182
GAB	-.361**	.169	.063	-.081	-.296**	.160	-.124	-.110	-.094
TAB	.358**	-.062	.114	.333**	.433**	-.090	.263*	.326**	.012
CAB1	.325**	-.049	.152	-.055	.445**	-.220*	.599**	.484**	.024
CAB2	.302**	-.100	.038	-.086	.443**	-.172	.798**	.706**	-.034

	PAY	DAY	SAY	VAY	EAY	GAY	TAY	CAY1	CAY2
PAY	1.000	-.182	.174	.121	.630**	-.187	.174	.365**	.069
DAY		1.000	.543**	.122	-.189	-.013	-.073	-.009	-.179
SAY			1.000	.022	.058	.068	.084	.048	-.060
VAY				1.000	.236*	-.170	-.055	.235*	-.024
EAY					1.000	.177	.463**	.484**	.029
GAY						1.000	-.177	-.200	.126
TAY							1.000	.456**	.005
CAY1								1.000	-.304**
CAY2									1.000

Note: The first and the third table are based on auctions in Bartın and Yenice separately, while the middle table is based on "interactions" among auctions in Bartın and Yenice.
** Correlation is significant at the 0.01 level (2-tailed);
* Correlation is significant at the 0.05 level (2-tailed).

In interpreting the correlation coefficients we first focus on the expected correlations, which are found in both SFEs and then continue with the unexpected correlations and try to explain them.

There are a number of correlations, which point in the same direction in both Bartın and Yenice. First of all there is positive significant correlation between the price and the price mark-up (PAB~EAB, PAY~EAY).[9] This implies that the prices and price mark-ups tend to move in the same direction and this confirms intuition, as prices are a function of the reserve price and the price mark-up (see footnote 6). Secondly, the quantity offered and sold is correlated (SAB~DAB, SAY~DAY). Inspection of the data show that the quantity offered

[9] Here we use the symbol '~' for convenience to denote a significant correlation.

and sold equate in a number of instances, while the difference between the quantity offered and sold is generally low, explaining the found correlations.

Thirdly, there is a positive low correlation between EAY with VAY in Yenice SFE, which means that a higher average volume per log in timber stacks leads to more demand for the stack and its price mark-up rises. Likewise, in another study in Spruce (*Picea orientalis* Lk. Carr) (Türker and Yazıcı, 1997), it was found that a higher average volume per log in timber stacks increased the demand for the timber stack and its price rose.

Next there is a very distinct pattern for the significant correlations with stack sizes. While there are no linkages in Yenice, the stack sizes (GAB) are weakly negatively correlated with price (PAB, EAB) and positively correlated with demand (SAB, DAB, RAB) in Bartın. An explanation for this is that an oversupply to the market generally leads to lower prices, where the larger supply is generally achieved by cutting larger trees as shown by the larger stack sizes in Bartın.

Finally, we can also give an interpretation to the timing among auctions. The time between two auctions is correlated to the price mark-up (TAB~EAB,PAB; TAY~EAY) and additionally to the price in Bartın. Furthermore, the time difference with the former auction in the reciprocal SFE is correlated with the price mark-up and the time difference with the previous auction (CAB1~EAB,TAB) and additionally to the price in Yenice. Also, the time difference with the next auction in the reciprocal SFE is correlated with the time difference with the previous auction in the reciprocal SFE (CAB2~CAB1,EAB) and additionally correlated with the price mark-up in Bartın. These results all point in the same direction, namely the price tends to go up once the time difference among auctions increases, both within the same SFE and with respect to the reciprocal SFE. Demand on the market tends to become higher once a new auction is delayed long enough and as a result the final prices rise.

There are many statistically significant cross-correlations as well. First of all, because of the strong correlation between price and price mark-up, the cross-correlations for (PAB, PAY) and (EAB, EAY) are nearly the same. Secondly, it is interesting to note that DAB and SAB in no way correlated to any variable in Yenice, while the opposite, namely DAY (and SAY) is correlated to price, price mark-up and log-volume in Bartın. Finally, there are numerous correlations among the cross-timing variables, but this is not surprising as these variables reflect the difference in time between the two consecutive auctions.

Thereupon, even if forest enterprises depend on the same Regional Forest Directorate, they are affected by each other’s marketing activities in the selling of principal forest products. Because of the structure of market requirements, the marketing activities of the Yenice SFE depend more on the marketing activities of the rival enterprise and are more affected by them. In another study on this subject, similar results were obtained from the timber sales by auctions of the Calabrican pine in Gazipaşa and Bucak SFEs of Turkey (Ok, 1997).

3.3. Explaining of Variations in the Selling Price

In order to explain variations in the timber prices at the SFE level and since our main aim is to investigate the factors that affect timber prices, multiple linear regression analyses are applied by taking PAB and PAY as the dependent variables. Two models are presented based on all variables of the same SFE and another one where additionally also variables of the reciprocal SFE could enter the equation. We have chosen for these two representations to compare the influence of intra-SFE local variables to inter-SFE variables. The results are presented in Table 7. The following equations are estimated:

$$lnPAB = Constant + \beta_1 lnRPB + \beta_2 lnDAB + \beta_3 lnSAB + \beta_4 lnVAB + \beta_5 lnGAB + \beta_6 lnTAB + error$$

$$lnPAB = Constant + \beta_1 lnRPB + \beta_2 lnDAB + \beta_3 lnSAB + \beta_4 lnVAB + \beta_5 lnGAB + \beta_6 lnTAB + \beta_7 lnPAY + \beta_8 lnEAY + \beta_9 lnDAY + \beta_{10} lnSAY + \beta_{11} lnVAY + \beta_{12} lnGAY + \beta_{13} lnTAY + error$$

$$lnPAY = Constant + \beta_1 lnRPY + \beta_2 lnDAY + \beta_3 lnSAY + \beta_4 lnVAY + \beta_5 lnGAY + \beta_6 lnTAY + error$$

$$lnPAY = Constant + \beta_1 lnPAY + \beta_2 lnRPB + \beta_3 lnDAB + \beta_4 lnSAB + \beta_5 lnVAB + \beta_6 lnGAB + \beta_7 lnTAB + \beta_8 lnEAY + \beta_9 lnDAY + \beta_{10} lnSAY + \beta_{11} lnVAY + \beta_{12} lnGAY + \beta_{13} lnTAY + error$$

The data are converted with logs in order to avoid the likely presence of heteroscedasticity and the price mark-up is replaced by the reserve price, which can be more meaningfully converted with logs, without loosing the impact of this variable.

Table 7. Results of Regression Analyses for Bartın and Yenice SFEs

	lnPAB		lnPAB		lnPAY		lnPAY	
	Coefficient	Std Error	Coefficient	Std Error	Coefficient	Std Error	Coefficient	Std Error
(Constant)	-0.168	(0.300)	0.418	(0.447)	-0.756^{**}	(0.353)	-1.014^{*}	(0.499)
lnPAB							0.610^{***}	(0.156)
lnDAB	0.140^{***}	(0.052)	0.100^{*}	(0.052)			-0.019	(0.063)
lnSAB	-0.135^{*}	(0.069)	-0.045	(0.069)			-0.029	(0.080)
lnVAB	0.105	(0.109)	0.059	(0.101)			-0.027	(0.117)
lnRPB	1.043^{***}	(0.054)	0.780^{***}	(0.109)			-0.361^{*}	(0.183)
lnGAB	-0.130	(0.093)	-0.146	(0.094)			0.090	(0.112)
lnTAB	0.071	(0.040)	0.028	(0.041)			0.037	(0.048)
lnPAY			0.453^{***}	(0.116)				
lnDAY			0.054	(0.037)	0.036	(0.043)	-0.003	(0.044)
lnSAY			-0.051	(0.037)	-0.065	(0.039)	-0.002	(0.044)
lnVAY			-0.104	(0.116)	0.239^{**}	(0.110)	0.259^{*}	(0.129)
lnRPY			-0.289^{*}	(0.158)	1.119^{***}	(0.048)	0.883^{***}	(0.130)
lnGAY			-0.038	(0.077)	-0.023	(0.090)	0.011	(0.090)
lnTAY			-0.052	(0.036)	0.080^{**}	(0.034)	0.097^{**}	(0.040)
R^2 adjusted	0.886		0.947		0.918		0.943	
F-statistic	99.4		94.6		130.0		94.6	
Observations	76		91		69		91	

The first observation of Table 7 is that the inter-SFE models perform somewhat better than the intra-SFE model, however the differences are only marginal as shown by the slightly higher adjusted R^2. This is because of the inclusion of more variables, which additional explain the variation in prices. The success level of intra-SFE regression model obtained for Bartın SFE is 89% (R^2=0.886). Here, lnRPB, lnDAB, lnSAB and lnTAB are statistically significant in the model. By including Yenice interactions as well, the adjusted R^2 increased to 0.947. Hence, 95% of the variation in timber price is significantly explained by lnDAB, lnRPB, lnPAY, and lnRPY variables, and 5% by other variables. Hence, the price in Bartın mainly depends on price mark-up in Bartın and the price in Yenice, while also the quantity offered and the price mark-up in Yenice play a role, albeit at a lower significance of 10%.

The success level of intra-SFE regression model obtained for Yenice SFE is 92% (R^2=0.918), where lnVAY, lnRPY, and lnTAY are statistically significant in the model. By including interactions with the Bartın SFE as well the R^2 increased somewhat to 0.943. Hence, the inter-SFE model explains 94% of the variation. In other words, lnPAB, lnRPB, lnVAY, lnRPY, and lnTAY variables in the model explained most of the variations in timber prices of Yenice. Accordingly, it can be said that timber prices in Yenice tend to rise, once the price mark-up rises, the average volume per log rises, the stack size increases. Concerning interaction with the Bartın SFE, the timber price and the price mark up play a role. Moreover, the same variables as under the intra-SFE regression stay significant in the inter-SFE regression in Yenice.

From the four regressions we can conclude that the rival prices and price mark-ups significantly explain the variation in prices. Additionally, the quantity offered during an auction significantly explains the prices in Bartın, while the log size and time between auctions significantly explains the prices in Yenice.

3.4. Determination of the Main Decision Variable for Beech Timber Auctions

A principal component analysis is applied on the ten variables defined for each enterprise in order to further interpret the dual correlation among variables by evaluating all variables simultaneously. With the principal component analysis we determine the most important factors affecting the dynamics in the timber auctions and to assist the enterprises to improve their marketing policies and strategies.

Data matrices with auctions (N) in rows and variables (n) in columns are the input for principal component analysis. *Principal component model* as extraction method and *Varimax criterion* with Kaiser normalization as factor rotation method are used in the principal component analysis.[10] The rotated factor matrixes obtained with the principal component analysis are shown in Table 88 and Table 99. The principal components (factors) of which the eigenvalues are larger than 1 (Kaiser criterion) are extracted in the principal component analysis for each SFE. In order to clearly see the variable groups, the factor loadings larger than 0.5 as an absolute value are shown in bold font in Table 88 and Table 99 assuming that this is the threshold value (Bennet and Bowers, 1977; Mucuk, 1978; Daşdemir, 1996).

[10] The application of factor analyses to forestry can also be found in related papers of interest, e.g., Daşdemir (1996, 2003), and Türker and Türker (1999).

Table 8. Results of Factor Analysis for Bartın SFE

Variables	F a c t o r s (Components)		
	Factor 1	**Factor 2**	**Factor 3**
PAB	**0.591**	0.021	**0.589**
DAB	0.124	**0.901**	0.153
SAB	-0.056	**0.938**	0.154
VAB	-0.139	0.077	**0.815**
EAB	**0.742**	-0.003	0.385
GAB	-0.183	**0.668**	-0.358
TAB	**0.672**	0.064	-0.111
CAB1	**0.642**	-0.287	-0.074
CAB2	**0.677**	0.015	0.002
% of variance	28.34	24.54	12.44
Total % of explained variance = **65.32**			

Table 9. Results of Factor Analysis for Yenice SFE

Variables	F a c t o r s (Components)			
	Factor 1	**Factor 2**	**Factor 3**	**Factor 4**
PAY	**0.763**	0.021	0.212	0.224
DAY	0.182	**0.898**	0.088	-0.065
SAY	-0.220	**0.838**	-0.208	0.112
VAY	0.105	0.085	-0.038	**0.877**
EAY	**0.868**	-0.059	0.034	0.180
GAY	-0.204	0.156	0.412	-0.416
TAY	**0.676**	-0.007	-0.289	-0.350
CAY1	**0.650**	0.030	**-0.511**	0.128
CAY2	0.084	-0.128	**0.835**	-0.011
% of variance	28.00	18.24	13.11	11.85
Total % of explained variance = **71.21**				

In Bartın SFE, according to the results of the principal component analysis (Table 88), 65% of total variance, has been explained by the first three components. The first component is the most important factor explaining 28%, while the other factors explain 25%, and 12% respectively.

The five dominating variables (PAB, EAB, TAB, CAB1, and CAB2; price, price mark-up, time between auctions, and time between auction in the rival SFE) constitute the first factor. This factor purely consists of price and time variables. Therefore, this factor is labelled as the ***price and timing of auctions***.

Factor 2 is of secondary importance and has three dominating variables, namely SAB, DAB and GAB, and has positive significant correlations among them. A typical characteristic of this factor is the supply and demand levels for timber during each auction and the stack size. Particularly, by considering that demand drives supply, Factor 2 is named and interpreted as the ***demand level of auctions***.

Factor 3 consists of two variables, namely VAB, but for the second time PAB. These variables are not correlated. This variable represents the average volume per log and, hence, this is a factor related to the standards of diameter and height of timber pieces, as well as theirs quality and stem shapes. Factor3 is named as the ***average volume per log***.

Subsequently, the weights and names of the factors that should be taken into consideration for improving marketing polices and decision making in Bartın SFE are summarized in Table 88. Hence, the most important factor in Bartın is the ***price and timing of auctions***.

In Yenice SFE, four factors are obtained with the principal component analysis (Table 99). 71% of total variance is explained by these four factors. The first component is the most important factor. The four factors respectively explain 28%, 18%, 13% and 12% of variance.

The first factor is nearly equal to the one in Bartın, except for the insignificant CAY2. Hence there are four dominating variables (PAY, EAY, TAY, and CAY1) in the first factor. As in Bartın, this factor purely consists of price and time variables. Therefore, this factor is labelled as the ***price and timing of auctions***.

Factor 2 is of secondary importance and has two dominating variables, namely SAY and DAY, and has positive significant correlations among them. In contrary to Bartın, GAY is not significant. A typical characteristic of this factor is the supply and demand levels for timber during each auction. Particularly, by considering that demand drives supply, Factor 2 is named and interpreted as the ***demand level of auctions***.

Factor 3 has two dominating variables, namely CAY1 and CAY2, which are both related to timing, where CAY1 has a negative sign. This shows that this factor represents a negative link to the previous auction in Bartın and a positive link the next auction in Bartın. We name the third factor as ***timing of the next auction in Bartın.***

Factor 4 consists of one variable, namely VAB. This variable represents the average volume per log and, hence, this is a factor related to the standards of diameter and height of timber pieces, as well as theirs quality and stem shapes. Factor 4 is named as the ***average volume per log***.

Subsequently, the weights and names of the factors that should be taken into consideration for improving marketing polices and decision making in Yenice SFE are summarized in Table 99. Hence, as in Bartın SFE the most important factor in Yenice is the ***price and timing of auctions***.

4. CONCLUSION

The General Directorate of Forestry and its 241 forest enterprises, working under imperfect competition to supply principal forest products, generally do not take demands of the population, customer satisfaction and criteria such as economic efficiency and profitability sufficiently into consideration. As a result, the state forest enterprises have been affected by the economic crises in the late 1990s and the beginning of 2000s in Turkey, because they could not raise the required incomes from sales, due to cost increases and threatened economic sustainability.

The sales price is important as it affects the total income of state forest enterprises. Insight into the factors affecting the sales price is useful for improving marketing polices. The factors affecting the timber prices and relationships between them are determined in this research, which is applied to the cases of Bartın and Yenice SFEs. These SFEs are the most important forest enterprises of the ZRFD, having a share of the market with 81% in the supply of beech timber in Turkey. Hence, this paper aimed at suggestions for improving marketing policies and strategies, based on the sustainability principle and customer satisfaction towards a sustainable management of forest resources.

Insight into seasonal and monthly variation in beech timber prices and time gaps between auctions is very important with regard to the preparation for production and stock keeping, providing a continuous cash-flow and necessary measures could be taken to improve marketability. The analyses in this paper indicates that the price is more or less stable, leading to satisfied customers, as sufficient and fresh goods are supplied to the markets in almost every season in the Bartın SFE. However, in the Yenice SFE, the beech timber prices have been fluctuating seasonally, because the supply of fresh wood could not always be achieved. Therefore, a sufficient supply of fresh goods to the markets by making a production (or harvesting) plan for demand in the Yenice SFE could avoid the negative effects of seasonal and monthly price fluctuations, leading to more stable prices and a higher level of customer satisfaction. On the other hand, the production and stock levels in both SFEs should be at a sufficient level in the spring when demand is particularly high.

The variables directly affecting the beech timber prices for Bartın SFE are: price mark-up, timber stack size and timing of Yenice's next auction in Bartın, and the timber price, demand, price mark-up, time between auction and the timing of Bartın's previous auction in Yenice. The variables directly affecting the beech timber prices for Yenice SFE are: price mark-up and timing of Bartın's previous auction in Yenice, and the price, price mark-up, stack size, and all timing variables in Bartın. According to this result, it is understood that when SFEs depended on the same Regional Forest Directorate, they are mutually affected by marketing activities and timber prices. Our analysis confirms this intuition where the timber prices of both SFEs explained the variation in the timber prices in the rival SFEs. Bartın and Yenice SFEs address the same customers. If the demand for one of them increases or one of them supplies more (less) timber for auction, the demand for other timber will be reduced (increased) and timber prices decrease (rise). Hence, the two SFEs compete with each other. We conclude that marketing activities should be planned in an integrated manner (by taking into account the auctions dates, the quantity and kind of supply and the reserve prices of the rival enterprises) for a successful management of the forest enterprises that are dependent on the same Regional Forest Directorate.

The principal component analysis is applied to the variables belonging to Bartın and Yenice SFEs. In the case of both SFEs, the factors constituting the main decision variables in auctions are determined as follows: (1) price and timing of the auctions, (2) demand level, and (3) average volume per log, while a factor is added in Yenice between (2) and (3), namely the timing of the next auction in Bartın.

Consequently, the beech timber prices in the case of both SFEs are affected by factors that are not easily controlled by the enterprise management, such as the demand level and the structure of the market, seasonal fluctuations, supply features, timber prices of the rival enterprises, as well as factors that are partially controllable by the enterprise management, such as the size of timber stacks, average volume per log, time between two auctions, quantity and quality of supply. Hence, goods suitable for sales (high in quality, freshness and suitable standards) need to be produced, the quantity of supply and sizes of timber stacks in auctions need to be adjusted for demand, the marketing activities need to be planned in an integrated manner (taking into consideration the auctions dates, the quantity and kind of supply and the reserve prices of the rival enterprises, customer satisfaction, e.g.) and the marketing decisions need to be made according to the joint interests of both SFEs. Therefore, by taking these factors into account, not only the economic sustainability of the enterprises will be improved because of stable timber prices, but also the contemporary marketing policies and strategies based on the consumer satisfaction will be put in practice.

REFERENCES

Acun, E. (1977). *Research on the Marketing of Principal Forest Products by State Forest Enterprises in Turkey*. Istanbul: I.U. Publication of the Faculty of Forestry, No: 236.

Atmiş, E., Daşdemir, İ., Lise, W., & Yıldıran, Ö. (2007a). Factors affecting women's participation in forestry in Turkey. *Ecological Economics*, *60(4)*: 787–796.

Atmiş, E., Özden, S., & Lise, W. (2007b). Public participation in forestry in Turkey. *Ecological Economics, 62(2)*: 352–359.

Bennet, S., & Bowers, D. (1977). *An Introduction to Multivariate Techniques for Social and Behavioural Science*. London: The MacMillan Press.

Brooks, D. J. (1997). Demand for wood and forest products: Macroeconomic and management issues. XI. World Forestry Congress, Vol.4, Antalya-Turkey. pp.66-75.

Bulow, J. I., & Klemperer, P. D. (1996). Auctions vs. Negotiations. *American Economic Review, 86*: 180-94.

Bulow, J. I., & Roberts, D. J. (1989). The simple economics of optimal auctions. *Journal of Political Economy, 97*: 1060-90.

Bulow, J. I., Geanakoplos, J. D., & Klemperer, P. D. (1985). Multimarket oligopoly: strategic substitutes and complements. *Journal of Political Economy, 93(3)*: 488-511.

Cillov, H. (1993). *Statistical Methods applied Economical Issues*. Istanbul: I.U. Publication of the Faculty of Economics, No:3801/545.

Çağlar, Y. (1989). *Cost Issue in the Principal Forest Products in Turkey*. Ankara: MPM Publication No:374.

Daşdemir, İ. (1996). *Determination of Success Levels in State Forest Enterprises (Example of North-East Anatolia and Eastern Blacksea Regions)*. Erzurum: Eastern Anatolia FRI, Technical Bulletin No: 1.

Daşdemir, İ. (2003). *Analysis of the Price in the Principal Forest Products (Zonguldak Regional Forest Directorate case study)*. Bartın: Z.K.U. Publication of the Bartın Faculty of Forestry, No: 26/12.

Dinler, Z. (2003). *Introduction to Economics (9th ed.)*. Bursa: Ekin Publisher.

DPT (State Planning Organization of Turkey) (2001). 8th Five Year Development Plan. Ankara: Report of Forestry Special Commission.

Duerr, W. A. (1993). *Introduction to Forest Resource Economics*. New York: McGraw-Hill.

Duncan, A. J. (1974). *Quality Control and Industrial Statistics (4th ed.)*. Illinois: Richard D. Irwin, Inc., Homewood.

Geray, A. U. (1998). *Economics*. Istanbul: I.U. Publication of the Faculty of Forestry, No: 3870/430.

Harman, H. H. (1967). *Modern Factor Analysis (2nd Rev. Ed.)*. Chicago: The University of Chicago Press.

Juslin, H., & Lintu, L. (1997). Responses to changes in demand and supply of forest products through improved marketing. XI. World Forestry Congress, Vol.4, Antalya-Turkey. pp.89-108.

Kalıpsız, A. (1988). *Statistical Methods*. Istanbul: I.U. Publication of the Faculty of Forestry, No: 3522/394.

Klemperer, P. D. (1995). Competition when consumers have switching costs: An overview with applications to industrial organization, macroeconomics, and international trade. *Review of Economic Studies, 62*: 515-39.

Klemperer, P. D. (1999). *The Economic Theory of Auctions*. Chelthenham: Edward Elgar.

Klemperer, W. D. (1996). *Forest Resource Economics and Finance*. Singapore: McGraw-Hill, Inc.

Kotler, P. (Trans.: Yaman Erdal) (1972). *Marketing Management, Analysis, Planning and Control. Volume 2, Second Edition*. Ankara: Society of Scientific Publications.

Lyke, J., & Brooks, D. J. (1995). World supply and demand for forest products. *Journal of Forestry, 93 (10)*: 22-26.

Maskin, E. S., & Riley, J. G. (1985). Auction theory with private values. *American Economics Review, 75*: 150-55.

McAfee, R. P., & McMillan, J. (1987). Auctions and bidding. *Journal of Economic Literature, 25*: 699-738.

Mucuk, İ. (1978). *Factor Analysis as a Modern Research Technique in Firm*. Istanbul: I.U. Faculty of Economics (Assoc. Prof. thesis).

Mucuk, İ. (1984). *Marketing Principles (Second Edition)*. Istanbul: Der Publications.

Nautiyal, J. C. (1988). *Forest Economics Principles and Applications*. Toronto: Canadian Scholars' Press Inc.

Rosenthal, R. W. (1980). A model in which an increase in the number of sellers leads to a higher price. *Econometrica, 48*: 1575-79.

Sokal, R. R., & Rohlf, J. F. (1995). *Biometry. 3rd ed*. New York: W.H. Freeman and Company.

Ok, K. (1997). Interaction between sales by auctions of state forest enterprises. Review of the Eastern Mediterranean FRI (DAO), Number 3, Tarsus-Turkey. pp.39-62.

Ok, K. (1998). The seasonal effect on the price of timber sale by auctions. I.U. Review of the Faculty of Forestry, Series A, Volume 48, Number 2, İstanbul. p.21.

Palo, M., Uusivuori, J., & Mery, G. (2001). *World Forests, Markets and Policies: Towards a Balance*. Dordrecht: Kluwer Academic Publishers.

Türkay, O. (1997). *Introduction to Economic Theory: Microeconomics (11th ed.)*. Ankara: İmaj Publisher.

Türker, M. F. (1996). Investigation of the Pricing Policy on the Sales of Auction Forest Products (Timber) (Eastern Blacksea Region case study). Project number with 93.115.002.1 of Research Found of K.T.U. Trabzon.

Türker, M. F., & Türker, E. S. (1999). Economic analysis of forest enterprises by the use of the multivariate statistical analysis (a sample of 25 State Forest Enterprises in the Eastern Black Sea Region). *Turkish Journal of Agriculture and Forestry, 23: Supplement 1*, 169-177.

Türker, M. F., & Yazıcı, K. (1997). Investigating some of the factors of auction sale affecting the economic success of the state forest enterprise (Maçka case study). K.T.U. Publication of the Faculty of Forestry, Number 4, Trabzon. pp.24-36.

Vickrey, W. (1961). Counterspeculation, auctions, and competitive sealed tenders. *Journal of Finance*, *16*:8-37.

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Chapter 12

ASSESSING SUSTAINABLE WELL-BEING: TRENDS IN ENVIRONMENTAL, SOCIAL AND ECONOMIC POLICY AT THE LOCAL LEVEL

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ABSTRACT

The Index of Sustainable Economic Welfare (ISEW), introduced by H. Daly and J. Cobb in 1989, is an ecological economic instrument that was created in order to integrate the information embodied in GDP. Actually, GDP has been deeply criticizing for long time as an indicator of welfare: some corrections and adjustments are hence necessary in order to consider those environmental and social aspects that are relevant for human life, either ignored or wrongly treated in the official estimates of GDP. The ISEW was already calculated for several national economies but rarely for a region. The aim of this paper is to show the ISEW calculation and the results for a local economy, the Province of Pescara in Italy. This case-study is one of the first time series analyses of the ISEW (1971-2003) for a local (sub-national) territorial system. The ISEW is a tool that provides a thorough representation of the local socio-economic organization, that is important in those cases of administrative decentralization, autonomy and responsibility at the local level. For this reason, public authorities need a more and more comprehensive knowledge of the characteristics and peculiarities of the territorial system they manage. The maintenance of identity for a local system constitutes a unique resource to be emphasized, because the diversities among different sub-national areas are a prerequisite of sustainability at the national level. The results show a stagnation of the ISEW after the 1980s, compared with a constant increase of GDP during the period 1971-2003.

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Keywords: ISEW, sustainability, ecological economics, time series analysis, local level, well-being

INTRODUCTION

The evaluation of human activities is often based on economic parameters such as consumption, income, investments, employment, etc. However, it is well-known that these tools do not provide a wide and comprehensive view of the complexity of anthropic systems, since economic indicators neglect some social and environmental aspects of the whole system.

Conventional economic reports are rather narrow. The core of a market economy is represented by the value of final goods and services that are produced and exchanged for money in a given lapse of time, and its aim is the maximization of this value, with the assumption that the more activity, the better off we are. Growth is thus assumed to be the primary policy of all institutions and also something that is sustainable indefinitely. Real life and its dynamics contravene this tendency [1].

In a finite planet and limited by its carrying capacity, growth pattern is not feasible. Human behavior must adapt to the ability of the environment to produce natural resources. In other words, sustainable development must not endanger the natural systems that support life on Earth, as well as atmosphere, waters, soils, and living beings. The concept of sustainability was operationalized by Herman Daly [2]: “For the management of renewable resources there are two obvious principles of sustainable development. First that harvest rates should equal regeneration rates (sustained yield). Second that waste emission rates should equal the natural assimilative capacities of the ecosystems into which the wastes are emitted. Regenerative and assimilative capacities must be treated as natural capital, and failure to maintain these capacities must be treated as capital consumption, and therefore not sustainable. There remains the category of nonrenewable resources which strictly speaking cannot be maintained intact short of nonuse (and if they are never to be used then there is no need to maintain them for the future!). Yet it is possible to exploit nonrenewables in a quasi-sustainable manner by limiting their rate of depletion to the rate of creation of renewable substitute”.

Actually, the concept of sustainability is still inconsistently addressed by public institutions as well as private subjects, due to the priority of other goals, especially socioeconomic, and to the difference in temporal horizon between economic and political cycles (medium–short run) and the environmental problems (long run). Hence, time becomes a focal point in the practical implementation of such principles, and the role of natural capital becomes crucial for life on Earth and, consequently, for sustainable human activities. Cyclically from year to year, the Earth produces a great quantity of environmental resources and services that humans can use. For example, the water cycle supplies domestic, agricultural and industrial uses. Food production, both plants and animals, depends on nature’s services. Forest production is an important part of the carbon cycle. Humans may draw on natural capital, namely, the set of elements and mechanisms that through natural laws, transform primary materials and solar energy into flows of products and services. At current rates of consumption of resources and production of wastes and emissions, it has been calculated that this flow of products is insufficient to satisfy annual human needs. Every year, humans use up the year's supply of natural resources and consume their capital. Greenhouse gases build up in the atmosphere, wastes and contaminants accumulate, resources become depleted and minerals are increasingly expensive to extract. The mechanisms of the machine called Nature grind painfully and its cogs are losing some teeth. The capacity of the environment to provide the same quantity of goods and services as it did the previous year is threatened. Future generations will pay the price in natural capital loss and lost natural production.

Various authors express the need of a system of indicators able to represent the complex world of economic and social dynamics in their relation to the environment. Despite much criticism, most economists continue to use the concept of economic growth as an unique aim. According to Christian Leipert, an obsolete concept of economic growth reigns: it counts the consumption of natural resources and degradation of the environment positively as income. The prevailing concept of growth produces a big illusion of growth and welfare. Politically speaking, the policy of pursuing the highest possible economic growth is discredited by the exposure of defensive expenditures. Under the present conditions of economic development, in which productive as well as destructive forces are at work, it can no longer be taken for granted that economic growth is equated with an increase in welfare. What is needed is a policy of differentiated development focusing on addressing the structural causes of environmentally damaging and costly patterns of production and consumption. Economics has long neglected the difference between wants and needs. Needs are finite, unlike wants which emerge continually in new forms and are therefore insatiable. Economics should address the political-ethical question of balancing the legitimacy of the demands of the living and of future generations for consumption/depletion of natural resources [3].

Although these concepts are essential for future development of ecological and socio-economic systems, they have not been implemented significantly within the ensemble of traditional economic tools. The main tool that is used in macroeconomics is Gross Domestic Product (GDP) that gives a measure of economic activity. It is defined as the annual market value of final goods and services purchased in a nation, plus all exports net of imports. However, as Daly said, GDP does not subtract either depreciation of man-made capital (such as roads and factories) or depletion of natural capital (such as fish and fossil fuels). GDP also counts so-called defensive expenditures in the plus column [4].

In order to give a more comprehensive measure of human activities, and take into account social and environmental aspects that are essential for human life (for example the level of welfare), some adjustments are proposed to the traditional framework of National Account. The first idea of a possible revision of GDP as a measure of welfare was proposed by Nordhaus and Tobin [5], who introduced the so-called Measure of Economic Welfare (MEW). In 1981, Xenophon Zolotas, proposed the EAW - Economic Aspects of Welfare, introducing pollution costs and depletion of resources, and he noted a decoupling between growth and welfare beyond a certain level of GDP. [6].

In 1989, Daly and Cobb [7] proposed further revisions and introduced the Index of Sustainable Economic Welfare (ISEW). Starting with consumption, they proposed some adjustments to allow for inequality of income distribution, environmental problems (such as pollution costs, long term environmental damage, depletion of non-renewable resources) and social issues (such as commuting costs, urbanization costs, public expenditure for health and education). Computations for different nations have shown that the ISEW increases together with GDP up to a point, beyond which it stagnates or even decreases, due to the environmental and social pressures of economic growth [8,9,10,11,12,13,14].

This paper shows the results from a time series calculation of the ISEW (1971-2003) for the Province of Pescara, a southern Italian province. This is one of the first attempts to perform such a calculation in time series at the local level. Creating awareness at the local level is important. The local (regional or provincial) level often represents the optimal dimension for implementing policies, especially in certain fields. This is mainly due to a major attention paid by local institutions for peculiar problems and to the administrative decentralization. For this reason, the ISEW has been calculated at the local level, in order to give local authorities further information for taking proper political decisions under a sustainable viewpoint.

METHODOLOGICAL BACKGROUNDS

The System: The Province of Pescara - Italy

The Province of Pescara was founded in the 1920s, close to the Pescara river. The territorial system (about km^2 1225) reaches out from the Adriatic sea to the Appenine mountains, presenting a great variety of places and urban settlements.

It is the smallest province in the Abruzzi region but the most populated one. Its capital, the city of Pescara, has a population density of about 3500 inhabitants per km^2. It is a large urban area, a fishing center of great importance and one of the major tourist and business centers on the Adriatic Sea.

The structure of production in the province reflects the transition of the economy from agriculture to industry and services. Agriculture, involving small farms, is succeeding in modernizing and offering high-quality products. Industry is consolidated in few fields but, in general, it is not so developed and dynamic. Anyway, relevant number of small and medium-sized factories characterizes the area. The most important fields are pharmaceutics, biomedicine, electronics, aerospace and nuclear physics.

A further activity, worthy of note, is seaside and mountain tourism, which is of considerable importance to the economy of the province of Pescara.

The Isew Method Item by Item

The ISEW calculation is divided into items (Table 1). Item A is the reference year; this analysis was performed in time series, from 1971 to 2003. Items B and D are row and adjusted consumption, respectively. The latter is calculated on the basis of the Gini index of income distribution (item C). Items E, F, G and H are positive values, corresponding to services that contribute to welfare, though not considered in conventional national accounting. Items I to Q are negative because they correct the overestimation of economic welfare with respect to the private level of consumption. Items R, S, T and U are usually negative since they are estimates of the consumption of structural fractions of Natural Capital without any real counterpart in terms of welfare.

Table 1. The Items of The Index of Sustainable Economic Welfare

	Sign	ISEW components	Reasons for inclusion/exclusion
A		Year	1971-2003
B		Personal Consumption Expenditure	The basic variable directly affecting economic welfare
C		Index of inequality distribution	Connected to social aspects, it is necessary to have a more realistic starting point
D		Weighted Personal Consumption	The basis to which all other positive and negative items are added or subtracted
E	+	Services of households' labour	An activity outside the market process that however leads to improvements in welfare
F	+	Durable goods services	Services connected with goods are benefits
G	+	Services from Public Infrastructure	In general people use infrastructures without paying any costs but in reality lots of money are spent for citizens' welfare
H	+	Public expenditure on health and education	It is only a portion, the non-defensive one, of public spending that is expected to affect welfare
I	-	Expenditures on durable goods	They must be subtracted because they are not strictly connected to welfare, but only the services from them
J	-	Defensive private expenditures on health and education	The total value is already present in B or D. Here the defensive portion, not affecting welfare, is removed
K	-	Local advertising expenditure	Not computed
L	-	Cost of commuting	It is included in consumption but it is due to traffic and overcrowding. Then it is not related to welfare
M	-	Cost of urbanization	Not computed
N	-	Cost of car accidents	Defensive costs, not related to welfare, that have to be removed from consumption
O, P,Q	-	Cost of water pollution Cost of air pollution Cost of noise pollution	These costs represent the contribution a society must pay to solve environmental problems. They are not related to welfare because defensive.
R	-	Loss of wetlands	Wetlands are important natural habitats, whose protection is one of the keys of sustainability
S	-	Loss of agricultural land	Agricultural land is threatened by urbanization and soil erosion, involving costs that must take into consideration
T	-	Depletion of non renewable resources	The consumption of non renewable natural capital, undermine the possibility for future generation to meet their needs
U	-	Long-term environmental damage	Long term damages caused by greenhouse gas emissions.
V	+	Net capital growth	To ensure an economic sustainability, the level of capital per worker must be maintained.
W		ISEW	The final value: it will be compared with the GDP trend.

The following sections are devoted to the description of the whole procedure, indicating the most important local data sources that prevented from scaling down the national data to the local level, thus avoiding arbitrary estimates. All monetary values were expressed in Euro in 2003.

The whole calculation procedure is shown in the report “Analisi di sostenibilità della Provincia di Pescara”, available at the Environmental Office of the Province of Pescara.

Private consumption - B, D

Private Consumption (B) is the primary variable directly affecting economic welfare because large household expenditures in goods and services are considered to be an indicator of a wealthy economy and a healthy society.

Data for this variable were obtained from the statistical reports for the Province of Pescara provided by the National Statistic Bureau (ISTAT), [15-20].

Private Consumption is adjusted according to the index of income distribution. The adjusted private consumption (D) is the basis to which all other positive and negative items are added or subtracted.

The index of income distribution - C

In general, private consumption it is not representative of the level of economic welfare, because, in general, an additional income of a thousand dollars is more beneficial to the welfare of a poor family than a rich one. Daly and Cobb [21] proposed the application an index of income distribution to adjust the level of private consumption: the Gini’s index. The range of variation of this index is between 0 and 1 (where 0 means perfect income distribution and 1 means maximum inequality). The value of Gini index was computed from data in Cannari and D’Alessio [22] and from Brandolini [23].

Services and domestic labour - E

Domestic labour (E) for cooking, cleaning, child care etc., contributes directly to economic welfare, even if it does not imply money exchange. The number of housewives, unemployed people and students involved in domestic labour was obtained from ISTAT. It was assumed that a housewife spends 8 h/day in housework, an unemployed person 4 h/day and a student 2 h/day. The income per hour attributed to domestic labour (data of market wages was obtained by ISTAT) [24] was multiplied by the hours spent at home by people over 14 years of age, as suggested by Guenno and Tiezzi [12].

Services from durable goods – F

Expenditure on durable goods such as cars and household appliances does not reflect the real welfare of consumers, because it is necessary to consider the utilization period of these goods. Hence, services (item F) from these goods are considered as benefits, while the initial capital (item I) is a cost that must be subtracted from private consumption. Furthermore, domestic appliances tend to wear out faster than they should and this causes an increase in private consumption that does not really contribute to economic welfare. According to Daly and Cobb [21], services are only 10% of the total stock (5% for houses).

The services from a variety of goods (houses, household appliances, personal computers, mobile phones and cars) were calculated on the basis of data from the Bank of Italy [25], Pulselli et al., [26], and from Chamber of Commerce [27].

Services from public infrastructure - G

Daly and Cobb [21] consider that public costs should not be a component of economic welfare because they are part of defensive costs, with the exception of services from public infrastructure (item G) and health and education costs (item H). Indeed, the growth of administration costs does not contribute to net economic welfare; it keeps economic welfare from declining, maintaining a healthy environment and conditions which contribute to trade and commerce.

In general, people use infrastructure without paying a direct monetary contribution. This item is the sum of the value of services of the road system (equal to the cost of their maintenance) and the value of current public expenses in urban development, water distribution, urban health, without which these services would be unavailable. All data are collected from personal communications from Agency of Land development and Viability of the Province of Pescara.

Public health care and education cost - H

Public health care and education costs are included in GDP because they are part of public expenses. However, it is not easy to link an increase in public expenditure to an increase in economic welfare, because of the inherent difficulty of measuring the demand for the types of services offered by the public administration.

Daly and Cobb [21] consider that 50% of this expenditure (both health care and education) is a defensive cost and should not be added to the index calculations, while Guenno and Tiezzi [12] believe that only 50% of health care costs are defensive. Hence, 100% of the public education costs and 50% of health care costs are added. Data are collected from ISTAT and from Ministry of Education and University [28-31].

Costs of durable goods - I

The consumption expenditure on durable goods (televisions, personal computers, cars) does not represent the welfare derived from their use. These goods provides utility (and thus welfare) along their life time. This figure must be subtracted from consumption. All data are collected thanks to ISTAT [15-20].

Private defensive expenditure for education and health care - J

Private defensive expenditure for education and health care is included in consumption but only a fraction of it is non-defensive and correlated to welfare. The defensive fraction (the 50% of both) must be subtracted from private consumption. Information is collected from ISTAT [15-20].

Local advertising cost - K

National advertising costs are mostly aimed at stimulating or maintaining the demand for a certain product, without a real counterpart in terms of collective welfare; on the contrary, at local level, advertising also plays a social rule by broadcasting information, hence a portion of local advertising costs should be added [21]. However, since no data on local advertising costs were available, this item was omitted.

Cost of commuting - L

Like Daly and Cobb [21] and Guenno and Tiezzi [12], we consider that the 30% of the costs related to private cars and public transport, together with the 30% of public and private maintenance cost, is not related to welfare because it depends on urbanization, overcrowding and traffic. Data comes from Automobile Club Italia (ACI) [32] and they are directly related to commuting costs, according to the following formula:

$$L = 0.3\alpha + 0.3\beta + 0.3\gamma$$

where:

Item L is the cost of commuting; is the total value of cars in the area; is the expenditure for public transport tickets; is the cost for public and private vehicle maintenance; 0.3 is the portion of services related to commuting (e.g. the time we are forced to spend because of traffic).

Urbanization cost - M

In general, growing population density in urban areas implies that land and house prices and rents rise without a compensating increase in economic welfare. Buying a house provides a high level of satisfaction and the monetary value of the investment tends to be maintained with rare contingent exceptions. The concentration of people in urban areas also stimulates an increase in the supply of houses that partially offsets the increase in prices due to the overcrowding and is regulated by multiyear urban plans approved by local authorities.

Hence this figure has not been subtracted from private consumption.

Cost of car accidents - N

This cost is a negative item because it corresponds to a decrease in economic welfare.

The cost of car accidents was calculated from total payments of insurance premiums. We used the method proposed by Guenno and Tiezzi [12]. Associazione Nazionale Imprese Assicuratrici [33-34] provided data.

Cost of water pollution - O

As in the case of Italy [12], we used total costs for water purification. Water quality is usually determined on the basis of parameters such as BOD (biological oxygen demand) and COD (chemical oxygen demand) and the cost of pollution is calculated on the basis of the cost of abatement of organic pollution. In this paper an estimate of the total cost necessary to purify water is obtained from a standard purification plant. The cost is referred to the equivalent inhabitants (E.I.) of the area, as the pollution measure. The number of E.I. of the Province of Pescara was obtained by summing resident population, E.I. of industrial sector, and E.I. of agricultural and zootechnic sector, and they are collected thanks to the regional environmental agency (ARTA) [35].

Cost of air pollution - P

Daly and Cobb [21] divided this cost into 6 categories: 1) damage to agricultural production; 2) material damage; 3) cost of cleaning implement; 4) damage caused by acid rain; 5) urban degradation; 6) damage to buildings and surroundings. Like Guenno and Tiezzi (1998) [12], we considered types of emissions and their cost per ton of emission abatement (SO_x: 2324 Euro/t; NO_x: 904 Euro/t; TSP (total suspended particles): 130 Euro/t; CO_2: 10 Euro/t).

Specific cost is multiplied by the quantity of emissions calculated following the 1996 Guidelines of Greenhouse Gas National Inventories [36].

Cost of noise pollution - Q

In calculating this item, we consider noise pollution due to vehicular traffic. The environmental agency of Abruzzo Region (ARTA) provided an estimation of the value of noise pollution attributed to cars.

Loss of wetlands - R

Wetlands host some of the most biologically productive habitats in the world. Their value has not been included in economic accounting because they are considered part of natural capital and difficult to monetize. The ISEW addresses this issue by estimating the value of services lost when wetlands are converted into other uses. In the Province of Pescara there was an increase in wetlands in the last years. The information was obtained at the website: http://www.riserveabruzzo.it/, and by personal communications with the Regional Agency of Land, Environment, Energy, and Parks.

Loss of agricultural land - S

Agricultural land productivity is fundamental for every society and has been progressively reducing for a long time due to two destructive processes. Urban expansion and bad land management (allowing erosion, intensive agricultural practices, decomposition of organic material) have led to the depletion of agricultural land and a reduction in yields.

Data is collected by ISTAT [37-40]. The result represents the definitive loss of available bio-productive land due to the change in use. The monetary value of agricultural land was obtained from the Provincial Office of Agriculture (personal communication).

Depletion of non-renewable resources - T

According to El Serafy [41], a portion of the economic profits of resource extraction should be reinvested to preserve the capacity of the economic system to produce a durable income for future generations.

El Serafy's formula is:

$$R - X = R \cdot \left(\frac{1}{(1+r)^{n+1}} \right)$$

where:

X annual income;
R revenue from extraction net of extraction costs;
r discount rate;
n residual life-time of the stock of resources.

Since discounting the utility of future generations is morally unacceptable, Daly and Cobb [21] assumed a discount rate of 0 that implies X=0. Therefore, the value of total net returns from the sale of non-renewable resources is counted as depreciation.

Long-term environmental damage - U

A fundamental element that contributes to well-being in the long run is the conservation and protection of natural ecosystems which represent a source of biological production. Modern economic infrastructure, with its industrial and commercial practices tends to ignore physical rules and the inability of ecosystems to support these practices indefinitely. The production of toxic wastes, carbon-dioxide, nuclear wastes and chlorofluorocarbons, with their durable deleterious effects, is a real cost which will fall on future generations.

Long-term environmental damage is directly proportional to the consumption of fossil fuels and energy. Hence, petrol, diesel fuel, fuel oil, methane and electricity consumption time series were considered and converted in oil barrel. Daly and Cobb [7] used a tax of 0.50$ in the 1972 per oil barrel (equal to 1.96 Euro in the 2003), in order to determine the amount of money that must be accumulated year after year and maintained for compensating future generations.

Net capital growth - V

In order to sustain long-term economic welfare, there should be an increasing or constant supply of capital per worker. ISEW calculates net capital growth (NCG) (item V) by adding the stock of new capital (ΔK) and subtracting the capital requirement (CR). CR is obtained multiplying the percentage variation in labour force (ΔL/L) by the stock of capital of the preceding year (K−1) [12].

$$NCG = \Delta K - CR$$

where $\Delta K = K - K_{-1}$ and $CR = (\Delta L/L)K_{-1}$

Data were collected from ISTAT [42]

The index of sustainable economic welfare and local GDP - W

The ISEW is obtained by the algebraic sum of all the items, depending on their positive or negative contribution to welfare, as previously indicated. The main features of ISEW, as a measure of sustainable economic welfare, with respect to GDP, are:

- Inclusion of changes in the distribution of income, reflecting the fact that an additional Euro means more to the poor than to the rich;
- Inclusion of household labour;
- Exclusion of expenses to offset social and environmental costs (defensive expenditure)
- Inclusion of long-term environmental damage and depreciation of natural capital;
- Inclusion of net production of man-made capital (i.e. net investment).

The method used in this paper is largely in line with that of Guenno and Tiezzi [12] with some exceptions: public maintenance costs of urban development, water distribution and urban health were added to private consumption; the urbanization costs, as described by Daly and Cobb [21], were not subtracted, because they are directly related to welfare; the cost of water pollution was calculated on the basis of the total purification costs of water (instead of an estimate based on a parameter such as biological oxygen demand, BOD); loss of agricultural land was computed by comparing the results of all censuses within the time series analysis to have a picture of irreversible changes in land use.

RESULTS AND DISCUSSION

Private Consumption and Adjusted Private Consumption

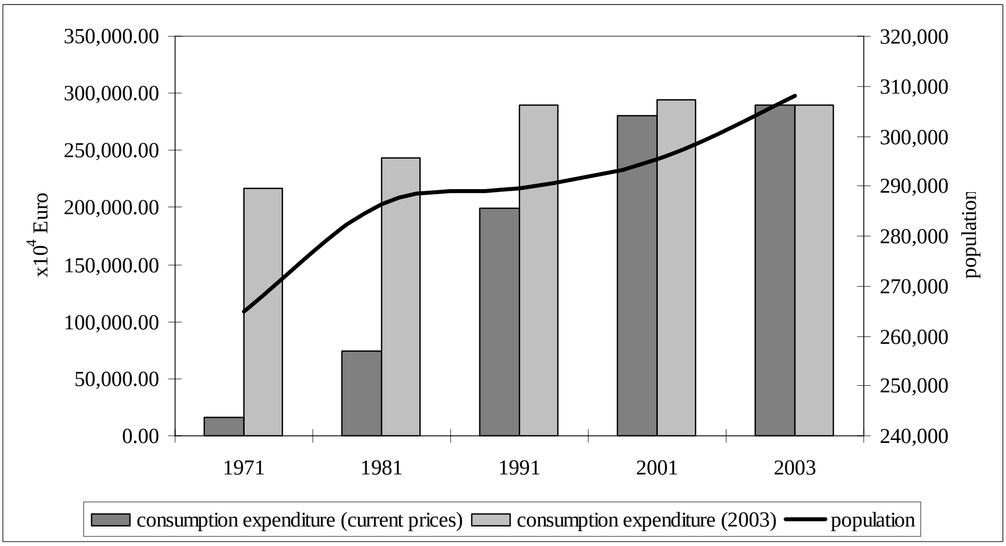

Figure 1. Consumption expenditure and population in the Province of Pescara.

Table 2. Population And Consumption Expenditure In The Province Of Pescara

	Population	Consumption expenditure (current prices)	Consumption expenditure (2003)
			B
1971	264,981	157,429,117	1,237,835,760
1981	286,240	741,841,342	2,428,065,624
1991	289,534	1,990,223,447	2,891,599,069
2001	295,481	2,804,139,092	2,942,943,977
2003	307,974	2,897,395,324	2,897,395,324

Figure 1 and table 2 show that consumption expenditure (b) grows fast: in the period 1971-1991, there is an increase in consumption of about 24%, while in the period 1991-2003 private consumption is almost constant.

In table 3, we can see the trend of adjusted private consumption (D), corrected by the Gini Index, as explained below.

Table 3. Row and Adjusted Consumption Expenditure and Gini Index

	Consumption expenditure (2003)	Gini Index	Adjusted consumption expenditure (2003)
	B	**C**	**D**
1971	1,237,835,760	0.362	908,836,828
1981	2,428,065,624	0.282	1,893,966,945
1991	2,891,599,069	0.287	2,246,774,724
2001	2,942,943,977	0.322	2,226,130,088
2003	2,897,395,324	0.305	2,220,226,302

Gini Index

It is not easy to obtain information about the Gini Index (C) at a provincial level. For this reason, it was necessary to look for figures at the regional or national level.

The values of the Index are shown in table 3. The value tends to diminish, varying from 0.362 in the 1970s to 0.305 in 2003. It corresponds to better conditions in income distribution among the population of the area.

These results seem to be quite homogeneous in southern of Italy. These systems traditionally present very high level of poverty and unemployment, partially counterbalanced by local public policies.

Domestic Labour, Durable Goods and Net Capital Growth

Domestic labour (E) contributes to the welfare of society, but it is not considered within the GDP framework. In the Province of Pescara, its monetary value slowly decreases. This is mainly due to the increasing number of women regularly employed.

The category of durable goods (I and F) includes cars, household appliances and houses (the latter have been considered only in terms of services). The need for a higher standard of living for the residents is the crucial point represented by the trend of this item. In fact, the value of durable goods rise constantly between 1971 and 2003. Cars represent the best and cheaper way to move in the urban systems, and home appliances are considered essential for all consumers. Moreover an increasing consumerism is causing an exponential accumulation of things in every house. For example, the 41% of total population has got a car in 2003 (only 24% in 1971). Italian building industry and trade does not know crises: in particular, in the Province of Pescara, the average price of houses increases constantly.

he net capital growth (V) is based on the trend of net capital and total investments in the industry sector at national level. Table 4 shows the decreasing trend of the net capital growth in the Province of Pescara. It reveals the structural difficulty of southern Italy to develop industry and manufacturing activities.

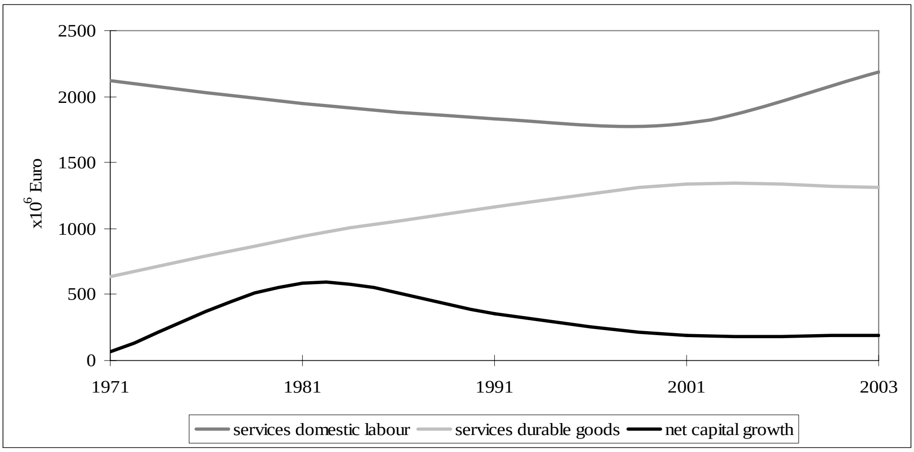

Figure 2. Services of domestic labour and durable goods, and the net capital growth.

Health Care and Education Public and Private Expenditure (H And J)

Both public and private expenditure for education tends to decrease due to the low birth rate of the last decades. The case of health care expenditure is more complex. Both public and private expense tends to increase, especially in the last years (Figure 3). This is due to two main reasons: environmental degradation and pollution may be an important source of disease (this emphasizes the defensive nature of these costs); a new and increasing demand of health (together with medicines, drugs and remedies) is artificially induced by the dynamics of market.

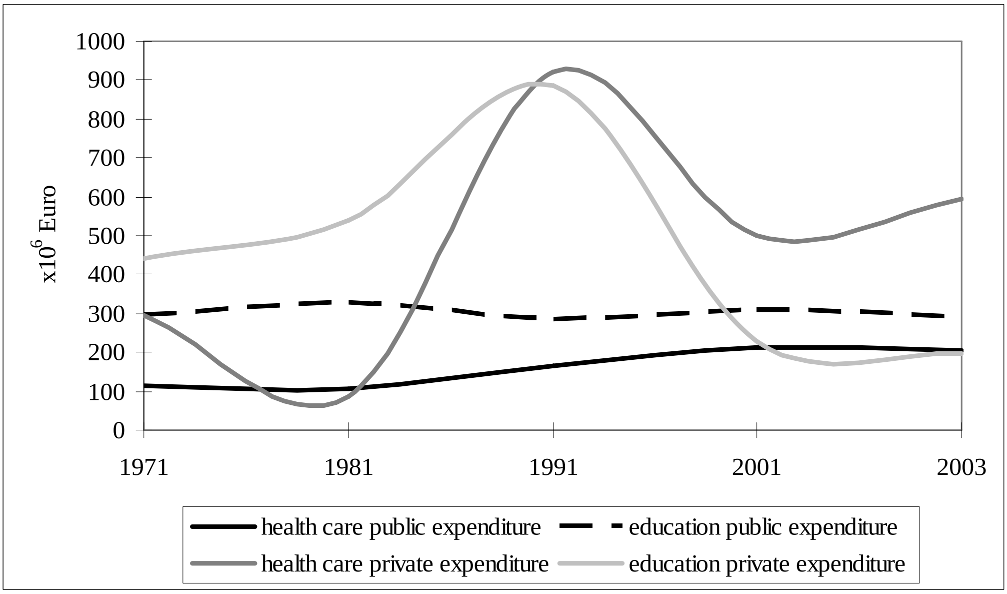

Figure 3. Health care and education public and private expenditures.

Environmental Items

Environmental items are water, air and noise pollution, exhaustible resources depreciation and long-term environmental damage.

Water pollution is calculated on the basis of a purification cost [26] of organic load coming mainly from industrial sector.

Air pollution depends especially on fossil fuel consumption. The evaluation of emissions related to the Province of Pescara follows the International Panel of Climate Change framework [36]. Air quality inside cities is the major focal point of environmental issues in Italy. It mainly depends on vehicular traffic emissions, especially due to private transport.

The value of noise pollution is estimated on the basis of the abatement cost of noise produced by cars and trucks. This item shows the increasing influence , occurred in the last decade, of traffic and urban congestion (see also the cost of commuting – item L) on the level of welfare.

The long-term environmental damage shows the increasing reliance of our society, and our productive processes on fossil fuels. The use of them causes the emission of greenhouse gases whose lifetime in atmosphere is often very long. The estimation of this item requires data on fuels consumption from 1971-2003. All data are converted in oil barrel, and a tax equal to 1.96 € per barrel (corresponding to 0.50$ in 1972, as Daly and Cobb proposed [7]) is applied. Tax receipts must be accumulated year after year in such a way that future generations will be refunded for the damage we are causing them today. In fact, future generations will have to face a depletion of the stock of fossil fuels as well as bad air quality and worse environmental conditions, especially due to greenhouse effect and climate change.

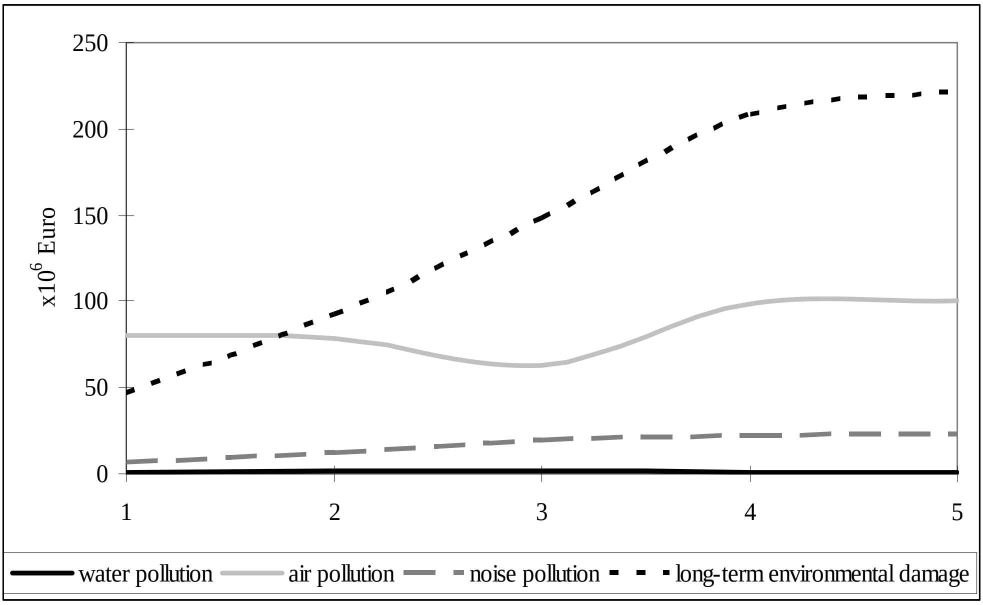

Figure 4. Trends of environmental items in the Province of Pescara.

Another important environmental item is the exhaustible resources depreciation (T). It is represented by the amount of non-renewable resources extracted from soil, such as gravel, marble, sand, etc. A complete lack of information in this field do not allow us to evaluate the monetary value of this important item.

Figure 4 shows the trend of environmental items for the Province of Pescara.

The Comparison between GDP And ISEW

The time series analysis of ISEW and GDP enables us to draw a picture of the evolution of the socio-economic system of the Province of Pescara during the last 30 years. The joint use of the instruments add information, that can be used by policy makers.

Services from domestic labor are quite constant (or slowly decreasing): this is due to an increase of the number of pensioners [43] offset by a decrease of the number of housewives, especially due to the emancipation of women in the last decades. The larger number of women with a job is one of the factors of the improvement in income distribution. In fact, also the Gini Index slowly decreases. The increase of expenditure in durable goods, as well as the increase of private consumption, partially reflects fairer conditions in income distribution. Among durable goods, cars are strongly preferred by consumers. Indeed cars' ownership rises fast, inducing relevant environmental and social consequences for the territorial system of Pescara, especially within urban areas. For example, urbanization dynamics and traffic are the main causes of noise and air pollution, but dangerous positive feedbacks also occur, such as the increase in commuting and car accidents that are costs for a society. Urban systems can be a source of unsustainability also because of water consumption and pollution and loss of natural ecosystems that are transformed into built area. However, cities offer many resources to a community in terms of job opportunities, meetings, entertainment, relations, information exchange etc.; in other words, a city itself represents a resource. A city is a complex system able to auto-organize and it is an extremely productive system, because it provides goods and services, improving the quality of life and welfare. For these reasons urban areas require a particular care.

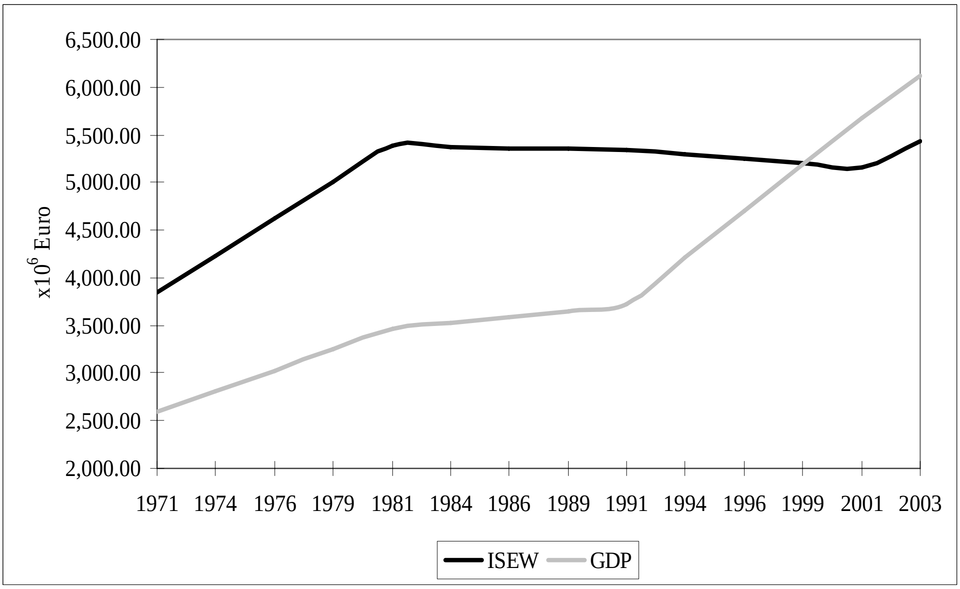

Figure 5. Comparison between GDP and ISEW trends.

The negative effects of the economic growth are also represented by the long-term environmental damage that is the monetary value of the environmental consequences of fossil fuels consumption. This item also underlines that our choices will influence the future of our sons. Continuous greenhouse gas emission is causing an increase in greenhouse effect that will persist for long time due to the lifetime of those gases in atmosphere (particularly CO_2). Future generations will pay the damage of climate change and greenhouse effect, and for this reason we should take into account today our responsibility. The value of long-term environmental damage reflects our growing dependence on non renewable energy and fossil fuels and it is thus constantly rising.

In sum, the Isew grows faster than GDP in the first years of the period. Afterwards, it tends to stagnate or even decrease. This is well represented by the trend of positive and negative items (see figures 6 and 7). In particular, the value of negative items is growing faster than that of positive ones, even though the latter is lower than the former. This is representative of the pressure connected to the economic growth. As Daly said: “When the economy’s expansion encroaches too much on its surrounding ecosystem, we will begin to sacrifice natural capital (such as fish, minerals and fossil fuels) that is worth more than the man-made capital (such as roads, factories and appliances) added by the growth. We will then have what I call uneconomic growth, producing bads faster than goods - making us poorer, not richer.” [4]

Table 4. The value of the items of the Index of Sustainable Economic Welfare for the Province of Pescara (in 1981 and 1991 an increase in wetland was recorded, that results in a positive value)

A	years	1971	1981	1991	2001	2003
B	consumption expenditure	1.237.835.760,97	2.428.065.624,66	2.891.599.069,98	2.942.943.977,10	2.897.395.324,20
C	Gini Index	0,362	0,282	0,287	0,322	0,305
D	adjusted consumption expenditure	908.836.828,90	1.893.966.945,91	2.246.774.724,15	2.226.130.088,57	2.220.226.302,07
E	services human labour	2.118.101.445,90	1.950.642.272,70	1.830.678.032,70	1.800.337.077,00	2.185.207.200,00
F	services durable goods	565.258.283,00	826.849.113,50	1.106.089.932,50	1.277.965.254,00	1.251.373.490,65
G	services from public infrastructure	21.676.500,00	21.676.500,00	22.755.508,00	23.140.868,00	23.140.868,00
H	health care and education public expenditure	408.961.031,45	431.895.516,87	449.954.057,20	520.440.383,77	495.893.648,18
I	expenditure on durable	39.365.703,41	75.748.187,51	213.230.287,11	352.304.782,95	372.989.118,36
J	health care and education public expenditure	73.339.543,00	62.560.037,42	180.867.425,68	72.957.758,85	79.055.978,04
K	local advertising					0,00
L	cost of commuting	52.961.314,84	91.956.253,84	146.391.741,11	168.272.273,46	52.961.314,84
M	cost of urbanization					0,00
N	cost of car accidents	1.354.757,12	1.094.812,05	1.795.447,08	5.576.807,91	5.917.344,00
O	cost of water pollution	11.787.550,83	14.302.764,51	14.616.317,19	13.137.515,10	13.327.789,13
P	cost of air pollution	81.375.904,01	78.402.622,49	63.047.666,52	98.314.055,52	100.763.460,54
Q	cost of noise pollution	6.463.955,58	11.944.041,43	18.762.525,75	21.992.228,38	22.830.259,14
R	loss of wetlands	0,00	53.964.196,43	51.934.705,96	0,00	0,00
S	loss of agricultural land	4.482.657,45	3.788.956,34	2.922.027,49	6.516.511,85	9.406.268,25
T	exhaustible resources depreciation	0,00	0,00	0,00	0,00	0,00
U	long-term environmental damage	46.349.668,16	92.341.511,36	147.644.199,21	208.224.330,60	222.080.355,58
V	net capital growth	1.074.194.999,90	2.881.602.751,64	357.333.063,64	190.900.749,36	190.900.749,36
W	ISEW	3.838.854.214,75	5.393.426.272,61	5.333.292.496,75	5.152.493.022,17	5.428.071.285,77

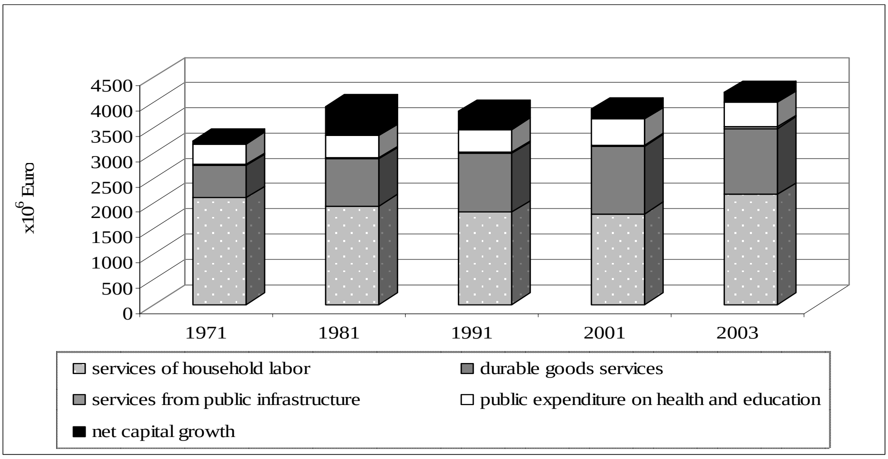

Figure 6. Positive items concurring to the calculation of the ISEW.

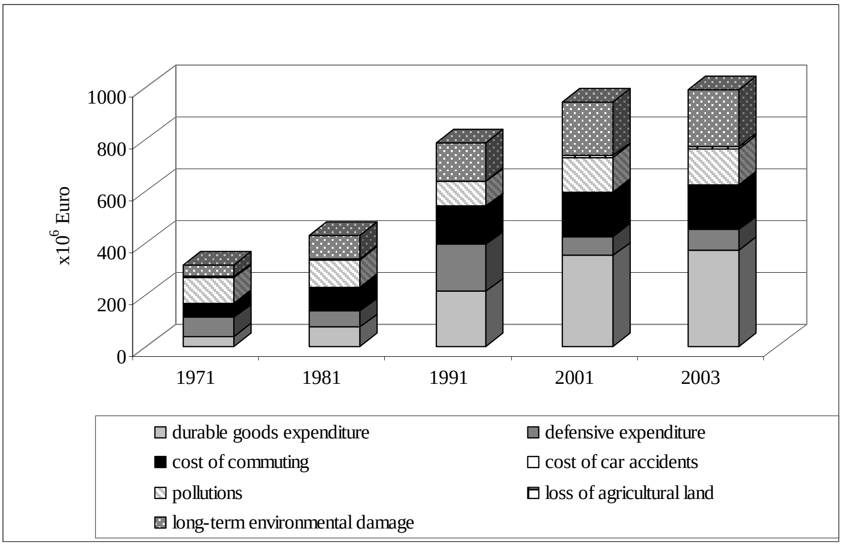

Figure 7. Negative items concurring to the calculation of the ISEW (cost of car accidents and loss of agricultural land are quite negligible).

Conversely, GDP tends always to grow, and it overshoots the ISEW curve at the end of the 1990s. Thenceforth, economic wealth does not completely correspond to well-being, as calculated by the ISEW: it means that a portion of wealth is not translated into welfare.

The analysis of the GDP indicates the economic performance of the socio-economic system, and not the level of welfare of a give community. The information from GDP that is used mostly for political reasons, is always insufficient for understanding the behavior of anthropic systems, especially because of their complexity. Since the System of National Accounts (SNA) fails to account for quality of life and the complexity of human activity in the biosphere, it is necessary to emphasize and improve diagnostic instruments that shift attention from merely economic parameters, such as consumption and GDP, to multidimensional objective measures.

CONCLUSION

GDP is the main indicator of economic performance. It measures the value of all goods and services produced in a territorial system (often a country) in a given year. It represents a milestone of economic theory, and it is the economic measure that no decision maker can neglect. The growth of GDP is a strategic aim for all, since a given increase in wealth constitutes the starting point to make all kind of political decisions. Some of these decisions are considered as collateral, for example social and environmental measures, because they often seem so far from a purely economic view.

Ecological Economics aims at redirecting economic thought towards society and the environment by providing proper instruments of analysis.

In this paper, the Index of Sustainable Economic Welfare (ISEW) is shown, putting on evidence the importance of a multidimensional view of anthropic complex systems under a sustainability viewpoint. Environmental and social pressures induced by economic growth, actually contribute to hold down the level of well-being of a population. Even in a not industrialized and overcrowded system, such as the Province of Pescara (southern Italy), in the last years, the ISEW is stagnating even if GDP continues to grow. This result confirms the so called threshold hypothesis according to which economic growth brings about an improvement in the quality of life but only up to a point beyond which, if there is more economic growth, the quality of life may begin to deteriorate [44].

REFERENCES

AA.VV. Annuario Regionale dei dati ambientali – ARTA Abruzzo, L'Aquila, 2003.

AA.VV. Listino dei Prezzi degli Immobili. Pescara Economica, 12. Camera di Commercio, Pescara, 2004.

ACI, 2005. Serie storiche sullo sviluppo della motorizzazione e sull'incidentalità stradale in Italia negli anni 1921-2003.

ANIA-Associazione Nazionale fra le Imprese Assicuratrici - L'assicurazione italiana 2003/2004; Roma 2004.

ANIA-Associazione Nazionale fra le Imprese Assicuratrici - Servizio Statistiche e Studi Attuariali; Roma 2004.

Brandolini A. The distribution of personal income in post-war Italy: Source description, data quality, and the time pattern of income inequality, Temi di Discussione, 350, Banca d'Italia, Roma, 1999Cannari L. e D'Alessio G. La distribuzione del reddito e della ricchezza nelle regioni italiane, Temi di discussione Banca d'Italia, n. 482, 2003.

Castañeda, B., E. An index of sustainable economic welfare (ISEW) for Chile. *Ecological Economics,* 1999, 28, 231-244.

Daly, H. E. Toward some operational principles of sustainable development. *Ecological Economics*, 1990, 2, 1-6.

Daly, H. Economics in a full world. *Scientific American*, September, 100-107, 2005

Daly, H. E., Cobb, J.B. For the common good: redirecting the economy towards community,

Daly, H. E., Cobb, J.B., Un'economia per il bene comune. Il nuovo paradigma economico orientato verso la comunità, l'ambiente e un futuro particolarmente sostenibile. RED edizioni, Como, Italy. 23, 1994.

El Serafy, S. The proper calculation of income from depletable natural resources. In: Ahmad, Y.J., Serafy, S.E., Lutz, E., (Eds) Environmental Accounting for Sustainable Development. An UNDP-World Bank Symposium, World Bank, Washington, DC. 10-18, 1988.

Gil, S., Sleszynski, J. An index of sustainable economic welfare for Poland. Sustainable Development, 2003, 11, 47–55.

Guenno, G., Tiezzi, S. The Index of Sustainable Economic Welfare (ISEW) for Italy. Worknote 5.98. Fondazione Enrico Mattei, Milano, Italy, 1998.

the environment, and a sustainable future. Beacon Press, Boston. 482 pp., 1989.

IPCC. Climate Change 1995. Impacts, Adaptations and Mitigation of Climate Change: Scientific-Technical Analyses. Cambridge University Press, 879 pp, 1996.

ISTAT, 1962. Dati provinciali su alcune principali caratteristiche strutturali delle aziende. 1° Censimento generale dell'agricoltura (1961) volume II – fascicolo provinciale Pescara n.° 68, ROMA.

ISTAT, 1971. Indagine campionaria sui consumi delle famiglie italiane, 1969. Note e relazioni, 49.

ISTAT, 1972. Dati sulle caratteristiche strutturali delle aziende. 2° Censimento generale dell'agricoltura (1970) volume II – fascicolo provinciale Pescara n.° 64, ROMA.

ISTAT, 1976. I consumi delle famiglie, 1975. Supplemento al Bollettino di Statistica, 13.

ISTAT, 1983. I consumi delle famiglie, 1979-1980-1981. Supplemento al Bollettino di Statistica, 5.

ISTAT, 1983. I conti della protezione sociale – sanità previdenza e assistenza anni 1960-1982. Supplemento al bollettino mensile di statistica, n.28, Roma ISTAT, 1986. Caratteristiche strutturali delle aziende agricole. 3° Censimento generale dell'agricoltura (1982) – tomo 1 – volume II – fascicolo provinciale Pescara n.° 68, ROMA.

ISTAT, 1991. Caratteristiche strutturali delle aziende agricole. 4° Censimento generale dell'agricoltura – fascicolo provinciale Pescara, ROMA.

I ISTAT, 1992. I consumi delle famiglie, anno 1991. Collana d'Informazione, ed.1992, 28.

STAT, 2002. I consumi delle famiglie, 2001. Annuari Famiglia e Società.

ISTAT, 2005. I consumi delle famiglie, 2003. Annuari Famiglia e Società.

ISTAT, 2005. Le retribuzioni contrattuali annue di competenza nel 2001. Settore lavoro, 15.

ISTAT, 2006. Investimenti fissi lordi per branca proprietaria, stock di capitale e ammortamenti Anni 1970-2005. Conti Nazionali, Statistiche in Breve. ISTAT, Roma.

ISTAT, 2007. I beneficiari delle prestazioni pensionistiche, anno 2005. informazioni January 11st, 2007 available online at *http://www.istat.it/salastampa/comunicati/non_calendario /20070111_00/testointegrale.pdf*Jackson, T., Marks, N. Measuring sustainable economic welfare. A pilot index: 1950-1990. Stockholm Environment Institute, The New Economics Foundation, 46 pp, 1994.

Jackson, T., Stymne, S. Sustainable Economic Welfare in Sweden: A Pilot Index 1950-1992. Stockholm Environment Institute, *The New Economics Foundation*, 46 pp, 1996.

Max-Neef, M., 1995. Economic growth and quality of life: a threshold hypothesis. Ecological Economics, 15, 115-118.

MIUR, 2000. La spesa per l'istruzione degli Enti locali. Elaborazione dell' ufficio di Statistica del Ministero dell'Istruzione, dell'Università e della ricerca su dati ISTAT – Ufficio VI. Roma

MIUR, 2003. La spesa pubblica per l'Istruzione. Elaborazione dell' ufficio di Statistica del Ministero dell'Istruzione, dell'Università e della ricerca su dati ISTAT – Ufficio VI. Roma.

Ministero del Bilancio e della programmazione economica, 1986. Relazione generale sulla situazione economica del Paese. Anni vari. La scuola italiana negli anni ottanta. volume 36, Roma.

Nordhaus, W., Tobin, J. Is growth obsolete? In: Economic Growth. National Bureau of Economic Research, General Series No. 96, New York, 1972.

Pulselli, F.,M., Bastianoni, S., Marchettini, N., Tiezzi, E. La soglia della sostenibilità. Rome, Italy. Ed. Donzelli, 2007, pp. 251.

Pulselli, F.M., Ciampalini, F., Tiezzi, E., Zappia, C. The Index of Sustainable Economic Welfare (ISEW) for a local authority: a case study in Italy. *Ecological Economics*, 2006, 60, 271-281.

Rosenberg, D., Oegema T. A Pilot ISEW for the Netherlands 1950-1992. Instituut vor Milieu-En systeemanalyse, Amsterdam, 1995.

Stockhammer, E., Hochreiter, H., Hobermayr, B., Steiner, K. The index of sustainable economic welfare (ISEW) as an alternative to GDP in measuring economic welfare. The result of the Austrian (revised) ISEW calculation, 1955 - 1992. *Ecological Economics*, 1997, 21, 19-34.

Tiezzi, E. *The end of time*. Southampton, UK. WIT Press, 2003.

Tomat, G.M. Beni durevoli, indici di prezzo e cambiamenti di qualità: un'applicazione ai prezzi delle automobili in Italia (1988-98), 439. Banca d'Italia, Roma, 2002.

Zolotas, X. Economic Growth and Declining Social Welfare. Bank of Greece, Athens, 1981.

In: Ecological Economics Research Trends
Editor: Carolyn C. Pertsova, pp. 273-291
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Chapter 13

THE DYNAMIC OF LAND USE IN BRAZILIAN AMAZON: A STRUCTURAL VAR WITH PANEL DATA ANALYSIS

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I. INTRODUCTION

Over the past two decades the international community has become aware of the global and regional environmental risks associated to possible massive forest losses in the Brazilian Amazon. The impacts on global carbon cycle, regional climate and the loss of biodiversity are among the main consequences of extensive land-use change processes in this region. Therefore a good understanding about the dynamics of land use in the Brazilian Amazon is fundamental in order to give us some indication about how to control this problem or even change the course of the situation in search for sustainable development.

Several studies have already contributed to a better understanding of underlying processes that drive land-use change in the Amazon. One of the main difficulty of land-use studies lies in the fact that pertinent economic and natural processes are fundamentally defined (and face constraints) on a fine geographic scale. This calls for an effort to brigde methodological gaps within a triangle of three approaches: classical economic approach (e.g. Angelsen, A., 1996), agent-based model (e.g. Soares-Filho et alli, 2006) and econometric modelling [(Reis and Blanco, 1997); (Pfaff, 1999), (Reis and Guzmán, 1994), inter allia]. Each of these approaches has advantages and drawbacks in function of the research objective. Here we decide to focus on the econometric approach. This methodology, applied to land-use, has been attacked to lack of economic consistency (in comparison for example to sectoral or general equilibrium models), but has definitively the advantage to take the most of existing data when the question is about describing land-use dynamics in the whole amazonian perimeter, inclusive non-optimal succession of uses from a strict economic standpoint.

Econometric and statistical methods have been used for example with the objective to determine main successional dynamics between antropogenic uses cropland, pastures, and regeneration that follows land clearing. Indeed, in order to seek for deforestation drivers, is important to know to what extent, in past deforestation trends, cropland or pastures uses are keener to immediately follow deforestation. Such trends also may differ for different "frontier patterns", for example the east versus southwestern frontiers of Amazonian deforestation. The issue of determining the main drivers of land-use in Brazilian Amazon is surrounded by controversies.

In one of the first attempts to determine a successionnal profile for the whole Amazon, Andersen et alli (2002, 1997) have estimated a vector autoregressive (VAR) involving the municipalities of Brazilian Amazon over the period 1970 through 1985. Regarding the use of this model to help and formulate environmental policies at least to considerations must be take place. A first difficulty lies in the fact that heterogeneities among cross-section unities (municipalities) have to be taken into account in order to contemplate the geographic, environmental, and economic diversity among the different sites of this region. A second difficulty is related to the fact that land-use census data in the Brazilian Amazon are only available on very few 5-years time steps. The environment in the Amazon is subjected to very dynamic forces that are capable to change the panorama of land use even in the "supposed" short period of 5 years. Combined with the fact that VAR approach is not structural model, it implies that the contemporaneous (i.e. at each studied time step) relationships among distinct land uses that occur during this period are not clearly treated. In this case, the model fails to adequately estimate successionnal dynamics to highlight causal orders of land-use change. In other words, such approaches are not adequate to simulate the effects of uncover stylized facts for the short-run impacts of the identified exogenous sources in the land use process. For instance, an exogenous technological shock or demographic shock can induce to a new unexpected evolution of soil occupation. Understanding of this "diffusion processes" in amazonian land-occupation can however be improved by using a model capturing the structural relations among land uses.

This article investigates the stochastic and dynamic relationship of land use in Brazilian Amazon. We adopt the structural VAR (SVAR) model with panel data to access for impacts of the identified exogenous sources. In this study we expand and refine previous works to tackle these two metodological issues. First, we take into account the heterogeneity among land units using the panel data[1] analysis model [(Hsiao, 1995), (Baltagi, 1995), (Arellano, 2003)], mixing information concerning variation of individual unities with variations taking place over time. Second, in the literature of structural VAR, the contemporaneous relationship is identified on the basis of prior information supported by theoretical considerations. Notwithstanding, other distinctive appeal of this article is the employment of Directed Acyclic Graphs (DAGs) estimated by the TETRAD (Spirtes et alli, 1993) to obtain the contemporaneous causal order of the SVAR.

The article is organized as follows. Section II defines the transitional land-use model. Section III discusses identification issues regarding to ascertain which are the true contemporaneous relationships among the land uses. We also introduce here the general

[1]. Important sources of variation may be left out if the data is only pooled in a single (temporal or spatial) dimension, and more precise parameter estimates can be obtained in panel approaches that explores the variation in data both across the counties and within counties over time.

issues regarding the DAGs model. In Section IV we propose a consistent methodology to estimate the model proposed in section II that contemplates the structural VAR with panel data. The description of database is done in Section V. The econometric results are presented in Section VI. Finally, we discuss the results and offer some concluding remarks on Section VII.

II. THE LAND USE MODEL FOR BRAZILIAN AMAZON

The need of using specific methodology that allows to dealing with the contemporaneous relationship existing among distinct land uses was soon perceived by many authors. Andersen and alli (1997, 2002) and Weinbold (1999) estimated a model of land use using vector autoregressive (VAR) evolving the municipalities of Brazilian Amazon. This model is reproduced in (1) [2]

$$\begin{aligned} crop_{it} &= \alpha_1 Dclear_{it} + \alpha_2 crop_{it-1} + \alpha_3 pasture_{it-1} + \alpha_4 fallow_{it-1} + \varepsilon_{1it} \\ pasture_{it} &= \beta_1 Dclear_{it} + \beta_2 crop_{it-1} + \beta_3 pasture_{it-1} + \beta_4 fallow_{it-1} + \varepsilon_{2it} \\ fallow_{it} &= \gamma_1 Dclear_{it} + \gamma_2 crop_{it-1} + \gamma_3 pasture_{it-1} + \gamma_4 fallow_{it-1} + \varepsilon_{3it} \end{aligned} \tag{1}$$

The indexes i and t are associated, respectively, to county and period of time. Our point here is that we can reformulate this model in order to contemplate some interesting issues on the process of land use and deforestation in Brazilian Amazon. The first issue is how we can identify the contemporaneous causal order subjacent to land use process. Any model that does not take this problem into account is not appropriate to answer any issue about public policy concerning the land use for this region. The second point is that the model presented in (1) does not deal explicitly with the heterogeneity regarding the municipalities of the Brazilian Amazon. Finally, how is it possible to exam the deforestation in the dynamic model of land use in order to include an explicit analysis of this subject.

The first and the third questions can be treated simultaneously with the same apparatus. Our proposal is to estimate the land use model using a structural VAR including more one endogenous variable associated explicitly to deforestation named here by forest.[3]. Considering that natural forest can be seen as another category of land use, the endogenous variables of our model can be represent by the following vector $(y_{1it}, y_{2it}, y_{3it}, y_{4it}) = (crop_{it}, pasture_{it}, fallow_{it}, forest_{it})$. Taking this notation we propose to use the following model to formalize the dynamics of soil occupation in Amazon,

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[2] Here $clear_{it} = crop_{it} + pasture_{it} + fallow_{it}$ and $Dclear_{it} = clear_{it} - clear_{it-1}$.

[3] This variable associated to natural land and it considers natural forest and natural pasture. More comments about the variables used in this research appear in Section IV.

$$\begin{pmatrix} 1 & a_{12} & a_{13} & a_{14} \\ a_{21} & 1 & a_{23} & a_{24} \\ a_{31} & a_{32} & 1 & a_{34} \\ a_{41} & a_{42} & a_{43} & 1 \end{pmatrix}\begin{pmatrix} y_{1it} \\ y_{2it} \\ y_{3it} \\ y_{4it} \end{pmatrix} = \begin{pmatrix} c_1 \\ c_2 \\ c_3 \\ c_4 \end{pmatrix} + \begin{pmatrix} b_{12} & b_{12} & b_{13} & b_{14} \\ b_{21} & b_{22} & b_{23} & b_{24} \\ b_{31} & b_{32} & b_{33} & b_{34} \\ b_{41} & b_{42} & b_{43} & b_{44} \end{pmatrix}\begin{pmatrix} y_{1it-1} \\ y_{2it-1} \\ y_{3it-1} \\ y_{4i-1t} \end{pmatrix} + \begin{pmatrix} \varepsilon_{1it} \\ \varepsilon_{2it} \\ \varepsilon_{3it} \\ \varepsilon_{4it} \end{pmatrix} \quad (2)$$

The spatial heterogeneity of the data calls for the use of can be treat using panel data method to estimate the model. According to the panel data analysis [(Hsiao, 1995), (Baltagi, 1995)] for each equation j the disturbance ε_{jii} in (2) can be decomposed by two stochastic components, such that, $\varepsilon_{jit} = \alpha_{ji} + \eta_{jit}$. Here α_{ji} is a stochastic term specific to the units, where $\alpha_{ji} \sim (0, \sigma_{j\alpha}{}^2)$, for $i = 1,...,N$, $t = 1,...,T$ and $j = 1,...,J$[4]. The term α_{ji} is introduced to model the heterogeneity among units. The term η_{jit} is the noise with expected value equal to zero and $E(\eta_{jit}{}^2) = \sigma_{j\eta}{}^2$. We assume that for any j and k, $E(\alpha_j \alpha_k^{'}) = \sigma_j^2$ if $j = k$, and $E(\alpha_j \alpha_k^{'}) = 0$, otherwise. And for any j and k, $E(\eta_j \eta_k^{'}) = \sigma_j^2$ if $j = k$, and $E(\eta_j \eta_k^{'}) = 0$, otherwise.

The system (2) is called the structural vector autoregressive[5] (SVAR) where $(y_{1it}, y_{2it}, y_{3it}, y_{4it})^{'}$ is the vector of endogenous variables. Adopting the vector representation where $Y_t = (y_{1it}, y_{2it}, y_{3it}, y_{4it})^{'}$, $Y_{t-1} = (y_{1it-1}, y_{2it-1}, y_{3it-1}, y_{4it-1})^{'}$ and $\varepsilon_t = (\varepsilon_{1it}, \varepsilon_{2it}, \varepsilon_{3it}, \varepsilon_{4it})^{'}$ this system can be rewritten such as

$$A_0 Y_t = c + A_1 Y_{t-1} + \varepsilon_t \quad (3)$$

If we assume that A_0 is invertible then (3) has a reduced form denoted VAR the is given by

$$Y_t = B + B_1 Y_{t-1} + u_t \quad (4)$$

with $u \sim N(0, \Sigma)$ where u is the reduced form of disturbance covariance matrix and it is assumed that $\varepsilon \sim (0, \Omega)$, Ω diagonal. The relationship between structural form and reduced form is based on the following identities, $B = A_0^{-1}\alpha$, $B_1 = A_0^{-1} A_1$, $u_t = A_0^{-1} \varepsilon_t$ and

[4] In our case it is clear that $J = 4$.

[5] For a concise reference on SVAR see Hamilton (1993). Enders (1995) provides a more intuitive treatment.

$$\Sigma = A_0^{-1}E(\varepsilon_t\varepsilon_t')(A_0^{-1})' = A_0^{-1}\Omega(A_0^{-1})' \tag{5}$$

Note that this representation does not allow for identifying the effects of exogenous independent shocks onto the variables, since in reduced form residuals are contemporaneously correlated (the Σ matrix is not diagonal)[6]. That is, the reduced form residuals u_t can be interpreted as the result of linear combinations of exogenous shocks that are not contemporaneously (in the same instant of time) correlated. It is not possible to distinguish whose exogenous shocks affect the residual of which reduced form equation. In evaluations of the model (and of economic policies) it only makes sense to measure exogenous independent shocks. Therefore, it is necessary to present the model in another form where the residuals are not contemporaneously correlated. Thus without additional restrictions on A_0 we cannot recover the structural form from the reduced form because Σ does not have enough estimated coefficients to recover an unrestricted A_0 matrix. Therefore, we need to impose a certain number of restrictions that will allow us to identify and estimate A_0. This procedure is named identification. It is possible to estimate the reduced form parameters B, B_1 and Σ consistently. But except forecast Y_t given Y_{t-1}, these are not the parameters of interest.

There are in general a large number of full rank matrices A_0 and D that allow us to reproduce from $\hat{\Sigma}$. That is, there are several conditional dependency and independency contemporaneous relations ("Markov kernels") between the variables – given by different specifications of which parameter in A_0 is free and which is equal to zero – that allow us to reproduce the partial correlations observed for the reduced form residuals[7]. In order to estimate the structural model it is necessary to identify a number of conditional independence relations (that is, parameters equal to zero in A_0) to satisfy the order condition for identification. Therefore, identifying A_0 is equivalent to identifying the conditional distributions ("Markov Kernels") of reduced form residuals from information about their joint distribution. These conditional distributions can be interpreted either distributionally or causally. The causal interpretation requires the knowledge that the conditional distributions would remain invariant under intervention. For SVARs to be useful, as we will show below, these conditional distributions must have a causal interpretation. Thus the structural form estimation proceeds in two steps. In the first step the reduced form VAR is estimated and this comprise the estimation of the equation's coefficients and of the covariance matrix of the

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[6] These shocks are primitive and exogenous forces, with no common causes, that affect the variables of the model.

[7] The matrices A_0 and D cannot have, together, a number of free parameters bigger than the number of free parameters in the symmetric matrix Σ. If n is the number of endogenous variables of the model then, to satisfy the order condition for identification of matrices A_0 and D, it is necessary that the number of free parameters to be estimated in A_0 be no bigger than n(n-1)/2 (matrix D is diagonal with n free parameters to be estimated). When n is smaller than n(n-1)/2 the model is over-identified. There exists no simple general condition for local identification of the parameters of A_0 and D. However, as has been shown by Rothenberg (1971), a necessary and sufficient condition for local identification of any regular point in R^n is that the determinant of the information matrix be different from zero. In practice, evaluations of the determinant of the information matrix at some points, randomly chosen in the parameter space, is enough to establish the identification of a certain model.

reduced form residuals ($\hat{\Sigma}$). In the second step the matrices A_0 and D are estimated using only the information given by the estimate of the covariance matrix of reduced form residuals (estimated in the first step).

III. IDENTIFICATION

Structural inference and policy analysis employing VARs require differentiating between correlation and causation, an issue known as “the identification problem”. The practice in the literature has been the use identifying assumptions based on “economic theory” or institutional knowledge to sort out the contemporaneous links among the variables in order to allow correlations to be interpreted causally.

The necessity of applying additional restrictions on the A_0 matrix is not neutral to the whole problem of identifying the dynamics of land-use transitions. Common statistical wisdom dictates that causal effects cannot be consistently estimated from observational data (non-experimental data) alone unless we have substantial *a priori* knowledge about the underlying mechanism. In others words without prior knowledge on contemporaneous effects it is not possible to identify matrix A_0 of the structural form. For this reason the contemporaneous causality restrictions used to identify structural VAR have been traditionally based on a priori restrictions, some of them arbitrary, others with some help of economic theory. It is fundamental to emphasis concerning to our case what the issue about the identification of the contemporaneous relationship really means. Given the exogenous impact on certain category of land it dictates the subsequent constraints by which the mechanism of land use take place during the period of five years in the Brazilian Amazon after this exogenous shock. In short, it detects how the other categories of land use react given a primary impulse that there exist on certain type of land use. In this paper it can be associated to the drivers or primer forcers of the soil occupation in this region.

The identification of this mechanism is an issue surrounded by controversies [(Mahar, 1989), (Myers, 1994), (Binswanger, 1991), (Almeida, 1992), (Reis and Blanco, 1997). (Fearnside, 2006)]. In the following lines we present a brief resume about the major proposals in the literature concerning the channels by which the land use process occurs and how these theoretical insights can be used to identify the contemporaneous causal order in the land use model.

During the initial stage of occupation in early sixties the Brazilian government played a decisive role (Reis and Blanco, 1997). Credit and fiscal subsidies to the agriculture jointly with the expansion of the road network pushed the agricultural frontier in the northwestern while the colonization programs aided to fix people in the interior. Strong incentives were created to clear land for pasture. In fact, the growth of cattle herds has consistently been cited as one of the primary factors behind the land clearing in Amazon. Regarding the problem of the identification this analysis implies that the subsequent effect of an enlargement of the area of pasture has a contemporaneous effect on the other land uses, likely on forest land whether a new area is required to be opened.

Neptad et alli (1999) identify a distinct pattern concerning the land use for Brazilian Amazon. These authors state that the logging starts a cycle that ends up when the soil is close to exhaustion. A similar viewpoint can be found in Walker and Homma (1996). The cycle can be briefly described in the following way. In the first place, it begins with the feeling of trees possibly to supply timber industry. Second, the process of land use starts with the implementation of agricultural activities evolving many perennials crops. In the following stage when the soil fertility declines, the site is used as pasture land. The cycle finishes when the soil is entirely exhausted and pasture area is converted in fallow land. A fallow period of ten years is required until the soil can be used to agricultural proposes again (Toniolo and Uhl, 1995). Thus it is more profitable for the farmer to convert the land to pasture and sell it to a nearby cattle ranch.

In time series analysis we must care about the data frequency, but in our case the situation changes. For instance, the land use cycle indicates by Neptad et alli (1999) requires about ten years to be complete what is the double of the frequency of our data. Notwithstanding, the cross-section dimension of the data composed by the municipalities is very large and definitively, this cycle takes places in different stages at the same time. Based on this analysis the land use process proposed by Neptad et alli (1999) can be represented by the following diagram

$$Forest \rightarrow Crop\,,\; Crop \rightarrow Pasture \text{ and } Pasture \rightarrow Fallow\,.$$

The identification of matrix A_0 constrained to this land use process can described in the following way

$$A_0 = \begin{pmatrix} 1 & 0 & 0 & A_{14} \\ A_{21} & 1 & 0 & 0 \\ A_{31} & A_{32} & 1 & 0 \\ 0 & 0 & 0 & 1 \end{pmatrix}$$

One other important question relates to the existence of specific and different patterns of land-use dynamics for different “frontiers”. Nowadays some authors advocate that because it precedes others activities slash-and-burn agriculture is identified as the cause of deforestation even though on many occasions it is a subsequent activity of cattle raising. This practice has commonly been used by small farmers, they burn a piece of forest and grow annual crop for about three years until the soil become useless. In contrast to slash-and-burn farming in forest areas, crop production in “cerrado” is dominated by large-scale agriculture technically advanced oriented to soybean production, and is also an obvious cause of deforestation in the state of Mato Grosso for instance. Fearnside (2006) corroborates this view pointing out that the drivers are by no means the same for all the sites of the Amazon. Other authors point out in the same direction. One of the remaining questions of several studies is about the relative strengh of crop versus pastures (and also forest exploitation) drivers of deforestation. In South-western Amazonian, the two factors (bovine meat production and crop production) may coincidate, while in eastern Amazonian, the pasture factor might be predominant.

Apart from ideological and theoretical divergences, the intellectual battle about the process behind land use process in Amazon is not supported by empirical evidence. In fact there is no clear explanation about the actors and forces that drive the land use in Brazilian Amazon. In this study we will restrict the use of prior theoretical restrictions in order to identify matrix of contemporaneous relationship A_0. The critical issue here is: it possible to obtain these contemporaneous causality relations, from the data, using just the contemporaneous correlation of the reduced form residuals, as summarized by their disturbance covariance matrix? Spirtes, Glymour, and Scheines (1993, 2000) [hereafter SGS] and Pearl and Verma (1991) claimed that it is possible to make causal inferences based on associations observed in non-experimental data without previous knowledge. These restrictions follow from directed acyclic graphs (DAGs) estimated by the TETRAD software developed by SGS using as input the covariance of reduced form disturbances. Moreover, if the causal relations can be represented by DAGs, SGS have shown that under some weak conditions – the Markov Condition and distribution of random variables "faithful" to the causal graph – there exist methods for identification of causal relations that are asymptotically (in sample size) correct[8]. The results of SGS are discussed in several articles[9].. In other words, this methodology enables to "read" from the data the contemporaneous relationships existing among distinct land uses in the Amazon. We describe next how the methodology developed by SGS can help in the identification of the structural form of system of equations.

DAGS and the Identification of Structural Form VAR

The SGS procedures allow us to establish the conditional independence relations that are equivalent to determining whose coefficients of matrix A_0 are equal to zero. The framework developed by SGS does not rule out the possibility of finding alternative sets of conditional independence relations for a given data set. In this case we arrive at a set of matrices A_0 that are observationally equivalent. It may be the case that the found conditional independence relations are not enough to allow for the identification of the matrices. In this case, additional restrictions are needed in order to identify the model. Next we show how DAGs can be used to impose restrictions that allows for the identification of Structural VARS (SVARs). In the example below we assume that the VAR has 4 endogenous variables.

The relationship between reduced form and structural form residuals is given by the following equation:

$$\nu_t = [I - A_0]\, \nu_t + \varepsilon_t$$

where :

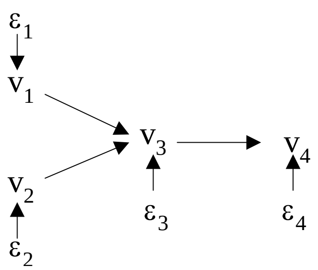
[8] Demiralp and Hoover (2003) evaluate the PC algorithm employed by TETRAD in a Monte Carlo study and conclude that it is an effective tool of selecting the contemporaneous causal order of SVARs.

[9] Swanson and Granger (1997) were the first to apply graphical models to identify contemporaneous causal order of a SVAR, although they restrict the admissible structures to causal chains. Bessler and Lee (2002) use error correction and DAGs to study both lagged and contemporaneous relations in late 19th and early 20th century U.S. data.

ν_t – column vector, with dimension 4x1, with reduced form VAR residuals at period t;
ε_t – column vector, with dimension 4x1, with structural form VAR residual at period t;
A_0 – full rank matrix with the relationship between the two types of residuals.

The above equations are a system of linear equations, where each variable (reduced form residual) is a linear function of its direct causes and an error term (structural residual), with error terms independent of each other. If the graph G that represents the model has no cycles (is a DAG) then the variables are generated by a Markovian model. Therefore, the model satisfy the property that guarantees the compatibility between its distribution function and graph G [10]. Because conditional independence implies zero partial correlation, Proposition 2 translates into a graphical test for identifying those partial correlations that must vanish in the model[11]. Therefore, equations (3) can be structured according to a DAG G, and the partial correlation coefficient $\rho_{V1V2.V3}$ vanishes whenever the vertices corresponding to the variables in V_3 d-separate vertex V_1 from vertex V_2 in G.

We present below an example of the relationship between a DAG (the graph below), equation (3) and the coefficients that are considered different from zero in A_0[12]:



$$v_1(t) = \varepsilon_1(t)$$
$$v_2(t) = \varepsilon_2(t)$$
$$v_3(t) = -A_{31}v_1(t) - A_{32}v_2(t) + \varepsilon_3(t)$$
$$v_4(t) = -A_{43}v_3(t) + \varepsilon_4(t)$$

Figure 1

$$A_0 = \begin{bmatrix} 1 & 0 & 0 & 0 \\ 0 & 1 & 0 & 0 \\ A_{31} & A_{32} & 1 & 0 \\ 0 & 0 & A_{43} & 1 \end{bmatrix}$$

Robins et al. (2003) [RSSW] use classical methods to analyze carefully the asymptotic properties[13] of SGS methodology. They show that in addition of being asymptotically

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[10] For a sketch of the proof see Pearl (2000).

[11] The partial correlation coefficient of X and Y, controlling for Z is given by $\rho_{XY.Z} = (\rho_{XY} - \rho_{XZ}\rho_{YZ})/(1-\rho_{XZ})^{1/2}(1-\rho_{YZ})^{1/2}$.

[12] ε_i is the structural error term of equation i (i=1,2,3,4).

[13] There are a variety of senses of asymptotic reliability in statistical inference, among which the most commonly discussed classical notions are pointwise consistency and uniform consistency. A pointwise consistent test is guaranteed to avoid incorrect decision if the sample size can be increased indefinitely. However, pointwise consistency is only a guarantee about what happens in the limit, not at any finite sample size. Pointwise consistency is compatible with there being at each sample size, some value of the parameter such that the probability of the estimator being far from the true value is high. A stronger form of consistency, uniform

consistent, the procedures are pointwise consistent, but not uniform consistent. Furthermore, they also show that there exists no causality test, based on associations of non-experimental data under the Markov and faithfulness assumptions, which is uniform consistent. Therefore, for any finite sample, it is impossible to guarantee that the results of the SGS causality tests (or any other causality test) will converge to the asymptotic results. The difficulties of the SGS procedure can be illustrated in Appendix IV by a simple example, taken from RSSW.

IV. ESTIMATION

In this section we will propose a consistent method to estimate the reduced form VAR with panel data (3) and as a consequence, to obtain the estimated error covariance matrix $\hat{\Sigma}$. We need this matrix for two reasons. First, we will perform the DGAs analysis to obtain the restriction necessary to identify A_0. Second, the matrix $\hat{\Sigma}$ and the constraints will used in order to retrieve the parameters of SVAR using the identification condition given by (4). In the pursuit of this target it is necessary to introduce the notations to accommodate the data sample. The observations related to each equation j can be framed in the following way. For $j = 1,...,J$, $y_j = (y_{j11},...,y_{j1T},...,y_{jN1},...,y_{jNT})'$ where y_j is $NT \times 1$ vector of dependent variables associated to equation j. Let $\alpha_j = (\alpha_{j1},...,\alpha_{jN})'$ the $N \times 1$ vector of individual effects and the $NT \times 1$ vector of noise is given by $\eta_j = (\eta_{j11},...,\eta_{j1T},...,\eta_{jN1},...,\eta_{jNT})'$. Thus the vector of stochastic disturbance ε_j of each equation j can be written as $\varepsilon_j = \eta_j + (I_N \otimes e_T)\alpha_j$ where e_T is the unit vector of dimension $T \times 1$. Thus the error covariance matrix is defined such that $E = (\varepsilon_j \varepsilon_j') = \Omega_{jj} = \sigma_{j\eta}^2 I_{NT} + \sigma_{j\alpha}^2 (I_N \otimes e_T e_T')$, where $\otimes$ is the Kronecker product.

We still need some additional notations in order to put the data in a compact system of equations. Then let $y = (y_1',...,y_j',...,y_J')'$ a $JTN \times 1$ vector of the endogenous variables, Z is the $JTN \times JM$ matrix of the lag variables such that,

$$Z = I_M \otimes X = \begin{pmatrix} X & 0 & ... & 0 \\ 0 & X & ... & 0 \\ ... & ... & ... & 0 \\ 0 & 0 & ... & X \end{pmatrix}$$

consistency, guarantees that it is possible to bound the decisions error rates with a finite number of observations. In what follows we will provide an informal discussion of pointwise and uniform consistency.

As it occurs in ordinary VAR, each equation j has the same set X of explained variables. Based on the above definitions the set of J equations of the reduced form VAR (3) can be written as,

$$y = Z\beta + \tilde{u}^{14} \tag{6}$$

where $\beta = (\beta_1,...,\beta_J)'$ is the vector of parameters where $\beta_j = (\beta_{1j},...,\beta_{Mj})'$ is the set of parameters of the j equation, $\tilde{u} = I_J \otimes (I_N \otimes e_T)\tilde{\alpha} + \tilde{\eta}$, $\tilde{\alpha} = (\tilde{\alpha}_1,...,\tilde{\alpha}_J)'$ is the $JNT \times 1$ vector of the reduced form of specific error component and $\tilde{\eta} = (\tilde{\eta}_1,...,\tilde{\eta}_J)'$ is the $JTN \times 1$ vector of reduced form. Here we have $\tilde{u} \sim \left(0, \overline{\Sigma}\right)$ where $\overline{\Sigma} = \Sigma_{\tilde{\eta}} \otimes Q_{\tilde{\eta}} + \Sigma_1 \otimes P_{\tilde{\eta}}$, where $\overline{\Sigma}$ is $JTN \times JTN$, $\Sigma_1 = T\Sigma_{\tilde{\alpha}} + \Sigma_{\tilde{\eta}}$, $P_{\tilde{\eta}} = I_N \otimes J_T$, $Q_{\tilde{\eta}} = I_{NT} - P_{\tilde{\eta}}$, $J_T = e_T e_T' / T$, $\Sigma_{\tilde{\eta}} = [\sigma^2_{\tilde{\eta}jl}]$ and $\Sigma_{\tilde{\alpha}} = [\sigma^2_{\tilde{\alpha}jl}]$, for $j = 1,..., J$ and $l = 1,..., J$; where $\Sigma_{\tilde{\eta}}$ and Σ_1 are of dimension $J \times J$.

The system (6) differs from the usual SUR[15] in view that Z correlates with $\tilde{\varepsilon}$ (Baltagi, 1980)[16]. To estimate (6) one also deals with Σ is not diagonal. This issue is solved premultiplying (5) by $\Sigma^{-1/2} = \Sigma_{\tilde{\eta}}^{-1/2} \otimes Q_{\tilde{\eta}} + \Sigma_1^{-1/2} \otimes P_{\tilde{\eta}}$ (Baltagi and Li, 1992) such that,

$$y^* = Z^*\beta + \tilde{u}^* \tag{7}$$

where $y^* = \Sigma^{-1/2} y$, $Z^* = \Sigma^{-1/2} Z$, $\varepsilon^* = \Sigma^{-1/2}\varepsilon$ with $\tilde{u}^* \sim \left(0, I_{JNT}\right)$. Using the instrumental variables W for Z^* and performing GLS[17], we get,

$$\beta^W = \left[Z^{*'} P_W Z^*\right]^{-1} Z^{*'} P_W y^*. \tag{8}$$

where $P_W = W\left(W'W\right)^{-1} W'$. In practice we need a feasible version of the estimator that replaces the unknown parameters $\Sigma_{\tilde{\eta}}$ and Σ_1 by consistent estimates, say $\hat{\Sigma}_{\tilde{\eta}}$ and $\hat{\Sigma}_1$. To derive one such consistent estimator, we follow the same path as in Baltagi (1995) and Cornwell et alli (1992). One can first estimate each (untransformed) by instrumental variable,

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14 The tilde is used over the disturbance term to indicate that it is not a primitive shock. In this notation the order between variables and parameters was change regarding the notation used in (3). It was done to ease the calculus and does affect the result.

15 Seemingly Unrelated Regressions.

16 The correlation between the disturbance and explained variables comes from the existence of lag variables. For any j there exist a correlation between the specific component α_{ji} and the lag of y_j due to for any j, y_{jit} is a function of α_{ji} and y_{jit-1} is also a function of α_{ji}.

17 Generalized Least Squares.

using instruments W. Let $(\hat{u}_1,...,\hat{u}_J)$ the set of residuals of the J equations. The consistent estimates of $\Sigma_{\tilde{\eta}}$ and Σ_1 in according to Baltagi (1995) can be obtained using,

$$\hat{\Sigma}_{\tilde{\eta}jl} = \hat{u}_j^{'} Q_\eta \hat{u}_l \Big/ N(T-1) \text{ and } \hat{\Sigma}_{1\eta jl} = \hat{u}_j^{'} P_\eta \hat{u}_l \Big/ N . \tag{9}$$

for $j = 1,..., J$ and $l = 1,..., J$. Finally, to obtain $\Sigma^{-1/2}$, Kinal and Lahiri (1990, 1993) suggest to use the Choleski decomposition of $\Sigma_{\tilde{\eta}}$ and Σ_1, such that,

$$S_1 S_1^{'} = \Sigma_{\tilde{\eta}} \text{ and } S_2 S_2^{'} = T\Sigma_{\tilde{\alpha}} + \Sigma_{\tilde{\eta}} . \tag{10}$$

V. DATABASE

The main original source of the data available for this study comes from Brazilian National Agriculture Census elaborated by the Brazilian Institute for Geography and Statistics (IBGE) which is usually conducted every five years. Others original data sources used are the Industrial Census and Commercial that were also elaborated by the IBGE for the same periods. The data were collected for the following for years 1970, 1975, 1980, 1985 and 1995 in municipality level. The data were cleaned, standard and merged with data of other sources by the Institute for Applied Economic Research (IPEA) managed by the team of IPEADATA[18]. The original database includes data on economic, demographic, ecological and agriculture variables.

In Brazilian Amazon a county can be subjected to ongoing change in its size mainly during the expansion of agricultural frontier in Amazon what obstructs the comparison inter periods at county level. That’s why it was created the concept of Minimum Comparable Area (MCA) which is the smallest spatial unity stable during these five census that accommodates the changing county boundaries over the panel The aggregation of counties in the later census years, in order to match the county area in 1970, is greatest in the more recently populated and sub-divided regions found in the legal Amazon.

The agricultural censuses group all land into private land and public land. Private land is stratified into eight categories according to agricultural use. They are (i) annual crops, (ii) perennial crops, (iii) planted forest, (iv) planted pasture, (v) short fallow and (vi) long fallow are classified in cleared land, while (vii) natural forest and natural pasture are considered non-cleared land. A small category of private non-usable land (rivers, mountains, etc.) is also considered non-cleared land. All land that is not claimed by anyone is considered public land and by definition non-cleared.

Based on these definitions the dependent variables used in the our land use model falls into any of the following four categories: cropland (*crop*), pasture (*pasture*), fallow (*fallow*) and natural land (*forest*). Cropland is land covers annual crops, perennial crops and planted forest. Pasture is planted pasture only. Fallow land includes short fallow, long fallow and

[18] http://www.ipeadata.gov.br

non-usable land like roads, dams, etc. Natural land considers natural forest and natural pasture.

VI. ECONOMETRIC RESULTS

In this section we present the econometric results for the land use model obtained in accordance with the methodology described in Section IV. We will also show here the results derived from the acyclic graphs (DGAs) estimated by the TETRAD introduced in Section III[19]. The objective of DGAs is indicating the contemporaneous relationships among the distinct land uses in Brazilian Amazon. This is the remarkable point of this research because it permits to identify some issues regarding the causal order in the land use process. Initially, we start our analysis presenting in Table 1 the parameters of the reduced VAR estimated by GLS-IV such as is shown in (8). The endogenous variables used to estimate the four equations is $crop$, $pasture$, $fallow$ and $forest$.

Although our database has a substantial number of cross-section unities, the temporal dimension of the data is very short and composed only by five observations. That is the why we perform a VAR with only one lag order. The set of instrumental variables used here is composed by the lag order two of the endogenous variables. The use of the second order lag of the endogenous variables as instruments reduces the dimension temporal of the data to $T = 3$. As it was shown in Section II, the disturbances of reduced form VAR are not the primary forces, but the linear combination of them. One consequence of this fact is that for every equation not only the lag of endogenous variable is endogenous but also the other explained variables, thus they must be also instrumented.

In order to verify the validity of the instruments we applied the Sargan test of overidentifying restrictions for a panel data fixed effects estimated via instrumental variables in which the number of instruments exceeds the number of regressors: that is, for an overidentified equation[20]. As one can see the Sargan test does not reject the null hypothesis that the instruments are valid. To test whether the specific disturbance must be treated as a random effect we used the specification test suggested by Hausman (1978) and Hausman and Taylor (1981). As it is shown in Table 1 the test statistics for the four equations far exceed the critical value of 5% level of significance

[19] More deep information about DGAs, can be obtained in Appendixes.

[20] A regression of the instrumental-variables residuals on the full instrument matrix gives rise to a Lagrange multiplier "N R-squared" test statistic for the joint null hypothesis that the equation is properly specified and the instruments are valid instruments (i.e. uncorrelated with the error term). The test statistic, under the null, is distributed Chi-squared(m), where m is the number of overidentifying restrictions. A rejection casts doubt on the validity of the instruments. See Davidson and MacKinnon (1993, p. 236) and Wooldridge (2002, p.123).

Table 1. Reduced Form VAR of the Dynamic Land Use Model (1970-1995)

Independent Variables	CROP (1)	PASTURE (2)	FALLOW (3)	FOREST (4)
L1CROPs	0,5080 (0,000)	0,6735 (0,006)	0,2408 (0,000)	0,8224 (0,2028)
L1PASTURE	-0,3980 (0,000)	-0,2291 (0,090)	-0,0779 (0,000)	-0,5295 (0,000)
L1FALLOW	-0.4385 (0,000)	-0,8282 (0,007)	-0.1748 (0,000)	-1.6763 (0.041)
L1FOREST	0,0303 (0,000)	-0,01702 (0,659)	-0,0025 (0,301)	0,0244 (0,692)
CTE	11944.81 (0,000)	5525.99 (0,000)	11383.09 (0,000)	14332.31 (0,000)
R^2	0,7299	0,8843	0,1058	0,5816
OBS	1028	1028	1028	1028
Notes: L1 = Lag order one. P-values in parenthesis. Test of over-identifying restrictions: 1.503 Chi-sq(1) P-value = 0.2203				

As we explained before, the estimating of the reduced VAR is not our main concern. The reduced form VAR presented in Table 1 is only an intermediary step by which the estimate of disturbance covariance matrix are generated in order to apply the specific procedure to retrieve the structural relations underlying the matrix A_0 in (5). This will be done in this study using the DGAs analysis proposed in Section III. To undertake this task firstly we need to obtain the error covariance matrix Σ that is composed by two matrices. As we known the residuals of reduced VAR can be used to obtain estimates of the error covariance matrices $\Sigma_{\tilde{\eta}}$ and Σ_1 as it is shown by (9). The covariance matrix of specific error Σ_α can be obtained using the fact that $\Sigma_1 = T\Sigma_{\tilde{\alpha}} + \Sigma_{\tilde{\eta}}$. Finally, $\hat{\Sigma}$ is obtained given that $\hat{\Sigma} = \hat{\Sigma}_\alpha + \hat{\Sigma}_\eta$.

Applying TETRAD at the 0.5% significance level[21] on $\hat{\Sigma}$ and assuming that the variables selected for the model are causally sufficient[22], we obtain what is known as a pattern[23], shown in figure 2. The pattern is a graphical representation of the set of observationally equivalent DAGs containing the contemporaneous causal ordering of the variables. The results of DGAs are shown in Figure 2.1. This figure indicates that five of DAGs is a valid representation of the contemporaneous causal ordering. In accordance with the pattern two of contemporaneous

[21] The significance level cannot be interpreted as the probability of type I error for the pattern output, but merely as a parameter of the search. Based on simulation tests with random DAGs, SGS suggests setting the significance level at 20% for sample size smaller than 100; at 10% for sample size between 100 and 300; and at 0.5% (or smaller) for larger samples. We followed their suggestion and set the significance level at 20%. However, slight changes in the significance level can produce large variations in TETRAD's output.

[22] A set of variables *V* is said to be causally sufficient if every common cause of any two or more variables in *V* is in *V*. TETRAD has a bias towards excluding causal relations present in the data, to overcome this problem it is suggested that a 20% significance level be used.

[23] A pattern is a partially oriented DAG, where the directed edges represent arrows that are common to every member in the equivalent class, while the undirected edges are directed one way in some DAGs and another way in others. Undirected edges (—) mean that there is causality in one of the two directions but not on both, while double oriented edges (↔) mean causality on both directions.

causal ordering display causality in one direction while the rest of them indicate causal ordering in both directions.

Based on the pattern of DGAs displayed in Figure 2.1 the following observations can be done. First, there does not exist contemporaneous relation between *pasture* and *fallow*. Second, there are two direct causality directions in the sense that *pasture* and *fallow* affect *crop* contemporaneously. Third, Figure 2.1 shows that there exists a not clear structural relationship between *crop* and *forest*, *forest* and *fallow* and *forest* and *pasture*. In other words, it means that for each of these pairs of land use there is causality in one of the two directions but not on both and model was not able to identify the true directions.

The information obtained by the pattern is useful but not enough to identify the matrix A_0. It gives four restrictions while we need more two constraints to make A_0 exactly identified[24]. In SVAR literature one of the usual way to support restricts is appealing to the prior or theoretical information. Following Faminow (1998) and Schnieder (1992), respectively, we assume that *pasture* and *crop* have contemporaneous effect on *forest* but not the reverse. The selected pattern is shown in Figure 2.2. Finally using equation the relation (5) can be used retrieve matrix A_0. The result is shown in Figure 3.

Figure 2.1
Pattern

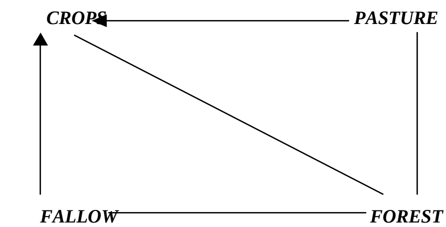
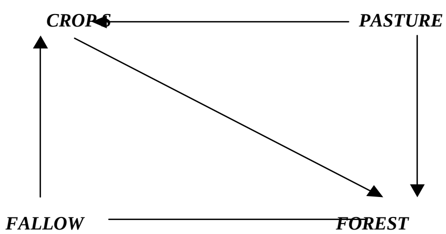


Figure 2.2
Selected



[24] See footnote X.

$$A_0 \quad \begin{matrix} 1 & A_{12} & A_{13} & 0 \\ 0 & 1 & 0 & 0 \\ 0 & 0 & 1 & A_{34} \\ A_{41} & A_{42} & A_{43} & 1 \end{matrix} \qquad \begin{matrix} 1 & 0.083(0.000) & 0.759(0.000) & 0 \\ 0 & 1 & 0 & 0 \\ 0 & 0 & 1 & 0.003(0.256) \\ 0.383(0.000) & 0.176(0.000) & 0.692(0.248) & 1 \end{matrix}$$

Figure 3[25].

VII. Analysis of Contemporaneous Relations and Concluding Remarks

Our objective here is to discuss the major points concerning the econometric results obtained in the previous section. Before to analysing the results from the estimated matrix A_0, it is necessary to explain how these results can be understood. Every element of A_0 is associated to a certain causal order between two distinct land uses. For instance, the coefficient A_{ij} means the contemporaneous effect of the land use j on the land use i. The order of the vector of land use is the same that appears in structural VAR shown in (2).

Based on the results displayed in Figure 3 we can note that some of contemporaneous relationships in A_0 can be associated to the theoretical concerns pointed out in Section III. The estimated coefficients A_{41} and A_{42} are significant and show, as one would expect, that marginal growths in crop or pasture happens mainly at the expense of natural land. Cattle ranching can be considered a prime force, because it creates a minimal infrastructure that allows the farmers to occupy an uninhabited site.

Coefficients in the in the first row of A_0 imply a less trivial result: in the sense that a marginal growth in pasture area also correlates with an increase in cropland area. For pastures, this highlights the fact that a small amount of croplands may be planted in complement to pasture, for example in order to produce crops for local or on-farm consumption. More surprinsing is the relative high coefficient related to contemporaneous relation between fallow and cropland. This means that substantial parts of cropland areas correlates with fallow within a 5-year time period. This would be coherent with the fact that croplands are often grown for a short period of time after deforestation for example, but that they are rapidly abandoned as yield decreases, if soil fertility is low The existence of a contemporaneous causality between fallow and crop land can be further justified by the necessity to maintain agricultural production within a municipality when a portion of land is abandoned. Finally, coefficients associated to the contemporaneous relation between fallow and natural land and vice-versa do not present statistical significance and therefore we will not discuss it.

These results suggest that public policies directed to sustainable occupation of the Brazilian Amazon should not treat separately cropland and pasture expansion control. Further improvements and application of the DAG method could consist in studying whether different

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25 P-values in parenthesis.

transitional patterns can be identified for different regions and deforestation fronts in the Amazon. For example, the economic structure of the deforestation frontier, and related drivers, may vary in the Eastern front with respect to the South Eastern front (Wassenaar et alli, 2007). This could reflect in different transitional patterns, and also suggest that public policies should be developed in function of the very economic and local structure of the deforestation fronts.

VIII. REFERENCES

Almeida, A. L. O. (1992). *Colonização Dirigida na Amazônia*. Série IPEA n. 135.

Andersen, L.E. (1996). *The Causes of Deforestation in Brazilian Amazon*. Journal of Environment and Development 5: 309-28.

Andersen, L. E. ,Granger, C. and Reis, E. J. (1997). *A Randon Coefficient VAR Transition Model of the Changes in Land Use in Brazilian Amazon*. Revista Brasileira de Econometria, 17(1): 1-13;

Andersen, L. E. ,Granger, C., Wunder, S. and Reis, E. J. (2002). *The Dynamic of Deforestation and Economic Growth in the Brazilian Amazon*. Cambridge University Press.

Arellano, M. (2003). Panel Data Econometrics. Oxford University Press.

Angelsen, A. (1996). *Deforestation: Population or Market Driven? Different approaches in modelling agricultural expansion*. CMI Working Papers 1996:9, Michelsen Institute, Bergen, Norway.

Awokuse, T. and Bessler, D. (2003). *Vector Autoregressions, Policy Analysis, and Directed Acyclic Graphs: an Application to the U.S. Economy*. Journal of Applied Economics VI(1): 1-24.

Baltagi, B. H. (1995). Econometric Analysis of Panel Data. John Wiley and Sons.

Baltagi, B. H. (1980). *On Seemingly Unrelated Regressions with Error Component*. Econometrica, 48: 1547-1551. (1981). *Simultaneos Equations with Error Components*. Journal of Econometrics, 17: 189-200.

Baltagi, B. H. and Qi, Li (1992). *A Note on the Estimation of Simultaneos-Equations with Error Components*. Econometric Theory, 8(1):113-119.

Bessler, D. and Lee, S. (2002). *Money and Prices: U.S. Data 1869-1914 (A Study with Directed Graphs)*. Empirical Economics 27: 427-446.

Binswanger, H.P. (1991). *Brazilian Policies that Encourage Deforestation in the Amazon.* World Development, 19(7): 821-829.

Browder, J. O. (1988). *The Social Cost of Rainforest Destruction: a critique of the hamburger debate*. Intercience, 13: 115-120.

Cornwell, C. Schmidt, P. and Wyhowski, D. (1992). *Simultaneos Equations and Panel Data.* Journal of Econometrics, 51: 151-181.

Davidson, R. and MacKinnon, J. G. (1993). Econometric Theory and Methods. Oxford University Press.

Demiralp, S. and Hoover, K. (2003). *Searching for the Causal Structure of a Vector Autoregression.* Oxford Bulletin of Economics and Statistics 65 (Supplement): 745-767.

Enders, W. (1995). Applied Econometric Time Series, John Wiley and Sons.

Faminow, M. D. (1998). Cattle, Deflorestation and Development in Latin American. CAB International.

Fearnside, P. M. (2006). *Desmatamento na Amazônia: dinâmica, impactos e controle*. Acta Amazônica, 36(3): 395-400.

Hamilton, J. (1993). *Time Series Analysis.* Princeton University Press.

Hausman, J. (1978). *Specification Tests in Econometrics*. Econometrica, 46(6): 1251-1271.

Hausman, J. and Taylor, W. (1981). *Panel Data and Unobservable Individual Effects*. Econometrica, 49(6): 1377-1398.

Hsiao, C. 1995. *Analysis of Panel Data*. Cambridge University Press.

Jacobs, R. Leamer, E. E. and Ward, M. P. (1979). Difficulties with Testing for Causation. Economic Inquiry, 17: 401-413

Kinal, T. and Lahiri, K. (1993). *On the Estimation of Simultaneous-Equations Error-Components with an Application to a Model of Developping Country Foreign Trade*. Journal of Applied Econometrics, 8(1): 81-92.

______________________ (1990). *A computational algorithm for multiple equation models with panel data*. Economics Letters 34:143-146**.**

Mahar, D. (1989). *Government Policies and Deforestation in Brazil's Amazon Region*. Washington, DC: World Bank

Myers, N. (1994). *Tropical Deforestation Rates and Patterns*. In: Brown, K and Pearce, D. (eds.) The causes of tropical deforestation, the economic and statistical analysis of factors giving rise to the loss of tropical forests, 172-91. University College London Press.

Nepstad, D. C., Moreira, A. G. e Alencar, A. A. 1999 *A Floresta em Chamas: Origens, Impactos e Prevenção de Fogo na Amazônia. Brasília: Programa Piloto para a Proteção das Florestas Tropicais do Brasil.*

Pearl, J. (2000). Causality: Models, Reasoning, and Inference. Cambridge University Press.

Pearl, J. and Verma, T. (1991). A Theory of Inferred Causation. In J. Allen, Fikes, R. and

Pfaff, A.S. 1999. *What drives deforestation in the Brazilian Amazon? – Evidence from Satellite and Socioeconomic Data.* Journal of Environmental Economics and Management, 37(1): 26-43.

Reis, E. J. and Blanco, F. A. (1997). *The Causes of Brazilian Amazon Deforestation*. Working Paper, IPEA.

Reis, E and Guzmán, R. 1994. *An Econometric Model of Amazon Deforestation*. In: Brown,K and Pearce, D. (eds.) The causes of tropical deforestation, the economic and statistical analysis of factors giving rise to the loss of tropical forests, 172-91. University College London Press, London.

Robins, J., Sheines, R., Spirtes, P. and Wasserman, L. (2003). *Uniform Consitency in Causal Inference*. *Biometrika* 90(3): 491-515.

Spirtes, P., Glymour, C. and Scheines, R. (1993). Causation, Prediction, and Search. Lecture Notes in Statistics 81. Springer-Verlag.

48(4): 555-568. 2000). *Causation, Prediction, and Search.* MIT Press.

Soares-Filho, B. S.; Nepstad, D.; Curran, L.; Voll, E.; Cerqueira, G.; Garcia, R. A.; Ramos, C.; McDonald, A.; Lefebvre, P.; Schilesinger, P. *Modeling Conservation in the Amazon Basin*. Nature, 440: 520-523, 2006.

Scheneider, R. R. (1992). *An Economic Analysis of the Environmental Problem in the Amazon*. Draft, World Bank.

Swanson, N. and Granger, C. (1997). *Impulse Response Functions Based on a Causal Approach to Residual Orthogonalization in Vector Autoregressions*. Journal of the American Statistical Association 92(437): 357-367.

Toiolo, A. and Uhl, C. (1995). *Economic and Ecological Perspectives on Agriculture in Eastern Amazon*. World Development, 23: 959-973.

Walker, R. and Homma, A. K. O. (1996). *Land Use and land Cover Dynamics in Brazilian Amazon: an overview*. Ecological Economics, 18: 67-80.

Wassenaar, T., Gerber, P. Verburg, P.H., Rosales, M., Ibrahim M. and Steinfeld, H. (2007). *Projecting land use changes in the Neotropics: The geography of pasture expansion into forest.*, Global Environmental Change, 17(1): 86-104.

Weinhold, D. (1999). Estimating the Loss of Agricultural Productivity in the Amazon. Ecological Economics, 31: 63-76.

Wooldridge, J. M. (2002). Econometric Analysis of Cross Section and Panel Data. The MIT Press.

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Editor: Carolyn C. Pertsova, pp. 293-307 ISBN 1-60021-941-1

Chapter 14

REGULATION OF NON-POINT SOURCE POLLUTION AND INCENTIVES FOR GOOD ENVIRONMENTAL PRACTICES - THE CASE OF AGRICULTURE

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INTRODUCTION

The early stages of environmental policy implementation were characterized by the regulation of point source pollution. This is partly explained by the easy identification of these sources and the broad and strong political support for their regulation. In contrast, the regulation of non-point sources began much later and is still being implemented. For instance, international agreements to reduce water and gaseous emissions within the EU are likely to lead to more regulations in European countries in coming years. The European Water Framework Directive (WFD) has great power to reduce non-point pollution in European member states. This initiative is supported by European Environmental Agency findings (2006) that point to agricultural non-point pollution as the primary cause of water quality deterioration in many European watersheds. From an economic perspective, agricultural non-point control involves difficult planning problems characterized by complexity, uncertainty and policy conflicts. The major feature of non-point pollution is that emissions are either not observable or cannot be observed at a reasonable cost. Therefore it is impossible to attribute emissions to particular polluters, and the use of first-best instruments is infeasible. Unfortunately, the economic literature does not clearly indicate which are the optimal second-best instruments to regulate non-point sources.

Given the difficulty of metering discharges with a reasonable degree of accuracy, Segerson (1988), Xepapadeas (1991) and Xepapadeas (1992) proposed taxing the

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concentration of the pollutant in the environmental media receiving the emissions (ambient tax). Even though recent work by Horan et al. (1998), Hansen (1998) and Hansen and Romstad (2006) has reduced the informational requirements on the regulator to implement ambient taxes, their political acceptability may still be severely limited as there is currently no direct relationship between individual behaviour and the size of the actual tax (Shortle and Abler, 1998).

Another strand of the economic literature focuses on instruments to control pollution indirectly, for instance, regulating inputs or the management practices of the firm. However, in order to be applicable, regulating inputs or management practices has to be truncated to a subset of choices that are easy to observe and highly correlated with pollutant emissions. The restriction on the number of inputs considered limits these approaches to being second-best. Studies like the ones by Mapp et al. (1994), Larson, Helfand and House (1996) and Vickner et al. (1998) analyze the environmental and economic impact of regulating a contaminating input, either by restrictions on the choice of the inputs or by a change in the applied technology. Among the techniques for the second-best regulation of inputs, little attention has been paid to the choice of management practices. Other policy options like voluntary controls resulting from moral suasion and/or education, or economic incentives for farmers to reduce emissions by changing or modifying their farming practices have also received little attention in the literature.[2]

Designing an environmental policy is very difficult in practice. Due to the lack of powerful instruments, regulation in EU member states is concentrated on control instruments like technology related standards and management rules. But these instruments often do not reveal economic incentive and farmers do not act voluntarily.[3] Consequently the introduction of clean technologies and/or the implementation of good environmental practices requires continuous and strict controls which in turn lead to high costs for the regulator. In addition, since information problems prevent the use of first-best instruments, a theoretical rationale for combining instruments may exist.

In this paper we propose a combination of incentives (deposit-refund system) to encourage the adoption of good environmental practices to reduce nitrate emissions due to livestock management. The main objective of this study is to describe the equilibrium conditions that make adapting good environmental practices desirable from social and private points of view. For this purpose we design a specific tax on a polluting input (deposit) and a subsidy for the voluntary adoption of good environmental practices (refund). In contrast to previous work (Fullerton and Wolverton, 2000), the deposit refund system is not linked to output or input but the way the input is applied. As the correct application of good environmental practices cannot be observed by the regulator, the payment of the refund does not depend on any control exercised directly by the regulator. Instead, the payment depends on the presentation of a certificate, issued by independent persons or firms certified to apply the polluting input. Those persons or firms guarantee with their reputation and future business perspectives that the polluting input has been applied in accordance with good environmental

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[2] For an excellent and still not outdated overview of the state of the art in non-point source pollution control see Shortle and Abler (1998) and Shortle et al. (1998).

[3] The certification of management practices may offer economic incentives if it refers to the quality of the produced good, for instance organically produced goods.

practices. In practice, the certified person can be a farmer of the region who has obtained a license to apply the polluting input.

The theoretical analysis is presented in the following section and thereafter we present an empirical analysis for the optimal management of livestock and cultivation activities.

ECONOMIC PROBLEM

To be more specific we consider a farmer that fattens hogs and cultivates corn. The farmer uses livestock slurry as a fertilizer for the production of the crop. The total net benefits obtained by the farmer originate from the net benefits of pig farming, denoted by π_p, and from the net benefits of corn production, denoted by π_c.

The net benefits from hog production basically depend on the number of places available for the animals. Hence we have:

$$\pi_p = p_p \cdot p - k_f \tag{1}$$

where p_p are the net benefits from each place (€/place), p is the number of places and k_f is the fixed cost of the installation.

As a consequence of hog production, a certain amount of slurry is generated that the farmer uses as a fertilizer in corn production. The farmer can choose between two different technologies, $j=i,s$, for the application of the slurry: a cheap but highly polluting technology (insecure technology) or a less polluting, more costly technology (secure technology). The subscript $j = i$ represents the insecure technology, and $j = s$ denotes the secure technology. The amount of nitrogen contained in the slurry and applied with technology j is denoted by n_j. The costs of the application of the slurry are denoted by p_j (€/kg N). All prices and costs cannot be influenced by the farmer and are therefore considered as given data. The amount of nitrogen assimilated by the crop is a function of the amount of the slurry applied, and is given by $n_a = \sum_j g_j(n_j)$, with $dg_j/dn_j \geq 0$; $d^2g_j/dn_j^2 < 0$. We assume that the secure technology uses the nitrogen more effectively, thus, for any given level of applied nitrogen $\hat{n}$, $g_s(\hat{n}) \geq g_i(\hat{n})$.

The function $y(n_a)$ denotes the crop production function (Tm/ha) and depends on nitrogen taken up by the plant. It has the regular properties of a neoclassical production function, i.e., $y_{n_a} > 0, y_{n_a n_a} < 0$.

Given the preceding definitions, the net benefits per hectare from corn production are given by:

$$\pi_c = \sum_j \left(p_y \cdot y(n_a) - p_j \cdot n_j \right) \cdot \delta_j - c + s \tag{2}$$

\where p_y denotes the crop price (€/Tm.ha), c is the total fixed cost for corn production (€/ha), and s denotes the direct payments according to the Common Agricultural Policy (CAP) (€/ha). Finally, δ_j reflects the choice of the technology, namely, δ_j =1 if technology j is employed and δ_j =0 if it is not.

The farmer can choose the technology, the amount of organic fertilizer and the number of places available for hog production. We assume that the farmer's objective is to maximize the total net benefits, and hence the decision problem can be formally stated as:

$$\underset{n_i, n_s, \delta_i, \delta_s, p}{Max} \quad \pi_p + \pi_c \tag{3}$$

subject to

$$n_i \delta_i + n_s \delta_s = u \cdot p \tag{4}$$

$$\delta_i + \delta_s = 1 \tag{5}$$

$$n_j \geq 0,\ j = i, s,\ 0 \leq p \leq P\,. \tag{6}$$

Condition (4) requires a balance between the amount of nitrogen applied with the slurry and the amount of nitrogen generated by hog production, where u indicates the amount of nitrogen generated by each place (kgN/place). Condition (5) establishes that both technologies cannot be used simultaneously, and equation (6) presents constraints on the admissible values of the control variables.

Taking account of the constraints leads to the Lagrangian:

$$\begin{aligned} L = {} & p_p \cdot p - k_f + \sum_j \left(p_y \cdot y(g(n_j)) - p_j \cdot n_j \right) \cdot \delta_j - c + s + \mu_1 \left(n_i \delta_i + n_s \delta_s - u \cdot p \right) + \\ & \mu_2 \left(\delta_i + \delta_s - 1 \right) + \mu_3 n_i + \mu_4 n_s + \mu_5 p + \mu_6 (P - p) \end{aligned}$$

where $\mu_1, \ldots, \mu_6$ are Lagrangian multipliers.

A solution to the problem has to satisfy the following necessary conditions:

$$\frac{\partial L}{\partial n_i} = \left(p_y \cdot \frac{dy}{dn_i} - p_i + \mu_1 \right) \delta_i + \mu_3 = 0 \tag{7}$$

$$\frac{\partial L}{\partial n_s} = \left(p_y \cdot \frac{dy}{dn_s} - p_s + \mu_1 \right) \delta_s + \mu_4 = 0 \tag{8}$$

$$\frac{\partial L}{\partial \delta_i} = p_y \cdot y(g_i(n_i)) - p_i \cdot n_i + \mu_1 n_i + \mu_2 = 0 \quad (9)$$

$$\frac{\partial L}{\partial \delta_s} = p_y \cdot y(g_s(n_s)) - p_s \cdot n_s + \mu_1 n_s + \mu_2 = 0 \quad (10)$$

$$\frac{\partial L}{\partial p} = p_p - \mu_1 u + \mu_5 - \mu_6 = 0 \quad (11)$$

$$n_i \delta_i + n_s \delta_s = u \cdot p \quad (12)$$

$$\delta_i + \delta_s = 1 \quad (13)$$

Necessary conditions (7) and (8) indicate for an interior solution that nitrogen should be employed for each technology up to the point where the value of the marginal product of applied nitrogen plus the shadow value of hog production equals the marginal cost of nitrogen use. Necessary conditions (9) and (10) determine that the technology with greater benefits should be chosen. Condition (12) establishes that all the generated nitrogen must be employed on the farm. Hence, equation (11) establishes for an interior solution that the net benefits from each place should equal the marginal cost given by the shadow value of hog production multiplied by the amount of nitrogen produced by each place. For an interior solution of equation (11) one expects the term $\mu_1 u$ to be positive, representing the marginal cost of the generated nitrogen from an additional place. However, if $p = P$ it could be the case that there is a shortage of nitrogen on the farm, and $\mu_1 u$ presents the marginal value of the generated nitrogen from an additional place. Equation (12), and the fact that p_p is strictly positive, guarantees that the solution of the first order condition (11) with respect to the number of places is finite.

Figure 1 presents the farm net benefits obtained with the insecure and secure technologies. In order to have an interesting economic problem we assume that the secure technology leads to lower farm net benefits than the insecure technology because of its higher associated costs. Consequently, the farmer will adopt the insecure technology since it generates higher profits.

The portion of nitrogen not assimilated by the crop percolates or runs off in the form of diffuse pollution leading to an accumulation of nitrate in aquifers or surface waters. The nitrogen emissions when technology j is used are given by $n_j - g_j(n_j)$, and total emissions by:

$$E = \sum_j \left(n_j - g_j(n_j) \right) \cdot \delta_j \quad (14)$$

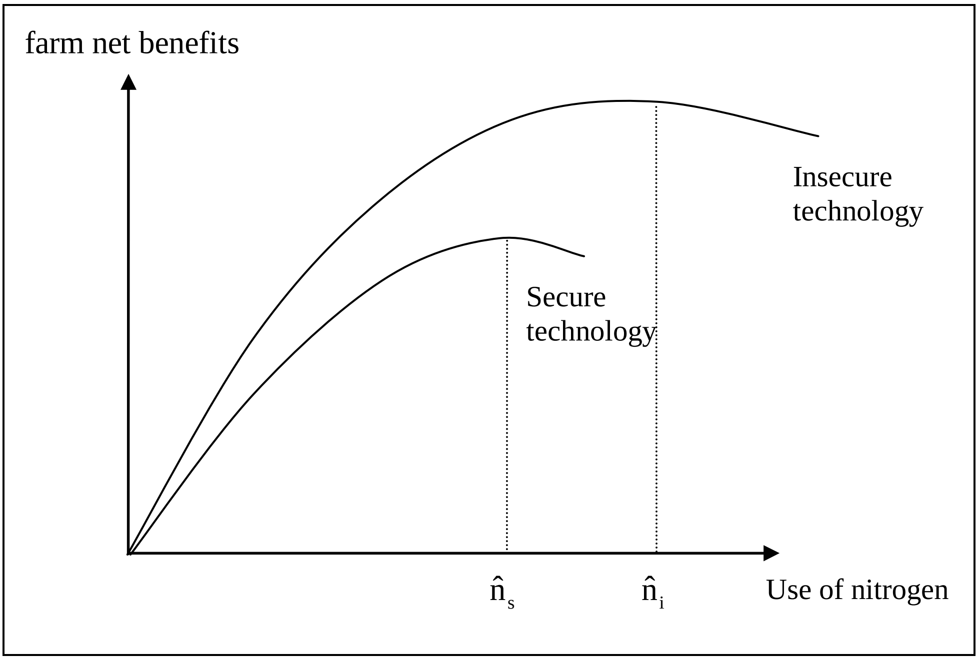

Figure 1. The farm net benfits as a function of nitrogen.

Since the insecure technology is less efficient with respect to the assimilation of nitrogen by the crop, it causes greater environmental damage than the secure technology. Let MD_j define the marginal monetary damage function of nitrogen emissions for technology *j*.[4]

Moreover, given technology *i*, let $\hat{\pi}$ denote the maximized net benefits of the farmer in the absence of any environmental policy. The optimal input level for the firm for the unconstrained profit maximum is $\hat{n}_i$. Likewise, let π_j denote the constrained maximum net profits of the firm if the level of applied nitrogen, n_j, cannot exceed $\hat{n}_j$, utilizing technology *j*. A reduction in nitrogen leads therefore to a reduction in the farm net benefits. Consequently, the constrained profits will be lower than the unconstrained profits. Let us define the firm's abatement costs C_i and C_s as the difference between unconstrained and constrained profits, that is:

$$C_i = \hat{\pi} - \pi_i$$

$$C_s = \hat{\pi} - \pi_s.$$

The abatement costs are a function of the established upper limit of nitrogen: the lower the limit, the higher the abatement costs of the firm. Given our assumptions about the farm net benefit functions, the marginal abatement costs in the case of the insecure technology are always greater than the marginal abatement costs of the secure technology. Economic theory

[4] For simplicity, the marginal damage functions of Figure 2 are depicted as linear functions, implying that the abatement cost functions are quadratic. Nevertheless other specifications are equally plausible.

establishes that the efficient level of emissions is achieved when the marginal abatement costs are equal to the marginal pollution damages. Figure 2 depicts the socially optimal use of nitrogen for each technology.

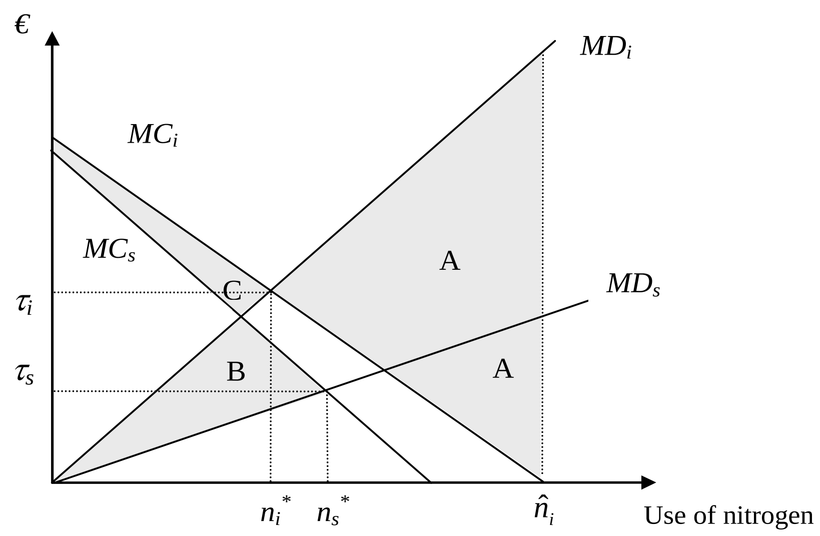

Figure 2. Adoption of the secure technology is socially preferable.

The farmer maximizes profits by applying $\hat{n}_i$ units of nitrogen. However, since nitrogen leaching causes environmental damage, the farmer's optimum $\hat{n}_i$ and the optimums of the society n_s^* and n_i^* do not coincide. Consequently, farmers will apply nitrogen at socially inefficient rates.

If the farmer employing the insecure technology was required to limit the application of nitrogen to n_i^*, the total abatement cost for the farmer would be the area below $MC_i(n)$, between n_i^* and $\hat{n}_i$, that is $\int_{n_i^*}^{\hat{n}_i} MC_i(n)dn$. However, since the avoided monetary damages correspond to the area $\int_{n_i^*}^{\hat{n}_i} MD_i(n)dn$, the overall social gains are positive and correspond to the areas A shown in Figure 2. On the other hand, if farmers employ the secure technology, the optimal level of applied nitrogen is n_s^*. Hence, the total abatement cost for the farmer will be given by the area $\int_{n_s^*}^{\hat{n}_i} MC_i(n)dn$ as a result of the fertilization restriction, and the area $\int_0^{n_s^*} MC_i(n) - MC_s(n)dn$ as a result of the change in the employed technology. In this case, the avoided damages correspond to the area $\int_{n_s^*}^{\hat{n}_i} MD_i(n)dn + \int_0^{n_s^*} MD_i(n) - MD_s(n)dn$.

The difference between these areas indicates whether or not it is socially preferred to adopt the secure technology. For instance, area B (avoided damages) in Figure 2 is greater than area C (additional abatement costs) and from a social point of view the secure technology is therefore preferable to the insecure technology. This condition is more likely to hold the lower the difference in application costs and the higher the difference in environmental damages between the two technologies. On the contrary, Figure 3 shows the case where the difference in environmental damages is not significant, and the adaptation of the secure technology is very costly. In this case it is socially optimal to employ the insecure technology.

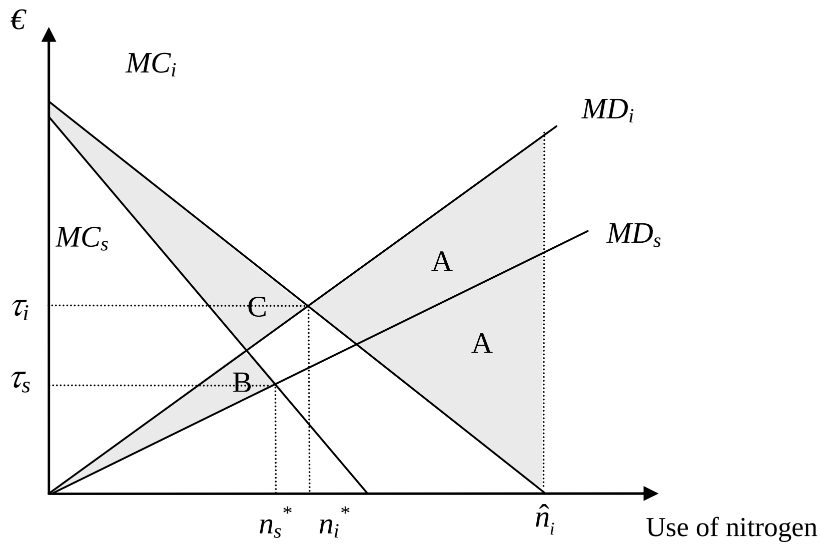

Figure 3: Adoption of the secure technology is socially **not** preferable.

Given the situation depicted in Figure 2, it would be socially optimal to adopt the secure technology. However, as it reduces their income, farmers are not likely to adopt the secure technology voluntarily.

To establish the social optimum in the case where adoption of the secure technology is efficient, we propose a solution based on the combination of a deposit (tax) and a refund (subsidy). In the deposit refund system the voluntary adoption of secure technology maximizes the farmers' net benefits. In the case of hog production the regulator can observe the amount of nitrogen generated at the farm but not whether the farmer employs insecure or secure technology, i.e., bad or good environmental practices. To obtain this information, the regulator creates the figure of a certified person who applies nitrogen in accordance with good environmental practices or oversees its correct application. Once the nitrogen has been applied, the certified person issues a certificate that shows the amount of nitrogen that has been applied correctly. Moreover, the regulator introduces a general tax on nitrogen that has to be paid by all farmers. The optimum level of the nitrogen tax is equal to the marginal cost of abatement at the socially optimal level of nitrogen applied with the insecure technology.

Given technology *i* and tax τ_i , the total costs for the farmer are the area $\int_{n_i^*}^{\hat{n}_i} MC_i(n)dn$ and the rectangular $\tau_i\, n_i^*$ (payments of the taxes). These costs correspond to the shaded area, denoted as D, in Figure 4. With the secure technology, abatement costs are $\int_{n_s(\tau_i)}^{\hat{n}_i} MC_i(n)dn$, the payment of the taxes $\tau_i n_s(\tau_i)$, and $\int_0^{n_s(\tau_i)} MC_i(n) - MC_s(n)dn$, that is, the shaded area D plus the area E in Figure 4. Consequently, the farmer will not adopt the secure technology since the abatement costs associated with the secure technology (D+E) are greater than the abatement costs associated with the insecure technology (E). These reflections show that the farmer's decision to adopt the secure technology or not is independent from whether or not the adoption of the secure technology is socially optimal.

To induce the farmer to use the secure technology, the optimum level of the subsidy is given as $\tau_i - \tau_s$. It reflects the decrease in the marginal environmental damage when the secure technology is adopted. If the received subsidy, $(\tau_i - \tau_s)n_s^*$, compensates the additional abatement costs associated with the adoption of the secure technology (area E), the farmer adopts the secure technology and applies the optimal social level of nitrogen n_s^*. In practical terms the farmer will receive refund $\tau_i - \tau_s$ for the amount of nitrogen stated in the certificate obtained from the certified person applying it.

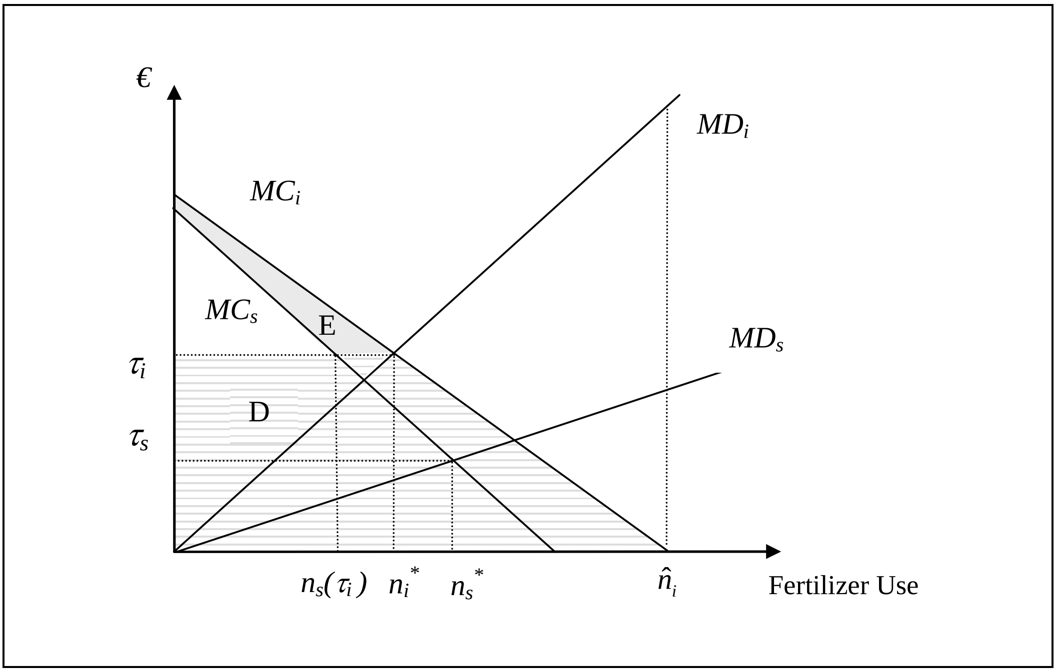

Figure 4: Instruments of a deposit refund system.

In the following section we illustrate the application of the deposit refund system by using representative data for farms located in the north-eastern part of Spain.

NUMERICAL ILLUSTRATION

In this section we portray the principal agricultural characteristics of the studied area and describe the data employed in the empirical part of this chapter. Furthermore, we specify the parameters and functions of the previously described economic model. Thereafter, we interpret the results of the solution of the model and conduct a sensitivity analysis of the results with respect to values of the marginal environmental damage from nitrogen emissions.

Data and the Area of the Study

Our empirical study is based in Aragon, an autonomous community in the north-eastern part of Spain and one of the main areas of intensive pig production. It accounts for 40% of the total Spanish swine population[5] while representing 54% of the total livestock production and 28% of the final value of the agricultural production (Iguácel 2006). The importance of the pig industry has been growing constantly in Aragon over the last decades, with more than 3.8 million places in 2003, and almost 4 million in 2004 (Iguácel et al. 2005).

For our empirical analysis we considered the operational costs of an average farm located in the study area and representing the behaviour of a farmer with two different yet related production activities: pig and crop production. The specified farm model reproduces the typical conditions of the region with respect to size and biophysical data, with corn being one of the principal crops (see Martínez 2002).

A numerical solution of the mathematical model (equations (3)–(6)) required the functions and parameters to be specified. Relevant data of nitrogen emissions and management costs of different technologies were collected from Iguácel (2006), Daudén et al. (2004) and Daudén and Quílez (2004). Corn production and nitrate leaching were estimated as a function of applied nitrogen, utilising biophysical data, which was previously generated with a process-oriented biophysical model, EPIC (Erosion Productivity Impact Calculator, Mitchell et al. 1998). Production costs, subsidies from the CAP and other data needed to estimate the net benefit functions were determined based on data published annually by the extension service of the Government of Aragon (1999-2007). Additionally, data for the selected region is available from an experimental farm of the Department of Soils and Irrigation (CITA, Government of Aragon)[6]. It includes data about crop production and applied nitrogen. The data generated on this farm allowed us to calibrate our model according to the conditions in the area of the study.[7]

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[5] After Germany, Spain has the second largest swine population in the European Union, representing 18% of its total production with a steadily growing trend over the last 10 years (Daudén and Quílez 2004).

[6] See Daudén et al. (2004) for more details with respect to the physical characteristics of the experimental farm.

[7] Details of the conditions of the experiments conducted can be consulted in Daudén et al. (2004) and Daudén and Quílez (2004).

Iguácel and Yagüe (2007) calculated the fixed and variable costs, and determined the efficiency with respect to nitrogen taken up by corn of the insecure and secure technologies. The remaining parameters of the model were determined based on data published annually by the extension service (Government of Aragon 2005, 2007).

Unfortunately, there is no regional data available to estimate water treatment costs as a function of nitrate concentration. Therefore, for simplicity, the function was specified linearly. The unitary cost for water treatment is 1.3 € per kg of nitrate and m^3 of water (Martínez 2002).[8] In Table 1.A of the appendix we present the values of the parameters employed in the empirical study. The identification of the parameters is based on the notation used in the previous section of the chapter. Table 2.A shows the coefficients for the quadratic nitrate leaching functions for each technology j, previously introduced as $n_j - g_j(n_j)$,. The parameters of the functions were estimated using the nonlinear least-squared regression procedure in SHAZAM (White 2002). The economic model was programmed with GAMS (Brooke et al. 1998) and solved numerically using the CONOPT solver.

RESULTS AND INTERPRETATION

The application of the economic model requires estimating the farm net benefits, obtained from pig and crop production, as a function of nitrogen, n , (pig slurry) based either on the secure or the insecure technology. The results are presented in Table 1 and show that the farm net benefit functions that best explain the underlying data are quadratic.[9]

Table 1. Benefit Functions with Secure and Insecure Technology

	Insecure technology	Secure technology
Intercept	295.88 (16.23)	70.756 (18.27)
Lineal coefficient (n)	6.0211 (5.31)	6.4396 (7.12)
Square coefficient (n^2)	$-0.79429 * 10^{-2}$ (-8.38)	$-0.79483 * 10^{-2}$ (-10.11)
Adjusted R^2	0.998	0.989

The T Statistics are shown in Parentheses.

After specifying the functions we solved problem (3) – (6) numerically utilizing GAMS. The results are summarized in Table 2. Farm net benefits present the net margin for the individual farmer while private welfare is derived from the farm net benefits by deducting the corresponding taxes and adding up the received subsidies in the case of secure applications. Social welfare before and after the implementation of the policy have to be identical by

[8] Foess et al. (1998) compared the cost of different processes applied in the USA to remove biological nutrients from water, and reported water treatment costs that range from 1.4 to 21 US$/$m^3$. The great cost discrepancy with respect to our cost can be explained in part because the costs considered here are independent of the pre-treatment nitrate level.

[9] We solved a farm decision model with GAMS where the amount of nitrogen that can be applied was restricted. The value of the objective function of this model provided the maximum farm net benefits. Different maximum farm net benefits were obtained by consecutively confining the amount of applied nitrogen from 450 kg/ha to 50 kg/ha. These maximum farm net benefits were then utilized to estimate the farm net benefits as a function of the amount of nitrogen applied.

definition. Hence, the implemented policy achieves its objective of establishing the optimal social outcome.

Table 2. Results with a Marginal Economic Damage of 1.3 € Per Kg of Nitrate Emissions Per M^3

Variables	**Insecure technology**	**Secure technology**
Farm net benefit (€/ha)	1432	1374
Nitrogen use from pig slurry (kg/ha)[10]	354	395
Nitrate emissions (kg/ha)	145.24	59.59
Social welfare (€/ha)	1243	1297
Input tax/subsidy (€/kg of nitrogen)	0.39 (tax)	0.23 (subsidy)
Farm net benefits in the presence of a tax/subsidy system (€/ha)	1292	1311
Social welfare in the presence of a tax/subsidy system (€/ha)	1243	1297

The results of Table 2 show that the farm net benefits with the secure technology are lower than the farm net benefits with the insecure technology. However, the socially optimal amount of applied nitrogen and social welfare are higher with the secure (less polluting) technology. Thus, in the absence of any environmental policy, farmers have no economic incentives to employ the more expensive but less polluting technology that is optimal from a social point of view. To overcome this inefficient private outcome we have designed the deposit/refund system described above so that the adoption of secure technologies is also optimal from a private point of view. Thus, in the presence of the deposit refund system not only does the social welfare have to be higher with secure than with insecure technology, but the private farm net benefits do as well.

In our numerical study, based on a marginal economic damage of 1.3 € per kg of nitrate, a tax of 0.39 €/kg with respect to the applied nitrogen and a subsidy of 0.23 €/kg (see Table 2) induce the socially optimal adoption of secure technology. As the regulator can observe the secure but not the insecure application of nitrogen, all farmers have to pay the tax. However, only the farmers that have adopted the secure technology receive the subsidy. To obtain information about whether individual farmers have adopted the secure technology or not, the regulator relies on the so-called authorized firms. These firms are responsible for the control of secure applications at the farm level.

An increase in water treatment costs from 1 to 2.5 €/kg of nitrate shows that the optimal values of the tax and subsidy have to be changed accordingly. The optimal taxes and subsidies as a response to a change in water treatment costs are presented in Figure 5. As the marginal economic damage of pollution rises, both taxes and subsidies increase. The difference between taxes and subsides increases in absolute terms as the marginal environmental damage increases. However, this difference stays the same in proportional terms since the marginal environmental damage is constant.

[10] Water Framework Directive establishes an upper limit for pig slurry application of 170 kgN/ha for vulnerable zones and 250 kgN/ha for the rest of the land. In this paper we have not considered such legal limitation.

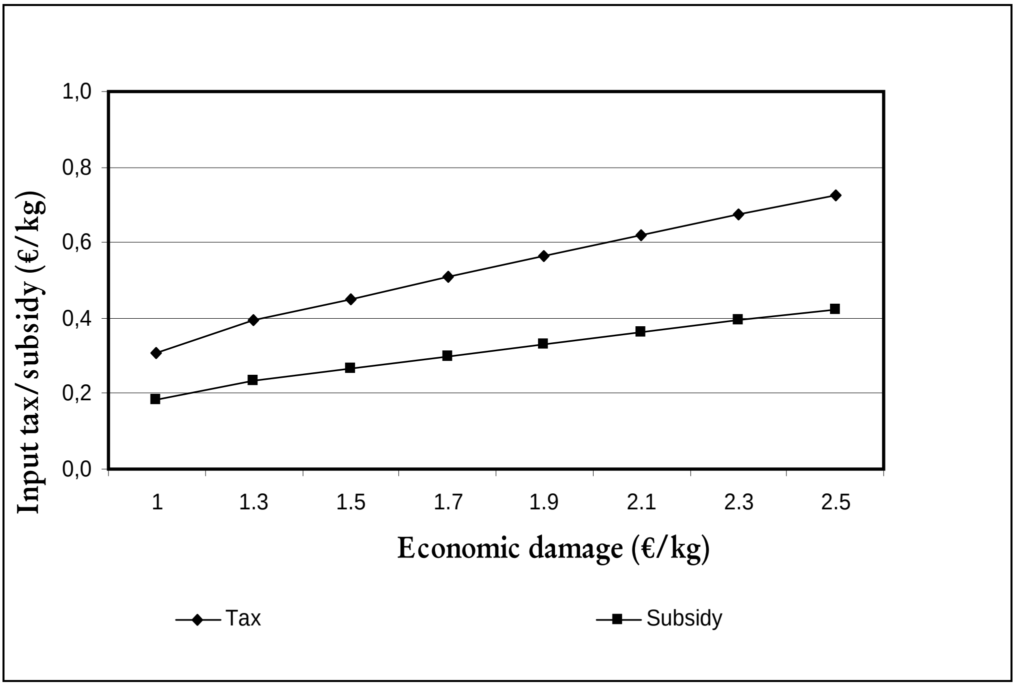

Figure 5: Effect of an increase in the marginal environmental costs of pollution on the tax and subsidy level.

CONCLUSIONS

The major problem of non-point pollution is that emissions are not observable at a reasonable cost. Within second-best regulation of inputs, little attention has been paid to the choice of the management practices or to economic incentives that provide signals to farmers to reduce the amounts of emissions by changing or modifying their farming practices.

In this study we propose a solution based on the combination of taxes and subsidies. In the absence of an environmental policy, farmers apply polluting inputs at socially inefficient rates and have no incentive to employ good environmental practices. The regulator can observe the total amount of polluting input applied but not whether the farmers follow those good practices or not. To distinguish between farmers who employ good environmental practices and those who do not, the regulator creates the figure of an authorized firm which is responsible for the controlled application of the input. The tax on the polluting input has to be paid by all farmers and the farmers who commission the service of the authorized firm receive a subsidy. This tax/subsidy schemes changes the incentives of farmers so that their privately optimal outcome coincides with the socially optimal outcome.

The proposed solution is illustrated for a farm that is representative of farms located the north-east of Spain. The model calculates the optimal level of the tax and subsidy. It thereby demonstrates how the adoption of good environmental practices can be achieved on a voluntary basis.

APPENDIX

Table 1A. Values of Parameters

Parameters	*Values*
p_p (€/place)	23.1
k_f (€/place)	18.27
u (m^3/place)	1.57
p_y (€/Tm)	124.83
p_s(€/kg)	2.82
p_i (€/kg)	1.98
c (€/ha)	650.23
s (€/ha)	343.52

Source: Government of Aragon (2005, 2007)

Table 2A. Leaching Function Coefficients.

	Insecure technology	Secure technology
Intercept	75.4 (4.80)	28.42 (5.97)
Lineal coefficient (n)	0.09126 (2.13)	0.034398 (6.27)
Square coefficient (n^2)	$0.299 * 10^{-3}$ (5.39)	$0.1127 * 10^{-3}$ (5.09)
Adjusted R^2	0.89	0.87

The T-Statistics are shown in Parenthesis

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REFERENCES

Brooke, A., Kendrick, D., Meeraus, A. and Raman, R., (1998). GAMS Tutorial by R. Rosenthal. GAMS Development Corporation. Washington.

Daudén, A. and Quílez, D. (2004). Pig slurry versus mineral fertilization on corn yield and nitrate leaching in a Mediterranean irrigated environment. European Journal of Agronomy 21, pp:7-19.

Daudén, A., Quílez, D., and Vera, M. V. (2004). Pig slurry application and irrigation effects on nitrate leaching in Mediterranean soil lysimeters. Journal of Environmental Quality 33, pp: 2290-2295.

European Environment Agency (2006). EEA Report No 2/2006 Integration of environment into EU agriculture policy—the IRENA indicator-based assessment report, Copenhagen.

Foess, G.W., Steinbrecher, P., Williams, K., and Garret, G.S. (1998). Cost and performance evaluation of BNR processes. Florida Water Resources Journal December: 11–16.

Fullerton, D., and Wolverton, A. (2000). Two Generalizations of a Deposit-Refund System, American Economic Review, Vol. 90, no. 2: 238-242.

Government of Aragon (1999). Base de datos 1T de superficie de cultivos por término municipal para Aragón 1987-99. Servicio de Estudios y Planificación. Secretaría General Técnica. Departamento de Agricultura. Gobierno de Aragón, Zaragoza.

Government of Aragon (2005). Anuario Estadístico Agrario de Aragón. Departamento de Agricultura y Alimentación.

Government of Aragon (2007). Anuario Estadístico Agrario de Aragón. Departamento de Agricultura y Alimentación.

Iguácel, F., Picot, A., and Gil, M. (2005). Resultados económicos del ganadero de porcino de cebo integrado. Informaciones Técnicas nº 155. Departamento de Agricultura y Alimentación. Gobierno de Aragón.

Iguácel, F. (2006). "Estiércoles y fertilización nitrogenada" en Orús F. coord. Fertilización Nitrogenada, Guía de actualización. Informaciones técnicas, número extraordinario. Departamento de Agricultura y Alimentación. Gobierno de Aragón.

Iguácel, F., and Yagüe, M. R. (2007). Evaluación de costes de sistemas y equipos de aplicación de purín (datos preliminares). Informaciones técnicas nº 178. Departamento de Agricultura y Alimentación. Gobierno de Aragón.

Mapp, H. P., Bernardo, D. J. Sabbagh, G. J., Geleta, S., and Watkins, K. B. (1994). "Economic and Environmental Impacts of Limiting Nitrogen Use to Protect Water Quality: A Stochastic Regional Analysis." *American Journal of Agricultural Economics* 76, no. 4 November: 889-903.

Martínez, Y. (2002). Análisis económico y ambiental de la contaminación por nitratos en el regadío. Ph.D. Dissertation. University of Zaragoza.

Mitchell, G., Griggs, R., Benson, V., and Williams, J. (1998). The EPIC model: environmental policy integrated climate. Texas Agricultural Experiment Station. Temple.

Larson, D. M., Helfand, G. E., and House, B. W. (1996). "Second-Best Tax Policies to Reduce Nonpoint Source Pollution." *American Journal of Agricultural Economics* 78, no. 4 November: 1108-17.

Segerson, K. (1988). "Uncertainty and Incentives for Nonpoint Pollution Control." *Journal of Environmental Economics and Management* 15, no. 1: 87-98.

Shortle, J. S., and Abler, D. G. (1998). Nonpoint pollution. *Yearbook of Environmental and Resource Economics* 1997/1998. Edited by H. Folmer, and T. Tietenberg. Cheltenham: Edwar Elgar.

Shortle, J. S., Horan, R. D., and Abler, D. G. (1998). "Research issues in nonpoint pollution control." *Environmental and Resource Economics* 11(3-4): 571-585.

Vickner, S. S., Hoag, D. L., Frasier, W. M., and Ascough, J. C. (1998). "A Dynamic Economic Analysis of Nitrate Leaching in Corn Production Under Nonuniform Irrigation Conditions." *American Journal Agricultural Economics* 80, no. 2 May: 397-408.

White K. J. (2002). SHAZAM - For Windows, Version 9.0.

Xepapadeas, A. P. (1991). "Environmental Policy under Imperfect Information: Incentives and Moral Hazard." *Journal of Environmental Economics and Management* 23, no. 2: 113 - 126.

Xepapadeas, A. P. (1992). "Environmental Policy Design and Dynamic Nonpoint-Source Pollution." *Journal of Environmental Economics and Management* 23, no. 1: 22-39.

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Chapter 15

PREFERENCES OF SPANISH CONSUMERS FOR ECOLABELLED FISH PRODUCTS*

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Abstract

Ecolabelled fish products, obtained under the sustainable fisheries certification programme, are proliferating in international food markets. This was boosted by the creation of the Marine Stewardship Council (MSC) in 1996 at the request of the WWF and the multinational company Unilever, whose function it is to accredit world fisheries sustainably managed in accordance with the directives put forward in the FAO's Code of Conduct for Responsible Fishing. The ecolabels guarantee consumers that a certain fish product comes from a fishery which conforms to regulations on sustainable fishing. And the principal factor which will determine the success or failure of ecolabelling is the acceptance of the products by the consumer.

This chapter aims to find out Spanish consumers' preferences for ecolabelled fish products. In order to do so, we have selected the national market's most highly-demanded fish products obtained by Spanish fleets, whose consumption has undergone a growth in trend in recent years. The results clearly show the preference of Spanish households for this type of product.

Keywords: seafood, ecolabelling, consumer` preferences.

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INTRODUCTION

Ecolabelling has experienced a notable growth in trend in recent years, especially in the forestry and agricultural sectors. In the specific case of fisheries, ecolabelling was boosted by the creation of the Marine Stewardship Council (MSC) in 1996 at the request of the WWF and Unilever. The MSC's present function is to accredit the world's fisheries which are sustainably managed in accordance with the FAO's Code of Conduct for Responsible Fishing. The evaluation of the fishery is carried out by a panel of independent experts not involved in the MSC. The accreditation programme makes it possible to obtain labels on which said information on sustainable fishing appears for the end consumer. Therefore, the aim of the MSC is to provide market-based incentives in order to improve world fishery management.

Ecolabels are, therefore, certificates given to fish products which have been obtained while generating the slightest possible impact on marine ecosystems[3]. They provide a guarantee for buyers and consumers that a certain fish product comes from a fishery which conforms to sustainable fishing regulations, thus allowing the consumer to exercise his/her environmental preferences when choosing a product.

On the other hand, ecolabels provide advantages for fishing companies and companies which distribute marine products. We could list the following, among others: they facilitate access to potential clients in ecological product markets; they make it possible to introduce improvements in fishery management, ensuring the long-term maintenance of exploitation (and, therefore, of fishing companies); they enable fish processing and distribution companies to become preferential suppliers; they imply a sales potential at significantly higher prices; they introduce improvements in the social recognition of “good practice”; and, consequently, they make it possible to improve the companies' image. The ecolabelling process, however, also generates an additional cost for economic agents: which must be borne when a certification process of these characteristics is being initiated. But, in any case, it would seem that the advantages deriving from the introduction of an ecolabelling system exceed the inherent costs, both in private terms (for the company itself) as well as in collective terms (i.e., in terms of fishery sustainability)[4]. And the success or failure of the ecolabelling process depends, in the last instance, on these products being accepted by the end consumer.

The increase in ecolabelling has led to an increase in the number of economic studies in recent years on these products being accepted by the end consumer. Among such studies we would underline the empirical papers by Nimon & Beghin (1999) and O`Brien & Teisl (2004) on forestry and agricultural products and the studies applied to fish products by Hannesson (1996), Holland & Wessels (1998), Wessels & Anderson (1995) and Wessels et al (1999). All of them tackle the empirical study of consumer preferences for possible ecolabelled products.

In this context, the aim of the study in this chapter is to understand Spanish consumer preferences for ecolabelled fish products. As such, the chapter is structured in the following manner: in epigraph 2 we look at the institutional frame in which ecolabelling practices and the certification process are included. In epigraph 3 the fish products to be studied are chosen and the questionnaire which has been designed for each of the products will be described.

[3] To date, aquaculture products are not subject to eco-certification on the part of the MSC.
[4] FAO (2005).

Then we will set out the results obtained on present consumption habits with regard to the products studied. In epigraph 5, we will comment on the results on Spanish consumers' preferences for certified products versus non-certified products. Lastly, in epigraphy 6, we sum up the most significant conclusions.

THE INSTITUTIONAL FRAMEWORK

At international level and in the face of the dynamic situation currently existing in ecolabelled fish product markets, the FAO (2005) has established a series of principles, criteria, minimum requirements and procedural aspects for the system by means of which products are ecolabelled which should be applied equally to all developing countries, countries in a transitional period and developed countries, among which we would underline the following: Firstly, the certifying body should be neutral and independent and have the technical capacity necessary in order to carry out the ecolabelling system in accordance with established regulations. Secondly, the initiative to establish an ecolabelling system can come from a government, an intergovernmental organisation, a non-governmental organisation, a fishing company or a private industry association. And, thirdly, the regulatory base for the legal framework of responsible fisheries is contemplated by the international fishing instruments made up, among others, of the United Nations Convention on the Law of the Sea of 1982 and the FAO's Code of Conduct for Responsible Fishing. Furthermore, it recognises that States and Regional Fisheries Organisations, where appropriate, can establish ecolabelling systems compatible with said regulations. The FAO has also differentiated between two types certification; fishery certification and the chain of custody certification, the latter covering the entire process from the moment the fish is caught until the moment the fish or fish product is sold to the end consumer.

Also, insofar as Europe is concerned, the precedents can be found in documents COM(1997) and COM(2002). In the former, the need to debate voluntary ecolabelling schemes which do not discriminate against products from developing countries is emphasised. However, and in spite of the fact that the Member States were in agreement as to the need to establish a common line within the EEC, an agreement on the way in which they should be implemented was not reached. In the latter, the Commission adopts an action plan for integrating environmental protection in Fishing Policy (CFP) and announces its intention to open a debate on the ecolabelling of fish products which is reflected in the document COM(2005) on which we will comment below. Furthermore, the fundamental aim of the reform of the CFP (EC 2371/2002) is to ensure the sustainable exploitation of marine resources from the socioeconomic and environmental point of view. In this context, the Commission itself recognises the need for progress in ecolabelling projects as a means of also progressing in the application of the reform of the CFP. Added to this is the concern of the Commission about matters related to competition, trade and consumer protection in the face of the growing proliferation of ecolabelled fish products in the markets of some European countries.

And all of this is reflected in EC document COM(2005) the fundamental aim of which is to initiate a debate within the EU on the possible implementation of voluntary ecolabelling schemes for fish products offered in EC markets. The EC poses three options on the possibility of introducing ecolabelling programmes in the EC ambit and depending on the degree of regulation which is to be applied in order to protect public interest. These options are as follows:

Option 1: No action. The ecolabelling of fish product programmes will continue to be introduced freely in the market of European countries, without any type of intervention on the part of the EC authorities[5].

Option 2: To create a single European fish product ecolabelling programme. Under this option, the EU would establish its own ecolabelling programme and the public authorities would be involved in all the stages of its development, running and control[6].

Option 3: To establish minimum requirements in voluntary ecolabelling programmes. In this option, programmes would be freely developed through private and/or public initiatives, complying with a series of minimum criteria to be defined by the EU authorities. Therefore, the role of the competent public authorities would be limited to registering the programmes and verifying compliance with the minimum requirements established.

To date, the debate in the EU is open and as a consequence no decision has been made yet on this matter.

Therefore, in spite of the recommendations made by the FAO and the timid and almost non-existing attempts made by the EU, at this moment in time the MSC is the only internationally-recognised institution that can carry out a programme for the certification of ecolabelled fish products. The MSC standard includes, in turn, a series of environmental principles designed to evaluate fishery management and sustainability, based on the FAO's Code of Conduct for Responsible Fishing. It is therefore necessary to define beforehand what is actually going to be certified. El MSC defines the "unit of certification" as a fishery or the combination of a stock, fishing method and fleet[7]. Once the unit of certification has been defined, it is necessary to verify compliance with the institution standard. In this way, we can sum up the environmental principles established in said evaluation standard as follows:

- Existence of fish biomass sufficient to guarantee the sustainability of the marine resource over time.
- Impact of the fishery (or unit of certification) in the marine environment or ecosystem.
- Management systems existing in the fishery (or unit of certification) subject to evaluation.

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[5] Given that, to date, the EU has adopted no measure in this ambit, this option corresponds to the present status quo.

[6] The EC itself considers that this option is not feasible because, in its opinion, it could generate contradictions between the policy of managing the marine resources in EC waters and the certification standards which are defined and, fundamentally, the high cost which adopting this option would suppose.

[7] It can imply the vessels which fish for a certain fish biomass as a whole or only a part of them.

After the evaluation process carried out by a group of independent experts and once these principles have been verified, the MSC issues the corresponding ecolabels. These labels will be valid for a period of five years[8].

But in the final instance, regardless of which the institution internationally legitimated to grant ecolabels in the sustainable fisheries is and of how their sustainability might be defined, the fundamental factor which will determine the success or failure of ecolabelling will be the extent to which ecolabelled products are accepted by the end consumer. If consumers do not choose these products in their shopping basket, the ecolabelling programme will not reach its end objectives. Therefore, it is fundamental to understand what consumer preferences are and, in particular, preferences for ecolabelled versus non-ecolabelled fish products.

PRODUCT SELECTION AND DESIGN OF THE QUESTIONNAIRE

In order to find out what Spanish consumer preferences for ecolabelled fish products are, it is necessary to identify the products which are susceptible to being certified beforehand. In this study we have selected species which, captured in the main by the different Spanish fishing fleet segments, are highly-demanded in the national market. This information can be obtained per categories or groups of products from the Ministry of Agriculture, Fisheries and Food (2005). In particular, in the fresh product category, hake and sardines/anchovies stand out, with a consumption of 21.5% and 19.0%, respectively, in the group of fresh fish products. With regard to hake, at 46.5% it stands out over the total amount of frozen fish. Lastly, with regard to the canned product, tuna is the most highly-consumed product, representing 53.3% of the total amount of canned fish. The remaining species are far from reaching these percentages insofar as Spanish consumption is concerned.

With respect to the average variation in consumption in Spanish households, and heeding the same source, the fish products which experienced the highest level of growth over the period 1999-2004 were: hake and tuna (4.0 and 5.3% respectively) in the fresh fish product category; cod, salmon and sole (19.9%; 14.9%; 8.1%, respectively) in the frozen category; and tuna (4.0%) in the canned category.

On the basis of such data, the products most demanded by Spanish consumers and which, furthermore, have experienced a higher level of growth in recent years are tuna and hake. These products will be included in the study to evaluate consumer preferences for ecolabelled products, differentiating between the fresh/canned and fresh/frozen versions, respectively. The study also includes the sardine (both fresh and canned) as it is a highly-consumed product in Spain and in spite of the fact that its annual variation rate in the period considered was not relevant, in 2004 it experienced a significant increase (6.3% in the fresh version).

Consequently, the products which are the subject of the study will be *tuna (fresh/canned), hake (fresh/frozen) and sardine (fresh/canned).* On the other hand, in the cases of tuna and the sardine, as well as differentiating between the fresh and canned versions we will consider where these species have been caught, differentiating between products caught in Spanish fishing grounds or nearby grounds and those caught in far-off waters. This is due to the fact that in the Spanish market products obtained by non-resident fleets are also commercialised,

[8] Before this period ends, it is necessary to carry out a new evaluation of the fish product in order to maintain said certification without interruption.

especially in the canning sub-sector. In the case of hake, we will look at the fresh and frozen options, given that most of the hake commercialised in the Spanish market is caught by Spanish fleets, either in nearby fishing grounds or far-off waters (and, therefore, it would not seem advisable to include the option of their origin in the analysis) and the possibilities of substituting one for the other.

In order to find out what the preferences of Spanish households for these types of products are, a questionnaire which takes into account the singularities of the products under study has been designed and, consequently, three standard questionnaires have been used. The questionnaires have basically been divided up into the following sections:

- General data on fish consumption: frequency of consumption, cost of the products selected for the study, type of establishment where the purchase is made, etc.
- Specific questions on ecolabelled products, among which the following stand out: willingness to consume such products, the amount buyers are prepared to pay for the ecolabelled products studied, etc.
- Information on the ecological consumption of those polled: concern for the environment, whether or not they belong to environmentalist groups, preference for ecological products which are environmentally friendly, etc.
- Socioeconomic data: city of residence, number of family members, academic record, relation with the fisheries' sector, family income, etc.

On the other hand, the size of the sample (1500 questionnaires completed) has been defined on the basis of the variables with regard to the Spanish population age pyramid and geographical location of the Spanish provinces (coastal/inland).

RESULTS ON PRESENT CONSUMPTION HABITS

1517 were finally obtained, with a reply rate (questionnaires completed/those contacted) of 51%. They were distributed in the following manner: 501 questionnaires correspond to tuna, 503 to hake and 513 to sardines. The sample corresponds to 720 municipalities from the fifty-two Spanish provinces and the interviews were carried out over the telephone. All of those interviewed were fish buyers and consumers (a necessary condition if the interview was to continue to be carried out), of which 80.3% were women, the average age of those polled being 47.3. In Table 1 we are shown the representativity of the sample in relation with the age pyramid and the geographical distribution of the Spanish population. As can be seen in said Table, the sample adapts quite well to the population distribution, indicating a high representativity; only modified slightly with regard to the age criterion, especially for the 18-34 years of age segment.

Table 1. Analysis of Representativity in Age and Geographical Location

Age groups	**Sample percentage (design)**	**Sample percentage (field)**
(18-34)	33.19	29.50
(35-49)	26.65	28.00
(50-69)	25.63	27.80
(>69)	14.53	14.70
TOTAL	100	100
Geographical location of provinces (coastal / inland)	**Sample percentage (design)**	**Sample percentage (field)**
Coastal	59.39	59.30
Inland	40.61	40.70
TOTAL	100	100

With respect to the socioeconomic characteristics of those polled, it is noticeable that in 45.4% of the households polled the income is contributed by two people, whereas in 41.8% of the households the income is contributed by a single person. With relation to academic record, 34.3% of those polled had completed primary education whereas only 17.1% had completed secondary education (it is worth remembering that, as we commented on above, the interviews took place with fisher buyers and consumers and, of these, 80.3% were women). Furthermore, 28.4% of those polled were full-time employees and 23.6% stayed at home to look after domestic affairs. Also, the highest percentage of those polled (11.2%) declared that their gross income was between 1000 and 1499 euros per month gross.

On the other hand, and in relation with the make-up of the households, the sample average was 0.88 under-18s per household, 2.33 people between 18 and 65 and 0.56 over 65s. 47% of the households in the sample did not have any under-18s; 24.7% lived with an under-18 and 23.2% with two under-18s (only 5.1% of the families had more than two under-18s in their charge). In the case of the over-65s, 63.1 % in the sample did not live with a person over the age of 65 and 16% with two. And only 1.3% of the households was made up of more than two over-65s.

If we analyse the data on fish consumption habits in the sample (see Table 2), the majority of those polled (82.2%) acquire fresh products and the establishments used in the main by those polled in which to buy their fish products are the market or the fish market on the quay (39.9%) and the supermarket (36.0%). As can be seen in Table 2, fish consumption is notably high; in the main 79% of those polled say they consume fish once or more a week. On the other hand, the average weekly expenditure of those polled on food is 98.10 euros, of which 26.44 euros is spent on fish (approximately 27% of the weekly expenditure on food).

Table 2. Place and Means of Buying Fish and Frequency of Consumption

Means Of Purchase	**Number Of Cases**	**Percentage**
Fresh	1247	82.2
Frozen And Loose	171	11.3
Frozen And Packed	71	4.7
Precooked And Frozen	10	0.7
Canned	5	0.3
Does Not Know/Does Not Answer	13	0.9
Total	1517	100.0
Place Of Purchase	**Number Of Cases**	**Percentage**
In The Market /Quayside /Marketplace	606	39.9
In The Supermarket	546	36.0
In Both Places, With Equal Frequency	148	9.8
In The Local Shop/Fish Shop	203	13.4
Others	14	0.9
Total	1517	100.0
Frequency Of Consumption	**Number Of Cases**	**Percentage**
Daily	183	12.1
Once A Week Or More	1203	79.3
Once A Fortnight	106	7.0
Once A Month	15	1.0
Once Every Two Or Three Months	5	0.3
Once Every Six Months	1	0.1
Less Than Twice A Year	3	0.2
Another	1	0.1
Total	1517	100.0

With regard to the ecological behaviour of those polled (Table 3), the majority are concerned with environmental problems. It is clear that a large majority avoid buying products which are non-environmentally friendly (84.2% of those polled), follow nature documentaries (81.9%), use energy-efficient light bulbs (72.8%), acquire energy-efficient domestic appliances (69.5%) and have visited a nature park in the last year (65.7%).

Lastly, only 9.3% of those polled stated that they knew of an over-exploited fishing ground. The replies are extremely diverse insofar as the species and location of the fish banks under threat are concerned, but of the fishing grounds in danger that they mention, the one that appears most frequently is the anchovy fishery (in particular, the northern Spain-Cantabrian fishing ground, this probably being due to public awareness of the moratorium introduced in recent months with regard to this species), followed at a distance by tuna and cod.

Table 3. Data on the Ecological Behaviour of the Sample

	Number of cases	Percentage
Belongs to an environmentalist group	64	4.2
Reads nature magazines	607	47.0
Watches nature documentaries	1243	81.9
Uses energy-efficient light bulbs	1104	72.8
Uses energy-efficient domestic appliances	1055	69.5
Avoids buying non-environmentally friendly products	1278	84.2
Tries not to buy products from environmentally irresponsible companies	1198	79.0
Has stopped buying products which harm the environment	796	52.5
Recreational activities	827	54.5
Has visited a nature park in the last year	997	65.7

On the other hand, in relation with consumption habits, the results obtained in the sample show slight differences between the products studied (see Table 4). In particular, of the 501 people who make up the sample to whom the questionnaire on *tuna consumption* was presented, 74.5% of those polled affirmed that they had bought tuna in the last month. Of these, 34% bought the fresh product and a large majority, 66%, bought the canned product, which enables us to rule out or, at least, limit the possible seasonal nature inherent to the exploitation of this highly migratory species (in the case of the fresh product).

Table 4. Frequency (%) of Tuna, Hake and Sardine Consumption

	TUNA		HAKE		SARDINES	
	Fresh	Canned	Fresh	Frozen	Fresh	Canned
Daily	0.8	22.2	4.2	3.2	1.2	1.2
Once a week or more	12.6	54.3	26.8	18.5	23.2	17.9
Once a fortnight	11.0	9.2	18.5	10.5	17.2	14.2
Once a month	16.6	6.0	20.5	16.1	23.4	15.0
Once every two or three months	10.6	0.6	8.2	7.2	8.8	10.3
Once every six months	4.8	0.6	2.2	3.0	3.7	3.9
Less than twice a year	10.2	0.6	5.0	5.0	4.1	3.3
Another	22.2*	4.0	8.9	24.7*	11.9	26.7*
Does not know/does not answer	11.4	2.6	5.8	11.9	6.6	7.4
Total	100.0	100.0	100.0	100.0	100.0	100.0

* The answers contained in this option are from consumers who state that they never consume the product in such a version.

With relation to the frequency of consumption of this product, 22.2% stated that they never buy fresh tuna and around 17% stated that they buy it at least once a month (see Table 4). On the other hand, canned tuna is more frequently consumed by those polled, as the majority affirm that they buy this product at least once a week (54.3%).

In respect of the results on *hake consumption*, of the 503 people who make up the sample, 64% of those polled affirmed that they had bought hake in the last month. And of these, 25.8% bought the frozen product whereas the large majority, 74.2%, bought the fresh product.

Also, a notable percentage of those polled (26.8%) stated that they consume fresh hake once or more a week, while in the case of the frozen product, the percentage drops to 18.5%. In the latter case, the percentage of those polled who state that they never consume this product is significant.

Lastly, with regard to the results on *sardine consumption,* of the 513 people who make up the sample to whom the questionnaire was presented for this product, 51.3% of those polled affirmed that they had bought sardines in the last month. Of these, the majority (89.3%) bought the fresh product and 10.7% bought the canned product.

With respect to the frequency of sardine consumption, around 23% of those polled stated that they consume fresh sardines once a week or more and a similar percentage stated that they consume this product once a month. In the case of the canned product, a significant percentage (26.7%) stated that they never consume this product.

CERTIFIED VERSUS NON-CERTIFIED FISH PRODUCTS

In order to find out consumer preferences for the certified products studied, in the questionnaire carried out we compared the purchase options between the certified and the non-certified product, both at a same price as well as at different prices, the price of the certified product always being higher.

Table 5. Purchase Choices for Tuna (Assuming that A and B are the Same Price)

TUNA			
Choice A	Choice B	Preferred option	Porcentage
Certfied tuna (A)	Non-certified tuna (B)	A	90.6
		B	1.4
		No answer	8.0
Tuna form nearby fishing grounds (North Atlantic/Mediterranean)	Tuna form far-off fishing grounds (S. Atlantic/Pacific/Indian)	A	73.9
		B	4.0
		No answer	22.2
Canned tuna, non-certified, from nearby fishing grounds	Canned tuna, certified, from far-off fishing grounds	A	29.5
		B	46.9
		No answer	23.6

In the first case (same prices) and with relation to the results of *tuna consumption choices* (see Table 5), those polled prefer, by an immense majority (90%), *certified tuna* and they also largely prefer *a nearby origin* (73.9%). In the third option, they were asked to “sacrifice” one of their favourite options, that is, or certification or origin, and it can be observed that the certification option triumphed over the proximity of the fishing ground option (47% as opposed to 30%). This third option demonstrates a strong preference for certification.

With relation to the results on *hake consumption choices* (Table 6), the wide majority (74.2%) prefer *certified fresh hake* (90%). In the third option they were asked to choose between their favourite options and, in contrast to the case of the tuna, there is less indecisiveness among those polled and a significant majority, 62%, prefer to reject the fresh product ahead of rejecting certification. Finally, it is clear that the option preferred by the majority, 88%, is for certified fresh hake.

Table 6. Purchase Choices for Hake (Assuming that A and B are the Same Price)

HAKE			
Choice A	Choice B	Preferred option	Porcentage
Frozen hake (A)	Fresh hake (B)	A	24.1
		B	74.2
		No answer	1.8
Certified hake	Non-certified hake	A	89.9
		B	5.0
		No answer	5.2
Certified frozen hake	Non-certified frozen hake	A	62.0
		B	30.6
		No answer	7.4
Certified fresh hake	Non-certified frozen hake	A	87.7
		B	8.2
		No answer	4.2

Lastly, and with respect to the results on *sardine consumption choices*, the wide majority (90%) also prefer *certified sardines* and also (82%) *from Spanish fishing grounds*. In the third option, they were asked to choose between their favourite options and here a large amount were indecisive (44%)[9] and, of the rest, a large percentage corresponds to those who reject origin in Spanish fishing grounds and choose certification (44% as opposed to 36%), although this is the case where the difference is lower.

Table 7. Purchase Choices for Sardine (Assuming that A and B are the Same Price)

SARDINES			
Choice A	Choice B	Preferred option	Percentage
Certified sardines	Non-certified sardines	A	90.6
		B	2.3
		NC	7.0
Canned sardines from Spanish fishing grounds	Canned sardines from other fishing grounds	A	82.1
		B	2.5
		NC	15.4
Non-certified canned sardines from Spanish fishing grounds	Certified canned sardines from other fishing grounds	A	36.3
		B	44.1
		NC	19.7

[9] We could interpret this to mean that origin and certification is equally important to them.

In the case of different price choices, the results obtained are shown in Table 8. With relation to the results for the *fresh tuna*, those polled had to choose between the certified and non-certified product where the price for the former was higher and with four price increases in euros/Kg (0.5; 1; 2; 3 euros, respectively, on top of the present market price). The results show that the probability of accepting the price increase is lower as said price increases, as was foreseeable. Also, the percentage of individuals who do not participate in the valuation exercise (don’t know/don’t answer) is low (under 6%), which reflects an understanding and acceptance of the economic valuation scenario proposed. On the other hand, the analysis of the last two rows in the table indicates that 78% of those polled accepted paying the proposed price increase. It particularly comes to our attention that for the highest price (an increase of 3 euros per kilo on top of the market price), more than 64% of those polled would accept having to pay this increase in order to buy certified fresh tuna.

With relation to *certified versus non-certified canned tuna*, those polled had to choose between the certified and the non-certified product with a price higher for the former and with four price increases in euros per 200gr.-can (0.1; 0.25; 0.5; 0.75 euros, respectively). It can be observed that the probability of accepting the price increase is lower as said price increases (with the exception of the payment of the second price). The percentage of individuals who do not participate in the valuation exercise (don’t know/don’t answer) is also in this case low (under 7%). An analysis of the last two rows in the table shows us that 85.8% of those polled accepted paying the proposed price increase. For the highest price (increase of 0.75 euros per 200gr.-can on top of the market price), approximately 79% of those polled would accept having to pay this in order to buy certified canned tuna.

Table 8. **Prices and Choices for Certified Versus Non-Certified Fresh and Canned Tuna.**

Fresh tuna			**Choices**			**Total**
			Certified tuna	**Non-certified tuna**	**Doesn’t know/ Doesn’t answer**	
Prices	0.50	Percentage	87.8%	9.2%	3.1%	100.0%
	1.00	Percentage	84.0%	11.5%	4.6%	100.0%
	2.00	Percentage	73.3%	21.5%	5.2%	100.0%
	3.00	Percentage	64.4%	30.8%	4.8%	100.0%
Total		Percentage	78.0%	17.6%	4.4%	100.0%
Canned tuna			**Choices**			**Total**
			Certified tuna	**Non-certified tuna**	**Doesn’t know/ Doesn’t answer**	
Prices	0.10	Percentage	89.3%	8.4%	2.3%	100.0%
	0.25	Percentage	90.1%	3.8%	6.1%	100.0%
	0.50	Percentage	87.4%	8.1%	4.4%	100.0%
	0.75	Percentage	78.8%	17.3%	3.8%	100.0%
Total		Percentage	86.8%	9.0%	4.2%	100.0%

In the case of *certified versus non-certified fresh hake,* those polled had to choose between the two options with a higher price for the former and with four price increases in euros/kg (1; 2; 3; 4 euros. See Table 9). In this case, it can also be observed that the probability the price increase be accepted is lower as said price increases (with the exception of the payment of the third price). Furthermore, an analysis of the last two rows of the table shows that 77% of those polled accepted paying the proposed price increase. For the highest price (increase of 4 euros per kilo on top of the market price), approximately 73% of those polled would accept having to pay the increase in order to buy certified fresh hake.

Table 9. Prices and Choices for Certified Versus Non-Certified Fresh/Frozen Hake

Fresh hake			Choices			Total
			Certified Hake	Non-certified hake	Doesn't know/ Doesn't answer	
Prices	1.00	Percentage	92.1%	5.5%	2.4%	100.0%
	2.00	Percentage	74.2%	22.7%	3.1%	100.0%
	3.00	Percentage	68.9%	28.0%	3.0%	100.0%
	4.00	Percentage	73.3%	24.1%	2.6%	100.0%
Total		Percentage	77.1%	20.1%	2.8%	100.0%
Frozen hake			**Choices**			**Total**
			Certified Hake	**Non-certified hake**	**Doesn't know/ Doesn't answer**	
Prices	0.50	Percentage	86.6%	8.7%	4.7%	100.0%
	1.00	Percentage	87.5%	8.6%	3.9%	100.0%
	2.00	Percentage	65.2%	29.5%	5.3%	100.0%
	3.00	Percentage	67.2%	25.0%	7.8%	100.0%
Total		Percentage	76.7%	17.9%	5.4%	100.0%

With respect to *certified versus non-certified frozen hake*, also with four price increases in euros/kg (0.5; 1; 2; 3 euros, respectively), the trend of the probability that the price increase be accepted drops as said price increases, although in this case there is a slight increase in the second and fourth prices with respect to the above. Similarly to the previous cases, approximately 77% of those polled accepted paying the proposed price increase. And for the highest price (increase of 3 euros per kg on top of the market price), around 67% of those polled would accept having to pay the increase in order to buy certified frozen hake.

Lastly, in the case of *certified versus non-certified fresh sardines,* those polled had to choose between certified and non-certified fresh sardines with a higher price for the former and with four price increases in euros/kg (0.5; 1; 1.5; 2 euros, respectively. See Table 10). The results obtained show that the trend of the probability that the price increase be accepted shows a clear drop as said price increases, and approximately 79% of those polled accepted paying the proposed price increase. For the highest price (increase of 2 euros per kg on top of the market price), approximately 68% of those polled would accept having to pay the increase in order to buy certified fresh sardines.

Table 10. Prices and Choices for Certified Versus Non-Certified Fresh/Canned Sardines

Fresh sardines			**Choices**			**Total**
			Certified sardines	**Non-certified Sardines**	**Doesn't know/ Doesn't answer**	
Prices	0.50	Percentage	83.5%	14.3%	2.3%	100.0%
	1.00	Percentage	81.9%	13.4%	4.7%	100.0%
	1.50	Percentage	80.5%	15.0%	4.5%	100.0%
	2.00	Percentage	68.3%	24.2%	7.5%	100.0%
Total		Percentage	78.8%	16.6%	4.7%	100.0%
Canned sardines			**Choices**			**Total**
			Certified Sardines	**Non-certified Sardines**	**Doesn't know/ Doesn't answer**	
Prices	0.10	Percentage	80.5%	13.5%	6.0%	100.0%
	0.25	Percentage	87.4%	4.7%	7.9%	100.0%
	0.50	Percentage	85.0%	9.8%	5.3%	100.0%
	0.75	Percentage	80.8%	10.0%	9.2%	100.0%
Total		Percentage	83.4%	9.6%	7.0%	100.0%

And with respect to *certified versus non-certified canned sardines*, we also posed four price increases in euros per 200gr.-can (0.1; 0.25; 0.5; 0.75 euros, respectively). In this case, the trend of the probability that the price increase be accepted shows a clear drop as the price increases, complying as in the previous cases with economic theory predictions, and approximately 83% of those polled accepted paying the proposed price increase. For the highest price (increase of 0.75 euros in a 200 gr.-can on top of the market price), 81% of those polled would accept having to pay the increase in order to buy certified canned sardines.

CONCLUSION

The ecolabelling of fish products is increasing in international markets as a means of improving the sustainability of fish populations and marine ecosystems. Acceptance of these products depends in the last instance on the consumer; if consumers do not choose these products ecolabelling will not achieve its objectives.

In the study presented in this chapter, we have studied the preferences of Spanish consumers for this type of product, choosing products which feature highly in the average family's shopping basket and which, furthermore, have experienced a notable increase in the five years considered in the study. In all of the cases studied, there is a clear preference of Spanish consumers for ecolabelled fish products: fresh and canned tuna; fresh and frozen hake; and fresh and canned sardines. Although slight differences exist between the cases studied in relation with the singularities of each product, around 70% of the sample would accept having to pay the highest increase proposed (on top of the present market price) for the ecolabelled product.

With respect to the specific case of tuna, an immense majority of Spanish consumers (90%) show a preference for the certified product, and with a higher intensity (73.9%) opt for the nearby origin. As was foreseeable, both insofar as the fresh and canned product is concerned, the preference for the certified product decreases as the price increases; this trend is clearer for the canned version, the consumption of which among those polled is higher than the fresh version.

In the case of hake, there is a clear preference for the fresh (74.2%) and certified (90%) product. And in both cases (fresh/frozen) the preference for the certified product once again decreases as the price increases. This trend is clearer in the fresh version. We can also observe a decided willingness to pay for the certified product, even where the highest price is concerned, 73% of the sample would accept having to pay an increase of 4 euros/kilo for fresh hake and 67% of the sample would accept having to pay an increase of 3 euros/kilo for the frozen product.

Lastly, in the case of the sardine, the majority of those polled prefer certified sardines (90%) and from Spanish fishing grounds (82.0%). And both in the case of fresh sardines as well as the canned product, most of those polled would accept having to pay a price increase for the certified product.

Therefore, of the results obtained in this study, it can be concluded that there exists a potential demand for this type of product in the Spanish market. This is in keeping with the growing concern of the Spanish population for the deterioration of the environment and the sustainability of marine ecosystems. All of this would seem to indicate that it is possible to use market-based incentives to modify the conduct of economic agents towards practices which show greater respect for the environment and its sustainability.

REFERENCES

COM(1997)719. *The future for the market on fisheries products in the EU: responsibility, partnership and competitiveness.* Brussels: European Commission (16.12.1997).

COM(2002)186. *Community Action Plan to integrate environmental protection requirements into the CFP.* Brussels: European Commission (28.05.2002).

COM(2005)275. *Launching a debate on a Community approach towards eco-labelling schemes for fisheries products.* Brussels: European Commission (29.06.2005).

European Commission 2371/2002. *The Reform of CFP*. Brussels: European Commission (20.12.2002).

FAO (2005). *Guidelines for the Ecolabelling of Fish and Fishery Products from Marine Capture Fisheries.* Roma: Food and Agriculture Organization of the United Nations.

Hannesson, R. (1996). *Fisheries mis-management: The case of the North Atlantic cod.* Oxford: Fishing News Books.

Holland, D & Wessells, C.R. (1998). Predicting consumer preferences for fresh salmon: The influence of safety inspection and production method attributes. *Agricultural and Resource Economics Review*, 27, 1-14.

MAPyA (2005). *La Alimentación en España, 2004*. Madrid: Ministerio de Agricultura, Pesca y Alimentación.

Marine Stewardship Council (2006). The MSC Fishery Assessment & Certification Process. www.msc.org

Nimon, W. & Beghin, J. (1999). Are Eco-labels valuable? Evidence from the Apparel Industry. *American Journal of Agricultural Economics,* 81, 801-811.

O'Brien, K.A. & Teisl, M.F. (2004). Eco-information and its effect on consumer values for environmentally certified forest products. *Journal of Forest Economics,* 10, 75–96.

Wessels C.R. & Anderson, J.G. (1995). Consumer willingness to pay for seafood safety assurances. *Journal of Consumer Affairs, 29(1),* 85-107.

Wessells, C.R., Donath, H. & Johnston R. (1999). *US consumer preferences for ecolabeled seafood: results of a consumer survey*. Rhode Island: University of Rhode Island.

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Chapter 16

ECOLOGICAL PRODUCTIVITY: DEFINITION AND APPLICATIONS

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ABSTRACT

The discussion on environmental problems and economic growth is closely related to the concept of sustainable development. The best known definition is given by the World Commission on Environment and Development - the Brundtland Commission, which promoted closer links between the environment and development. They emphasized issues of social and economic sustainability. Sustainable development is now featured as a goal in dozens of national environmental policy statements, and in the opening paragraphs of Agenda 21 adopted by the Earth Summit in Rio de Janeiro in June 1992. The Johannesburg World Summit 2002 had a renewed political commitment to Agenda 21. The Brundtland report's Our Common Future defines sustainable development as "development that meets the needs of the present without compromising the ability of future generations to meet their own needs". This definition is ambiguous and raises more questions than it answers. A more precise definition would be, for example, requiring utility levels, or resource stocks, or total capital stocks including natural capital and human capital to be non-decreasing over time. Thus, sustainable paths confront standard optimal solutions as formalized in the traditional theory of economic growth. Key point for sustainable development is continuous technological improvements or productivity progress. This chapter provides measures from theoretical and empirical models. Additionally, empirical results and their interpretations are provided.

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Whether pollution abatement technologies are used most efficiently is crucial in the analysis of environmental management because it influences, at least in part, the cost of alternative production and pollution abatement technologies. The role of environmental policy in encouraging or discouraging productivity growth is also well documented in the theoretical literature. As a result of this policy, two possibilities are likely. First, abatement pressures may stimulate technological innovations that reduce the actual cost of compliance below those originally estimated (e.g., Jaffe, Newell, and Stavins, 2003). Second, firms may be reluctant to innovate if they believe that regulators will respond by 'ratcheting-up' standards. In addition to the changes in environmental regulations and technology, management levels also influence ecological productivity. Therefore, whether the productivity and technological frontier expands over time is an empirical question. The principal focus of this chapter is to measure total factor productivity within a joint-production model that considers both market and environmental pollution variables, and then provide an analysis of environmental policy and productivity, especially employing an example in China.

1. INTRODUCTION

The discussion on environmental problems and economic growth is closely related to the concept of sustainable development. The best known definition is given by the World Commission on Environment and Development - the Brundtland Commission, which promoted closer links between the environment and development. They emphasized issues of social and economic sustainability. Sustainable development is now featured as a goal in dozens of national environmental policy statements, and in the opening paragraphs of Agenda 21 adopted by the Earth Summit in Rio de Janeiro in June 1992. The Johannesburg World Summit 2002 had a renewed political commitment to Agenda 21. The Brundtland report's Our Common Future defines sustainable development as "development that meets the needs of the present without compromising the ability of future generations to meet their own needs". This definition is ambiguous and raises more questions than it answers. A more precise definition would be, for example, requiring utility levels, or resource stocks, or total capital stocks including natural capital and human capital to be non-decreasing over time. Thus, sustainable paths confront standard optimal solutions as formalized in the traditional theory of economic growth. Key point for sustainable development is continuous technological improvements or productivity progress. This chapter provides measures from theoretical and empirical models. Additionally, empirical results and their interpretations are provided.

The principal focus of this chapter is to provide measurement methods of productivity change which include total factor productivity, and technological/efficiency change for environmental (*i.e.*, nonmarket) outputs. Additionally, this paper measure market productivity following the traditional productivity literature. It is important to note that the regulations requiring more stringent pollution abatement do not necessarily change ecological productivity. This is so because the linear expansion of pollution abatement costs and pollution reduction does not necessarily change the pollution reduction per abatement cost.

Whether pollution abatement technologies are utilized more efficiently is crucial in the analysis of environmental management because it influences the cost of alternative production and pollution abatement technologies, at least in part (e.g., Jaffe, Newell, and Stavins, 2003). It is well documented in theoretical literature about the role of environmental policy in encouraging or discouraging productivity growth. As an effect of the policy, two possibilities are concerned. Abatement pressures may stimulate technological innovations that reduce the actual cost of compliance below those originally estimated. The other possibility is that firms may be reluctant to innovate if they believe regulators will respond by 'ratcheting-up' standards even further. In addition to the changes in environmental regulations and technology, management levels also influence ecological productivity. Therefore, whether the productivity and technological frontier level increases over time is an empirical question. Most current empirical studies focus on developed countries. To the authors' knowledge, there are no existing studies that have estimated the efficiency changes of environmental technology or management in developing countries with the exception of our former studies (see Managi and Kaneko, 2005). This chapter provides a brief review of existing studies.

The chapter is structured as follows. Section 2 discusses a basic concept of productivity. Section 3 provides an idea of environmental or ecological productivity. Section 4 presents the review and discussion of environmental policy and productivity.

2. PRODUCTIVITY CONCEPT

Productivity is a key concept of production theory. In general, the idea of measuring changes in productivity of a firm, industry, and country is based on comparison of its performance in one period relative to another. In other words, a productivity index is defined as the ratio of an index of output growth divided by an index of input growth and the inputs over two accounting periods. I focus on the comprehensive productivity measure known as total factor productivity (TFP). This measure attempts to include all outputs and all inputs used in the production process. It gives a more accurate picture of performance than partial productivity measures such as labor productivity. Changes in the TFP index tell us how the amount of total output that can Figure 2 be produced from a unit of total input has changed over time. If the firm produces only one output using one input then a simple measure of productivity change is:

$$TFP = (y_{t+1}/y_t) / (x_{t+1}/x_t), \quad (1)$$

where the superscripts *t*, *t+1* indicate the time periods in which x and y are observed. This is also the ratio of the average products of the two periods.

There are two basic approaches to the measurement of productivity change: the econometric estimation of a production, cost, or some other function, and the construction of index numbers using nonparametric methods. Most general approach is the latter approach because it does not require the imposition of a possibly unwarranted functional form on the structure of production technology as required by the econometric approach.

Nonparametric frontier technologies approaches called Data Envelopment Analysis (DEA) can be used to quantify productivity change, and can be decomposed into various constituents, as described below. TFP includes all categories of productivity change, which decomposed into technological change, or shifts in the production frontier, and efficiency change, or movement of inefficient production units relative to the frontier (e.g., Färe *et al*., (1994). Fig. 1 illustrates a two-output example of DEA[2]. The output vectors represented by points A and B are on the efficient frontier, so the associated distance functions equal 1. Output vector C is not on the efficient frontier and, thus, the distance function associated with observation C is less than one. The value of the distance function for C is calculated by extending both outputs by the scalar φ, which is increased until the associated projection reaches the efficiency frontier, illustrated by point C*. Using the first set of constraints in (3), DEA determines C* as maximum radial expansion of C that does not exceed the linear combination of the adjacent efficient output vectors.

$$\mathrm{C}^* = \max_{\varphi, \lambda_{\mathrm{A}} \lambda_{\mathrm{A}}} \varphi \mathrm{C}$$
$$\text{s.t.} \varphi \mathrm{C} \le \lambda_{\mathrm{A}} \mathrm{A} + \lambda_B \mathrm{B}$$

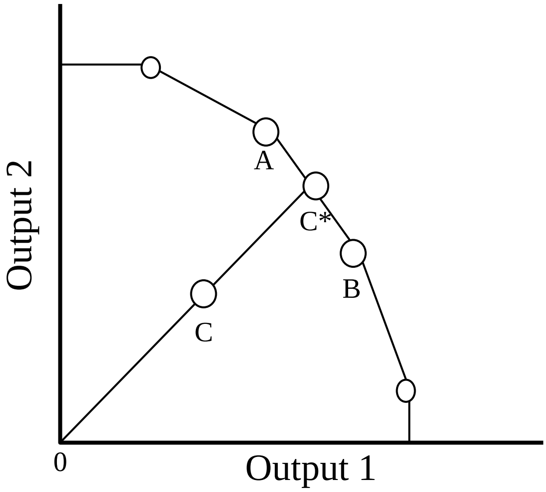

Figure 1. DEA Efficiency Frontier and Inefficiency Score.

Thus, which determines the maximum feasible value for φ. The final constraints in (3) require $\lambda_A, \lambda_B \le 1$ and $\lambda_A + \lambda_B = 1$, so that φC is restricted to not exceed a linear combination of the adjacent efficient points. The distance function for C is calculated as $d(\mathrm{x}^c, \mathrm{a}^c, \mathrm{y}^c) = \overline{0C} / \overline{0C}^* < 1$, where $\overline{0C}$ and $\overline{0C}^*$ are the distances from the origin to the associated points in Fig. 1. As the number of observations gets large, the piecewise linear production frontier approaches a smooth curve defined by the set of efficient observations. Thus, DEA does not specify a parametric form for the production technology.

I measure productivity change in a joint production model, with a vector of market and nonmarket outputs using production frontier analysis. In the literature of nonparametric frontier technologies approaches, several approaches are proposed, which include Malmquist

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[2] Note that for expository purposes, this simplified illustration considers two outputs, and ignores inputs and attributes. An actual application involves multiple outputs, inputs and attributes.

productivity index and Luenberger productivity index. Although each has different characteristics in their assumption and estimation, all of them share the following characteristics. It measures the TFP change between two data points by calculating the ratio (or difference) of two associated distance functions. A key advantage of the distance function approach is that it provides a convenient way to describe a multi-input, multi-output production technology without the need to specify functional forms. Thus, these approaches allow for a very flexible characterization of productivity changes.

2.1. Revenue Oriented Approach: Productivity Index

Most technology involves more than one output and input. In the presence of multiple outputs, it is not possible to express the production technology in the form of a simple production function since the production function is a "real-valued function" showing the maximum level of output that can be produced with the given level of inputs. As a result, I need to use sets to represent the technology.

Malmquist productivity index, which is the most frequently used nonparametric frontier technologies approach to measure productivity, requires the choice between of an output or input orientation corresponding to whether one assumes revenue maximization or cost minimization as the appropriate behavioral goal.

This is the Malmquist index in output-based measure of TFP allowing for inefficiency in each decision unit (e.g., Färe *et al*., 1994). Distance function is convenient way to describe a multi-input, multi-output production technology without the need to specify a behavioral objective.

2.2. Profit Oriented Approach: Luenberger Productivity Index

Luenberger productivity index is the dual to the profit function and does not require the choice of an input–output orientation. Since the Luenberger productivity index can be applied with an output or input-oriented perspective, it is a generalization of, and superior to, the Malmquist productivity index. Luenberger (1995) generalizes the previous notion of distance functions as a shortage function and provides a flexible tool to take account of both input contractions and output improvements when measuring efficiency. This shortage function, also called as directional distance function, is the dual to the profit function (Luenberger, 1995).

3. Ecological Productivity Index

The DEA estimates the relative efficiency of production units, identifies best practice frontiers, and provides various measures of changes in productivity over time. TFP_{Env} is estimated from two Total Factor Productivity (TFP) estimates: (i) productivity of market output (i.e., agricultural production), denoted by TFP_{Market} and illustrated in Fig. 2 (a); and (ii) the sum of the productivity of nonmarket output (i.e., the reduction in environmental risks) plus market output, denoted by TFP_{Joint}, which is illustrated in Fig. 2 (b), following Managi et al. (2005). Fig. 2 illustrates the simple conceptual model of this technique. Fig. 2 (a) shows the two good (i.e., market) outputs case, whereas Fig. 2 (b) shows the one good output and one bad (i.e., nonmarket) output case.

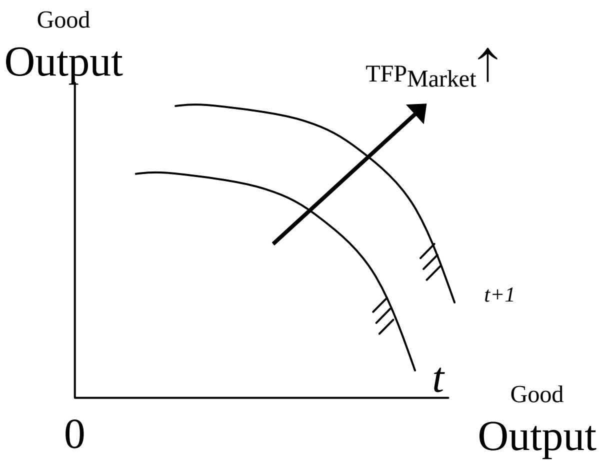

Figure 2. (a).: Market Technology.

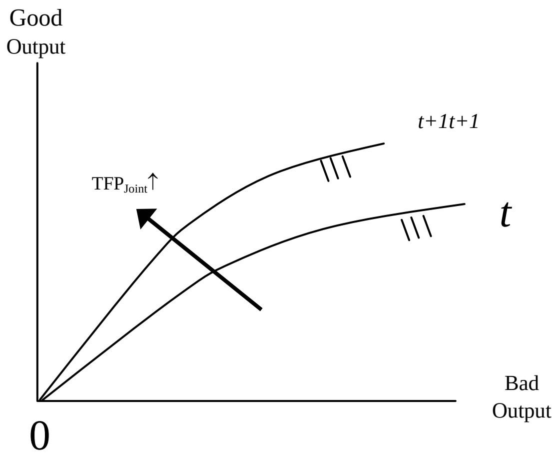

Fig. 2 (b). Non-Market Technology with Market Technology

Figure 2. Conceptual Illustration for Technology Index

Note that DEA can handle multi-output/multi-input analysis. TFP_{Market} includes the usual production inputs and outputs, and TFP_{Joint} includes environmental degradation and abatements effort, as well as production inputs/outputs. Given the input level, an increase in output raises the usual productivity, TFP_{Market}. Holding inputs and environmental output constant, an increase in good output raises TFP_{Joint}. Furthermore, holding inputs and good output constant, a decrease in the environmental output raises TFP_{Joint}. Thus, the residual effects of two factors explain the productivity resulting from changes in technology for the nonmarket goods (environmental degradation).

The total factor productivity (TFP) associated with environmental outputs, TFP_{Env} or ecological productivity, is then calculated as:

$$TFP_{Env}(M) = TFP_{Joint}(M) / TFP_{Market}(M), \quad (32)$$

$$TFP_{Env}(L) = TFP_{Joint}(L) - TFP_{Market}(L), \quad (33)$$

where TFP(M) is Malmquist index, is the ratio of the two models. TFP(L) is Luenberger index, is the difference of the two models. These are because Luenberger productivity indices employ the difference method while Malmquist productivity indices use the ratio method (see Färe *et al.,* 1994; Chambers *et al.,* 1998).

In a country level analysis, TFP includes not only the change in technology, but also the effect of management-level changes in institutions, including environmental regulations. Production frontier analysis yields the ratio or difference indexes, which can then be used to quantify productivity change. The index-based approach measures the TFP change between two data points by calculating the ratio or difference of two associated distance functions or shortage functions. This approach has several advantages. One advantage is the immediate compatibility with multiple inputs and outputs. This is important for environmental applications since pollutants, as the by-product of market outputs, can be multiple. This technique estimates the weight given to each observation, such as the weight or shadow price for each item of environmental pollution data, and implicitly combines these into the one index. In addition, this approach can incorporate the inefficient behavior of the decision maker and avoid the need for the explicit specification of the production function (see Managi *et al.* (2004, 2005) for further details).

4. ENVIRONMENTAL POLICY AND PRODUCTIVITY

Technological progress and productivity increase play a key role in the solution of ecological problems in the face of these increasingly stringent environmental regulations. Therefore, environmental policy needs to be properly designed to promote technological innovation, and to favor its diffusion (Jaffe *et al*., 2003). There is a discussion that whether environmental regulations have played a major causal role in impairing the competitiveness of industries. Conventional wisdom tells that environmental regulations impose significant costs to industry and slow productivity growth and impede technological progress (Jaffe *et al.* 1995). Within this context, the key issue is how to design environmental regulations to attain environmental goals while controlling the adverse impact on industry to the extent feasible, *i.e.*, minimizing productivity loss. Recently, however, revisionist proposes the alternative hypothesis that environmental regulations can have a positive effect on competitiveness in the industry. Environmental regulations pressure firms to innovate and thus enhance growth and competitiveness, named Porter hypothesis (Porter, 1991; Porter and van der Linde, 1995). While there are market failure in technological innovation (e.g., Romer, 1990) that are (1) technological knowledge is a public good - a non-rival and non-excludable good; (2) many beneficiaries (free riders) do not contribute to the cost of technological knowledge; and (3) uncertainty unable to make estimations of technical and commercial returns to innovations, there are situations environmental regulations can lead to long-term benefits to industry (Mohr, 2002).

The notion of the Porter hypothesis is somewhat ambiguous. This study provides the two different interpretations of the hypothesis (see Jaffe *et al.* (1995) for more interpretations). One form of the hypothesis is that, in the long run, environmental regulations are not to worry, because it really will not be all that expensive. This is so since the regulations encourage innovations and these innovations might, in general, more than fully offset the compliance cost and lead to a net benefit for the regulated firm (Porter and Linde, 1995). Many economist disagree with this hypothesis since addition of constrains on a firm’s set of choices cannot be expected to result in an increased level of profits. The other interpretation of the Porter hypothesis is that tougher environmental regulations make firms more internationally competitive than weaker environmental regulations. This interpretation was first advanced as a response to the claim that U.S. firms had become less competitive due to strict environmental regulations. Porter (1991) argues that the critics were wrong and the right form of more stringent regulation could spur competitiveness. I define competitiveness in terms of productivity following literature. This is so because technological change lies at the heart of *long-term* economic growth and social benefit (e.g., Romer 1990).

Depending on how targeted index is measured, the Porter hypothesis may be formulated in different ways (e.g., Jaffe et al., 1995). In a recent study by Managi and Kaneko (2006), they follow the framework presented in Managi *et al* (2004) that measures productivity and tests the causality between environmental regulations and productivities. They examine three versions of the Porter hypothesis. In the standard, or “strong”, version, productivity is measured in terms of market outputs only (e.g., crop and livestock productions). In the common, or “weak”, version, productivity is measured in terms of environmental outputs only, which is green productivity (i.e., efficient utilization of pollution abatement technologies). The re-cast version of the hypothesis considers joint production model including both market and environmental outputs which environmental outputs are non-market outputs.

Their paper contributes to the literature on productivity change in several ways. First, they apply a distance function approach to a province-level data set tracked from 1987 to 2001 in China to measure various components of total factor productivity (TFP) within a joint-production model of market and environmental outputs (see Fig. 3). This contributes to our understanding of the various components of total factor productivity change in China. In addition, their study contributes to better economic and environmental policy design for sustainable development in China by empirically estimating the role of economic and environmental management on market and non-market (i.e., ecological) productivity.

Their result for market output is consistent with the literature that there has been considerable TFP growth in China, while environmental managements in China have not effectively regulated wastewater, air and solid waste pollutants emissions over our study periods. Over the last two and a half decades, China's economy has recorded an average annual growth rate of close to 9%. As a result of this China's extremely rapid economic growth, the scale and seriousness of environmental problems are no longer in doubt. Whether pollution abatement technologies are utilized more efficiently is crucial in the analysis of environmental management because it influences the cost of alternative production and pollution abatement technologies, at least in part.

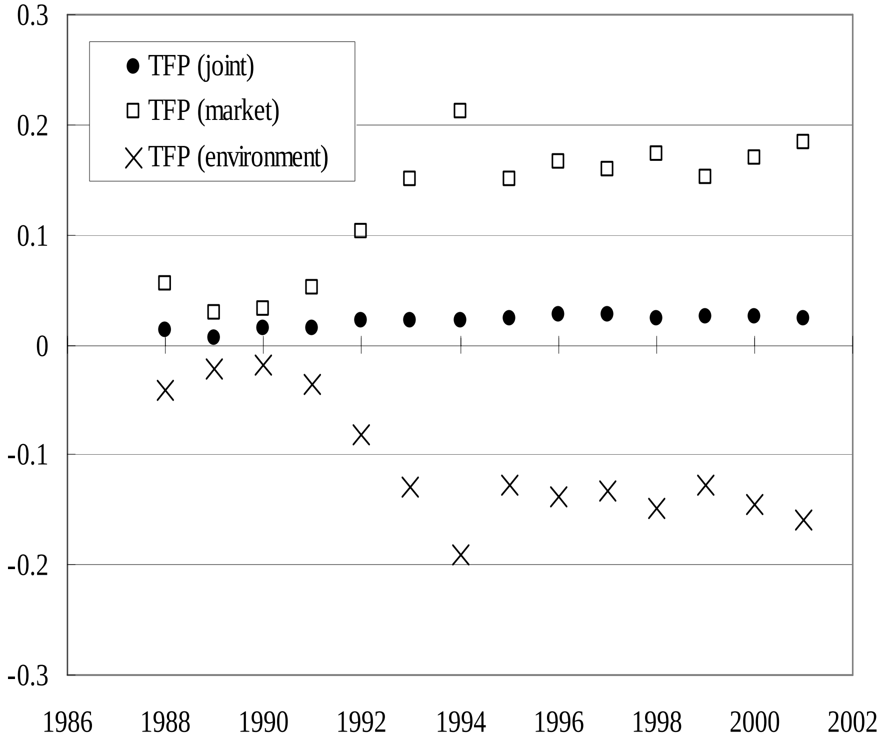

Figure 3. Productivity for Joint, Market and Environmental Sectors.

Larson et al. (2003) summarizes an assessment of future energy-technology strategies for China. Target in their analysis is find the solution to continue its social and economic development in China while ensuring national energy-supply security and promoting environmental sustainability over the next 50 years. Identification of the technological configuration for an energy system is essential and MARKAL, which is a linear programming model, is used to build a model of China's economic system representing all sectors of the economy and including both energy conversion and end-use technologies.

Their analysis indicates a business-as-usual strategy that relies on coal combustion technologies would not be able to meet all ecological and energy security goals. However, an advanced technology strategy emphasizing (1) coal gasification technologies co-producing electricity and clean liquid and gaseous energy carriers (polygeneration), with below-ground storage of some captured CO_2; (2) expanded use of renewable energy sources (especially wind and modern biomass); and (3) end-use efficiency would enable China to continue social and economic development through at least the next 50 years while ensuring security of energy supply and improved local and global environmental quality.

In the future, more stringent comprehensive pollution control and energy strategies could be obtained by implementing new technologies and more effective management. In addition, it will be crucial for China to rely on private initiatives in order to take major steps in turning current environmental dilemmas around (see Economy, 2004). This is especially so since privately owned firms have less bargaining power in complying policies than state-owned enterprises.

However, China has many challenges in implementing these strategies. First, China is a developing country, and economic development is a primary consideration. Balancing development with ecological protection to realize sustainable development is difficult. Second, the dominance of coal to supply energy cannot be changed in the near future. Also, in general production efficiency is very low, so large investments and long times are needed for improvement. Thus, shift to improved environmental management need to be cost effective.

It is important to understand the ecological performance to realistically estimate the future possibility of pollution reduction. This study analyzes how the performance of environmental management changed over time. Their results find mixed results of ecological productivity using nonparametric productivity indices technique. The productivity increases in overall pollution's case, wastewater, and water uses. However, it decreases in solid waste case. The case for waste gas remains constant relatively in 2003 compared to 1992 although there were some fluctuations.

Managi and Kaneko (2005) provide three interpretations regarding the changes in ecological productivity. Managi and Kaneko (2005) support the interpretation that if the productivity decreases, there is a less efficient utilization of pollution abatement technologies and incompleteness in monitoring and enforcement although China has implemented many environmental policies in the past, and the stringency of these regulations is increasing. As a short-term outlook prevails, investment in waste treatment or new conservation efforts diminish (Economy, 2004). The report by Economy suggests the development of rural areas has contributed to alarming levels of pollution. There is also an evidence for problems in environmental protection management at the local level (Ma and Ortolano, 2000). Ma and Ortolano (2000) show that the administrative rank of the environmental protection bureaus is sometimes lower than that of the enterprises it is intended to oversee (also see Economy, 2004). For example, there are several Environmental Protection Offices where officer was not permitted to monitor wastewater from the paper manufacturing company. This is because the administrative rank of the environmental protection office was lower than that of the company director. Therefore, barriers to effective monitoring and enforcement efforts remain relatively constant, even though the stringency of regulation has increased.

Their results of measuring the market and ecological productivity and connecting it to policies have general implications. These are; there are important factors connecting law, institutions, finance, and growth that are not well understood. A better understanding of how these non-standard mechanisms work to promote growth can shed light on optimal development paths for many other countries. In the next stage, China would need to achieve at least three objectives: first, to set the goal of transition to a market system, second, to establish market-supporting institutions incorporating international best practices, and third, to privatize and restructure state-owned enterprises. For example, the linkages between China and the other countries are assessed by Cheung et al. (2003). They recognize that there are non-negligible restrictions on both physical and financial flows between the China and other economies. Smooth transitions will encourage the transfer of better use of technology and better management.

Detecting the determinants of these factors, they found the "international spillover, FDI" instead of "domestic invention, Patent" is the major factor to increase the market productivity growth. They also found significant negative impacts of pollution abatement and control expenditure (PACE) on market technological progress, although elasticity is small. This PACE, in contrast, positively affects to ecological productivity and technological progress as expected.

While FDI helps economic development by encouraging market productivity improvements, it does not lead to a positive consequence for environmental technologies where FDI does have negative coefficients though they are not significant. Thus, they are able to say that FDI may lead to more environmental damage since firms in advanced countries might avoid stiff environmental regulations.

They find the negative consequence of levy to ecological productivities. Therefore, it seems reasonable to conclude that the levy system needs to be re-considered and they point out several problems of the current system in the following areas: 1). Enforcement of environmental laws is limited and policies and firms' environmental managements are insufficient. For example, the levy rate is less than the average cost of pollution abatement partially because the levy fees are not indexed for inflation, and, for state-owned enterprises, they can be included under costs and later compensated through price increase or tax deductions (Sinkule and Ortolano, 1995). 2). Smaller enterprises tend not to pay levy though they share significant rate of total industrial outputs. 3). The cost of installing pollution abatement facilities is usually not subject to financial assistance from the commercial banks. It should be noted that further studies in developing countries are important since results relies on each country's policy and industry structures among others.

REFERENCES

Chambers, R.G., Chung, Y. and R. Färe (1998) Profit, Directional Distance Functions, and Nerlovian Efficiency. *Journal of Optimization Theory and Applications* 98: 351–364.

Cheung, Yin-Wong, Chinn, Menzie D., Fujii, Eiji. 2003. The Chinese Economies in Global Context: The Integration Process and Its Determinants. NBER Working Paper No. 10047.

Economy, E.C. (2004) The River Runs Black: The Environmental Challenge to China's Future, Ithaca: Cornell University Press.

Färe, R., S. Grosskopf, M. Norris, and Z. Zhang. 1994. Productivity Growth, Technical Progress, and Efficiency Change in Industrialized Countries. American Economic Review 84(1): 66–83.

Jaffe, A.B., S. Peterson, P. Portney, and R. Stavins. 1995. Environmental Regulation and the Competitiveness of U.S. Manufacturing: What Does the Evidence Tell Us? Journal of Economic Literature 33: 132-163.

Jaffe, A.B., Newell, R.G. and Stavins, R.N. (2003) Technological Change and the Environment. In: Karl-Göran Mäler and Jeffrey Vincent, (eds) *Handbook of Environmental Economics*. North-Holland. Elsevier Science, Amsterdam, pp 462-516.

Larson, E.D., Z. Wu, P. DeLaquil, W. Chen, and P. Gao, "Future Implications of China's Energy-Technology Choices," Energy Policy, 31(12): 1149-1204, 2003

Luenberger, D. G. (1995) Microeconomic Theory, McGraw-Hill, New York.

Ma, X. and Ortolano L. (2000) Environmental Regulation in China, Lanham, Rowman and Littlefield Publisher

Managi, S. and S. Kaneko. 2005. " Productivity Change, FDI, and Environmental Policies in China, 1987-2001" *Working Paper, Yokohama National University*.

Managi, S. and S. Kaneko. 2006. " Productivity of Market and Environmental Abatement in China." *Environmental Economics and Policy Studies* 7(4) 459-470.

Managi, S. Opaluch, J.J. Jin, D. and Grigalunas, T.A. (2004) Technological Change and Depletion in Offshore Oil and Gas, *Journal of Environmental Economics and Management* 47 (2): 388–409.

Mohr, R. 2002. Technical Change, External Economies, and the Porter Hypothesis. Journal of Environmental Economics and Management 43 (1): 158-168.

Porter, M. E. 1991. America's Greening Strategy. *Scientific American* 264, 168.

Porter, M.E. and C. van der Linde. 1995. Toward a New Conception of the Environment-Competitiveness Relationship. *Journal of Economic Perspectives* 9 (4): 97-118.

Romer, P.M. (1990) 'Endogenous technological change', *Journal of Political Economy*, Vol. 98, No. 5, pp. S71-S102.

Sinkule, B.J. and Ortolano, L. (1995) Implementing Environmental Policy in China, Westport, CT: Praeger.

In: Ecological Economics Research Trends
Editor: Carolyn C. Pertsova, pp. 337-348
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Chapter 17

THE APPRAISAL OF PROJECTS WITH ENVIRONMENTAL IMPACTS. EFFICIENCY AND SUSTAINABILITY[1]

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ABSTRACT

It is usually assumed that the appraisal of the impacts experienced by present generations does not entail any difficulty. However, this is not true. Moreover, there is not a widely accepted methodology for taking these impacts into account. Some of the controversial issues are: the appropriate value for the discount rate, the choice of the units for expressing the impacts, physical or monetary units —income, consumption or investment— and the valuation of tangible and intangible goods. When approaching the problem of very long term impacts, there is also the problem of valuing the impacts experienced by future generations, through e.g., the use of an intergenerational discount rate. However, if this were the case, the present generation perspective would prevail, as if all the property rights on the resources were owned by them. Therefore, the sustainability requirement should also be incorporated into the analysis. We will analyze these problems in this article and show some possible solutions.

1. INTRODUCTION

Reviewing the literature on project evaluation it might be thought that, if all the impacts of a project affected present generations, then there would not be any difficulty in the evaluation of any project from a social perspective. However, the reality is quite different.

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Arrow's impossibility theorem (Arrow, 1951) and the theorems of Gibbard (1973) and Satterthwaite (1975) show that, given a group of alternatives, it is not possible to arrange them or find the best choice from a social perspective if a minimum of logical properties is required to the result. This is a general result and affects any way of making social choices, including market and voting systems. In short, it is obvious that there is not any available procedure for public project evaluation—and it is not possible to design it—free of paradoxical results.

Therefore, there is not a completely satisfactory system for public project evaluation. We are neither able to find a procedure superior to all the others. As a consequence, there have appeared numerous methods, which respond to different approaches, have different logical properties and tackle with greater or lower success different difficulties.

There is an obvious conclusion: it will be preferable one or other system depending on the specific case to solve and on the objective pursued. For example, Osborne and Turner (2007) conclude: "We find that a referendum leads to higher welfare than a cost benefit analyses in "common value" environments. Cost benefit analysis is better in "private value" environments".

Leaving aside the purely qualitative methods, there are several quantitative evaluation procedures incorporating in an explicit or implicit way a relative prices system. Not all of them are acceptable. Remer and Nieto (1995) present 25 quantitative procedures for measuring the desirability of a project, although most of them are not advisable at all. Twenty of them can be rejected because of their lack of rationality. Some give the same weight to impacts occurring in different moments of time, obviating the need for time discounting. Other methods compute costs and benefits following accounting criteria that, such as amortization or the imputation of general expenses, contradict the basic economic notions of cost and benefit. For any selected method it is necessary: a- identifying the relevant costs and benefits, b.- quantifying them; c- valuing them and d- weighting the impacts according to the moment they happen. Two problems will be examined below: time discounting and sustainability.

2. THE PROBLEM OF DISCOUNTING

Let us assume that, such as in Cost-Benefit Analysis, it is possible to quantify and valuing all the impacts in all the years of a project. The aggregation of the flows in each year t, t = 0, 1, …, T, is done by means of the well-known Net Present Value function:

$$NPV = \sum_{t=0}^{T} a_t (1+r)^{-t} \tag{1}$$

where r is the social discount rate. The NPV is a profitability measure in absolute terms. It measures the change of wealth in year 0 that is equivalent to undertake the project. Given a_t, the function depends on the value given to the social discount rate r.

Before we tackle the controversial problem of choosing the appropriate methodology for determining the discount rate, it is necessary to decide whether we follow methodological individualism or, following a paternalistic approach, decide the discount rate value independently of individual preferences.

In any case, the discount rate r, can represent a- the social time discount rate STDR, b- the social opportunity cost of capital, or c- the minimum profitability that the decision maker expects to achieve in order to undertake the project. In ideal conditions both will lead to the same value for r. However, in practice, the result will substantially differ according to the methodology which is followed.

The time preference rate of an individual might include various factors: consumption impatience, survival probability, and the decreasing marginal utility of consumption if the individual expects having an increased consumption over time. However, it might be argued that discounting for impatience is not rational, and that individuals die, but not society, so these components should not be taken into account. In this case the STDR would be very low, around 0.5%, while if all the relevant factors at individual level are taken into account, this rate might be around 5%.

The opportunity cost of capital approach is not an easier way because there are very different alternatives. 1- The interest rate of capital markets. 2- The relative profit of the economy computed as the ratio between total benefit and capital stock. But this is an average value and we are looking for the marginal one. Moreover, there are definition and measurement problems with profit and capital stocks, especially in the public sector. Values around 20% are normal. 3- The marginal productivity of capital computed by means of the production function of the economy. This is a more rigorous method than the previous one, and gives similar values.

Other interesting approach is the shadow price of investment (SPI). The SPI computes the present value of the flows generated by a unit of investment with a rate of profitability *q* during a time T, which are discounted through the STDR. In its simple formulation it results:

$$PCI = \frac{q}{STDR} \tag{2}$$

It is assumed that part of project flows is allocated to consumption and the rest is allocated to investment. The discounting of the flows is undertaken by multiplying the funds allocated to investment by the SPI.

The problem is choosing the most appropriate model for determining the shadow price of investment. As Souto (2002) shows, the result of the SPI computation is very sensible to the hypothesis of each model with respect to the duration of the investment and to the consideration of reinvestment possibility. In such a way that the SPI might be infinite, 2.5, 1.2 or -7.1 for the same project according to the model employed.

Paradoxically, the computation of the SPI does not use to include the so-called marginal cost of public funds (MCF). The MCF measures the marginal costs of the inefficiency caused in the economy when collecting funds by means of taxes or other distortionary instruments—see figure 2. There are also competing methodologies for the measurement of these costs—see Triest (1990) and Liu (2003)—and none of them is superior to the others, which makes more difficult their application. Obviating this component involves undervaluing the necessary costs for undertaking an investment. Actually, the MCF is much higher than the average cost, is roughly 0.2 per collected unit, and might achieve the high value of 0.5, which is the result of Sancho (2003)—see an example in Annex I. Moreover, administration and compliance costs should also be added to this welfare cost. It is not necessary to highlight that the debate on the MCF is not less important than the one on the STDR, with regard to its possible impact on the result of the measurement of the desirability of a public project.

After reviewing the issue, there is a clear conclusion: there is not consensus, not only on the best way of computing the appropriate discount rate, but also with respect to the factors that this rate should include. Consequently, it is not unusual the choice of a third way: directly choosing a reasonable value. Then, it would be convenient to achieve an agreement on which is the discount rate value to be applied in public project evaluation. However, there is not a unique notion of reasonable rate, so that it might happen that one public agency employed a discount rate of 3% while other public agency employed a discount rate of 12% for evaluating the same project. This happens in the real world, especially in international cooperation projects. This lack of consensus might lead to paradoxical results, as we show below.

Let A and B be two different agencies that apply r_A = 3% and r_B = 12% discount rates respectively. They have to evaluate projects I and II—see Figure 1. Under these conditions, it might happen that $NPV^{I}(r_A) > NPV^{II}(r_A)$ but $NPV^{I}(r_B) < NPV^{II}(r_B)$ so that agency A would choose project I and agency B would choose project II—see Figure 1. Of course, r_A and r_B values correspond to rates applied in real world valuation of public projects.

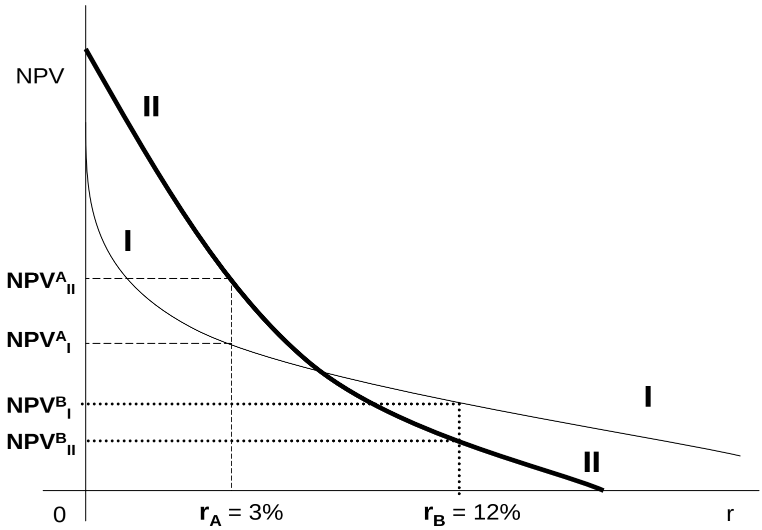


Figure 1.

3. The Problem of Sustainability

With all its limitations, the conventional discount rate r shows the preferences of an individual on the availability of a good or resource today versus tomorrow. Under the immortality hypothesis—which is implicitly done in the conventional computation of the NPV—it is relatively easy to aggregate individual discount rates in order to get a social discount rate. But it is not possible to get an efficient result by means of this approach, because it does not take into account the preferences between own and descendants' consumption, or in other words, between present generations (PG) and future generations (FG). The result is a systematic bias in the NPV computation, which undervalues costs and benefits that occur in what is called long term—in human terms. The most important bias arises in the evaluation of the impacts that do not affect PG but have much importance for FG.

The problem when computing long term impacts is clear. With a discount rate of r = 5% the Net Present Value (NPV) of $ 1,000,000 in 100 years is lower than $ 8,000. However, for the generations that will live in 100 years, the impact will be $ 1,000,000, *ceteris paribus* and, in any case, 100 years is an insignificant lapse of time for the planet. In the limit, PG would measure the value of a catastrophe for FG as a little cost that can be compensated with a small benefit. With the purpose of avoiding this kind of results, several proposals have been made.

The most straightforward proposal is extremely simple. If the problem worsens as the discount rate increases, then it might be solved employing a zero discount rate or might be alleviated with a sufficiently low rate—see Daly and Cobb (1989) among many others. A zero discount rate involves a practical problem: all the projects yielding a positive flow of net benefits for an unlimited time would lead to an infinite value for the profitability computation—unless the time scope is limited to a finite and arbitrarily low one. But this proposal also has serious conceptual problems. It is an *ad hoc* solution, because the discount rate is a data, a parameter, and not a decision variable. The rate which represents the preferences between present and future consumption of a generation is arbitrarily modified. Moreover, although the proposal is justified because it favors future generations, it does no take into account citizens' preferences between own and descendants' consumption.

There is a more appealing proposal, the proposal of a decreasing discounting over time. Heal (1997) suggests that a discount rate decreasing in a logarithmic way over time would be more appropriate than a constant rate. Weitzman (2001) proposes a similar approach: hyperbolic discounting. Weitzman starts from a survey in which the next question was asked "(…) what real interest rate do you think should be used to discount over time (expected) benefits and (expected) costs of projects being proposed to mitigate the possible effects of global climate change?". The data obtained fitted a gamma function, and in this way "even if everyone relieves in a constant discount rate, the *effective* discount rate declines strongly over time" - Table 1.

Table 1. "Approximate Recommended" Sliding-Scale Discount Rates

Period of years	1–5	6–25	26–75	76–300	> 300
Discount rate	4%	3%	2%	1%	0%

Source: Weitzman (2001).

The formal analysis is appropriate. However, some economic objections might be done:

a) The question mixes the costs—which are tangible and are pure private goods—with the benefits—which are intangible and are a pure public good in the Samuelson sense. Actually, environmental costs and benefits should be discounted through a different rate than the appropriate one for market goods (see Almansa and Calatrava, 2007).

b) It might be thought that the same data would adjust other functions and, under this hypothesis, the result would have been qualitative different.

c) It might be questioned if it had not been preferable asking directly about the appropriate rate for each period of time, without imposing the restriction of a constant rate.

However, the result is reasonable and appealing. Actually, the proposal, with small variations in values and intervals, is included in Treasury (2003)—Table 2.

Table 2. The Declining Long Term Discount Data

Period of years	0–30	31–75	76–125	126–200	201–300	> 300
Discount rate	3.5%	3.0%	2.5%	2.0%	1.5%	1.0%

Source: Treasury, http://greenbook.treasury.gov.uk/

The result alleviates the loss of importance of a very long term cost or benefit which would result from the application of a conventional rate. A hyperbolic rate of discount has also the advantage of simplicity. Therefore, it has clear virtues when comparing it with other more soundly based methods.

However, there is a serious criticism; it would be too daring to state that *one* hyperbolic rate is able to show *two* different preferences: time preferences with regard to citizens' own consumption and preferences on intergenerational allocation. The result is that the application of one rate cannot be efficient as far as it ignores, completely or partially, citizens' preferences and intergenerational externalities.

The problem, correctly exposed, consists in accounting and weighting the costs and benefits of a project in a model with overlapped generations, which is a relatively new and complex approach. First, it is necessary to take into account the conventional rate r, which shows the preferences of an individual between present and future consumption. Second, it is necessary to explicitly incorporate other rate R, which represents the preferences between own and descendants' consumption.

Kula (1988) determines the profitability of a project for each generation through the computation of the NPV with the conventional discount rate. These NPVs are aggregated giving the same weight to all generations. That is to say, with a zero intergenerational rate. Several authors, such as Collard (1981), Bellinger (1991), Pasqual (1999), Sumaila and Walters (2005) and Almansa and Calatrava (2007), among others, propose the use of two rates—r for computing each NPV and R for aggregating them—or what might be equivalent, the conventional rate r and an intergenerational weighting.

Therefore, there is a good theoretical basis for computing the costs and benefits of a project which affects several generations. But there is an applied problem, as there is still not a reasonable estimation of intergenerational weights (or discount rates), or of their order of magnitude.

The solution to the issue exposed above is only part of the problem. It might improve efficiency, but only consists in taking into account present generations' preferences. Thus, it might imply acting as if future generations had not any right. The use of two discount rates—intragenerational and intergenerational—might be appropriate, but does not guarantee the fulfillment of the sustainability requirement.

Padilla (2002) and Pasqual and Souto (2003), among others, highlight that, in order to fulfill the sustainability requirement, it is necessary to employ other instruments. There is not enough with the use of economic tools, it is also necessary to employ political mechanisms and to develop institutional innovations and reforms. The basic problem is simple, present generations (PG) have much and varied tools for passing on their preferences to future generations (FG), but the same does not happen in the inverse way. FG cannot communicate with PG and do not have any possibility of negotiating with PG. Then, FG will find the resources voluntarily legated by PG.

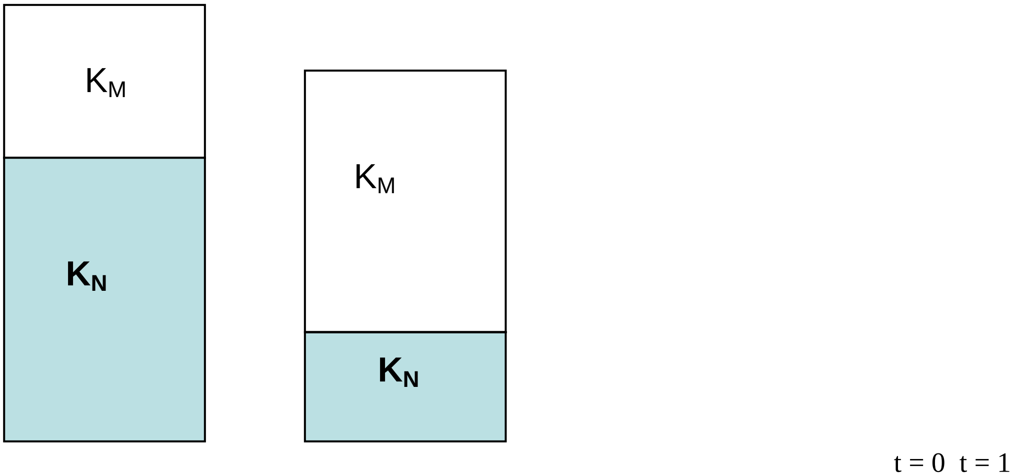
Each one of the successive PG acts according to the least efficient economic regime—the *open access* one—with respect to the following generations. Thus, it might be foreseeable a collapse of the system in a relatively brief period. However, this prediction would not be much realistic as it ignores important factors that act in opposite sense. As far as PG have well defined preferences on the welfare of FG—and there are the appropriate institutions—it would be possible that the behavior of PG led to a sustainable result.

Sustainable development might be defined as the "development that meets the needs of the present without compromising the ability of future generations to meet their own needs", WCED (1987). Let be R_{jg} the initial resources of type j that are available for generation g, which has population N_g. These resources might be exploited by generation g with a rate of return k_g, obtaining $(1+k_g)\cdot R_{jg}$. Let C_g and c_g, $c_g = C_g/ N_g$, be the total and average needs of resource j for generation g, with $C_g = h_g(1+k)\cdot R_{jg}$, $h\in\Re_+$. In the same way, the following generation g+1 will obtain resources $R_{jg+1} = [(1-h_g)(1+k_g)(1+k_{g+1})\cdot R_{jg}]$ and *per capita* resources $r_{jg+1} = [(1-h_g)(1+k_g)(1+k_{g+1})\cdot R_{jg}]/N_g$. Sustainability with respect to the use of resource j by generations g and g+1 would be obtained if:

$$h_g \leq 1 \text{ and } C_{g+1}/ N_{g+1} \leq [(1-h_g)(1+k_g)(1+k_{g+1})\cdot R_{jg}]/N_g \tag{3}$$

If $h_g > 1$, then the generation g needs exceed the resources available for them and it would not be sustainable for present generation g. The result, in terms of the size of the populations of g and g+1, depends on whether these needs are the subsistence ones or exceed this biological limit. If $h_g \leq 1$, then sustainability for generation g is achieved. Sustainability for generation g+1 would be more feasible the lower the populations (N_g and N_{g+1}) and the needs (c_g and c_{g+1}) and the greater the resources productivities (k_g and k_{g+1}).

If the conditions in (3) were widely fulfilled, then it would not be necessary to continue the analysis. It might be though that a path leading to this goal is being followed. The arguments for sustaining such opinion would be several and important changes that are happening. One of these changes is green national accounting. This consists in adjusting conventional accounting systems with the aim of properly including all the—tangible and intangible—costs and benefits in national accounting (Ahmad, Serafy, and Lutz, 1989). In this way the generation of wealth will not be mixed up with the simple transformation of natural capital (K_N) into manufactured capital (K_M)—Figure 2.



In moment t = 0 there are K_N units of natural capital and K_M units of manufactured capital. In the following period t =1, K_M has considerably increased, although the reduction of natural capital K_N is even greater. There has been a loss that green accounting would take into account while conventional accounting would interpret as spectacular growth.

Figure 2.

Among the changes experienced, one might also highlight innovations and improvements in the methodology for the design of projects with great environmental impact. As an example, the concept of *habitat equivalency analysis* (HEA)—see Dunford, Ginn and Desvousges (2004) and Zafonte and Hampton (2007)—is attractive and relevant, both from a theoretical and from an applied perspective. The HEA procedure aims to compensate in terms of present value for the environmental damages in a habitat. The complete repairing of a damaged habitat would not be sufficient compensation: as a consequence of considering time discounting, in order to maintain the value of 1 unit lost today it is necessary to get (1+r) units tomorrow. In order to compute the compensation, the units that have been repaired and their relative value are taken into account, as well as the quantity of equivalent habitat that has been produced.

It is important to highlight that the compensation for an environmental impact by means of the HEA is undertaken with exactly the same kind of natural resources. This would avoid some logical problems, such as the Scitovsky reversal paradox (Scitovsky, 1941), as well as ethical ones, which might appear when monetary *compensations* are used.

Project evaluation methods have also been adapted to the new requirements of environmental quality—see e.g., EBRD (2006). These methods overcome the possibilities of classical procedures, such as Cost-Benefit Analysis. Among the new evaluation methodologies, the so-called social multi-criteria evaluation (Munda, 1996 and 2004) stands out due to its potentiality and flexibility.

It could not be denied that both the changes in the design of projects and the appearance of modern evaluation systems for taking into account environmental variables have more relevance for FG than for PG. The same might be said about much of the policies undertaken by governments that, such as some of the policies for mitigating climate change, are more favorable to FG than to PG.

From a less optimistic—or more exigent—perspective the current evolution of methods and policies appears as clearly insufficient. It might be thought that there is still much to be done and that it is urgent to pursue environmental sustainability as a top-priority goal and to determine specific lines of action. In the words of Goodland, Daly and Serafy (1993):

> "Environmental sustainability can be approached by implementing four priorities: first, by using sound microeconomic means; second, by using sound macroeconomics to differentiate between use and liquidation of natural capital by means by environmental accounting; third, by using environmental assessment to incorporate environmental costs into project appraisal; and fourth - until the first three become fully achieved - by following operational guidelines for sustainability".

It might be not enough with correctly incorporating the preferences and the point of view of PG. It might be necessary to explicitly admit that PG do not have all the property rights on the Earth but that, at least in part, these rights also belong to FG. Under this hypothesis the goal is to advance in the design of new institutions with the aim of representing and defending the interests and rights of FG.

Actually, any PG has the capacity of modifying and abolishing institutions, laws and norms. For this reason, it is necessary to build up a protecting network before the formulation of any reform proposal in favor of the interests of FG. In short, this might start with a constitutional amendment in order to difficult the derogation of legal dispositions to be established in favor of FG. Protected natural spaces would be a typical example.

Of course, in order to protect a maritime zone placed in international waters an international agreement would be necessary. The same applies for an appropriate management of strategic resources and of some residuals. The solution implies the creation of specialized international agencies, such as a World bank of natural and environmental resources and an International bank of radioactive residuals.

Besides legal protection, the market incentives might also be used. Fiscal incentives might be used by entities and foundations whose goal was the purchase of natural spaces for their effective preservation.

Last, it would be useful to create the figure of the FG representative. The goal of this agency would be monitoring the use of the resource wealth belonging to FG. In case of conflict, it would claim in markets and in front of the administration or political system an adequate compensation.

ANNEX I. THE MARGINAL COST OF PUBLIC FUNDS. AN EXAMPLE.

Let $C' = 20$ be the marginal cost of production of a consumption good and $p = 100 - 20X$ the inverse demand function—see Figure 3. A perfect market would lead to an allocation of $X_0 = 4$ with a price $p_o = 20$.

If X was charged with a specific production tax $T = 10$, then the quantity would be $X_1 = 3.5$, and consumer price would be $p_1 = 30$ while producer price will remain at p_0. Tax collection would be $R_1 = (p_1 - p_0)X_1 = 35$, (area ①+② in the figure). The inefficiency caused by the price distortion, measured through the excess burden, is $W_1 = (X_0 - X_1)\cdot(p_1 - p_0)/2 =$

2.5, (area ③) while the excess burden per unit of collected resources would be $\underline{w_1} = W/R_1 =$ 7.14 %.

Let us assume that the tax is increased by 10% for financing a project. The tax is now T' = 11 and the new equilibrium is $X_2 = 3.45$, $p_2 = 31$, while the producer price remains at p_0. Tax collection increases to $R_2 = (p_2 – p_0)X_2 = 37.95$ (area ①+④ in the figure) and the excess burden is now $W_2 = (X_0 – X_2)\cdot(p_2 – p_0)/2 = 3.025$, (area ②+③+⑤) which in the next percentage of total tax collection, $\underline{w_2} = W/R_2 = 7.97$ %.

The result of the 10% increase in the tax T charged on good X leads to an increase in collection by $\Delta R = 2.95$, (area ④-② in the figure) with the corresponding excess burden increase $\Delta W = 0.75$, (area ②+⑤). The marginal cost of public funds is then MCF = $\Delta W/\Delta R = 25.42$ %, (area [②+⑤]/[④-②]).

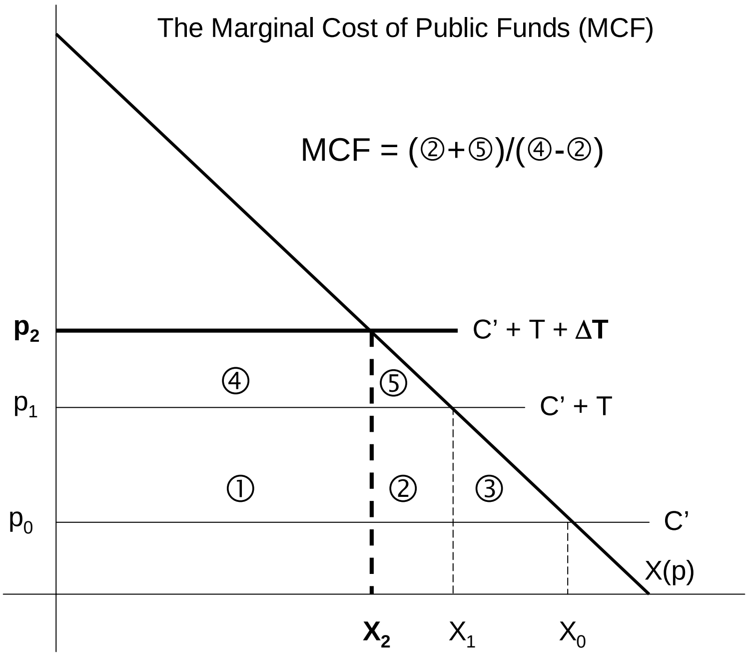


Figure 3.

REFERENCES

Ahmad, Y. J., Serafy, S and Lutz, E. (1989) *Environmental accounting for sustainable development.* World Bank, Washington, D.C.

Almansa, C. and Calatrava, J. (2007) "Reconciling sustainability and discounting in Cost-Benefit Análisis: A nethodological proposal". Ecological Economics, 60, 4:712-725,

Kenneth J. Arrow and Hervé Raynaud (1986) *Social Choice and Multicriterion Decision-Making,* MIT Press., Bellavance, F., Dionea and Lebeau, M (2006) "The Value of a Statistical Life: A Meta-Analysis with a Mixed Effects Regression Model". *Cahier de recherche/Working Paper 06-46,* Centre interuniversitaire sur le risque, les politiques économiques et l'emploi

Bellinger, W. K. (1991) "Multigenerational value: modifying the modified discount method" *Project Appraisal. 6:101-108.*

Coase, Ronald, (1960), "The Problem of Social Cost," Journal of Law and Economics, 3:1-44,

Collard, D. (1981) *Altruism and Economy.* The Pitman Press, Bath.

Daly, H. B. and Cobb, J.B. Jr. (1989) *For the common good: redirecting the economy toward community, the environment and a sustainable future*. Beacon Press, Boston.

Dunford, R. W., Ginn, T. C. and Desvouges, W. H. (2004) "The use of habitat equivalency análisis in natural resource damage assessments" *Ecological Economics,48: 49-70.*

European Bank for Reconstruction and Development (2006), *Sustainability Report 2005 - Working towards a sustainable future*. EBRD, London.

Gibbard, A. (1973), "Manipulation of voting schemes: a general result", *Econometrica,* 41: 587-601

Goodland, R, Daly H, Serafy, S. (1993) "The urgent need for rapid transition to global environmental sustainability" *Environmental Conservation, 20:44,*297-309.

Heal, G. M. (1998). "Discounting and climate change". *Climate Change, 37: 335-343*

Hong-Kyun Kim (2007 "productivity effect of air pollution"Applied Economics Letters, *14: 385–387.*

Kula, E. (1988) "Future generations: The modified discounting method", *Project Appraisal, 3:85-88.*

Liu, L (2003) "A marginal cost of funds approach to multi-period public project evaluation: implications for the social discount rate" *Journal of Public Economics, 87: 1707-1718*

Osborne M. J. and Turner M. A. (2007) "*Cost benefit analysis vs. referenda*" Working Paper 286, University of Toronto, Department of Economics

Munda, G. (1996) "Cost-benefit analysis in integrated environmental assessment: some methodological issues". Ecological Economics 19, 2:157-168.

Munda, G. (2004) "Social multi-criteria evaluation: Methodological foundations and operational consequences" *European Journal of Operational Research,158, 3: 662-677.*

Navrud, S. (2002) "The State of the Art on Economic Valuation of noise". Report to the European Comisión, DG Environment.

Padilla, E. (2002) "Intergenerational equity and sustainability" Ecological Economics, 41:69-83.

Pasqual, J. (1999) *La evaluación de políticas y proyectos.* Icaria editorial y UAB. Barcelona.

Pasqual, J. and Souto G. (2003) "Sustainability in natural resource management" *Ecological Economics,* 46, 1: 47-59.

Remer, D. S. and Nieto, A. P. (1995) "A compendium and comparison of 25 project evaluation thecniques " *Internacional Journal of Production, 42; 79-96 and 101-129.*

Sancho, F. (2003) "Una estimación del coste marginal en bienestar del sistema impositivo en España. Working papers; 592.03 Fonaments de l'Anàlisi Econòmica Institut d'Anàlisi Econòmica.

Satterthwaite, M.A. (1975), "Strategy-proofness and Arrow's conditions existence and correpondence theorems for voting procedures and social welfare functions", *Journal ofEconomic Theory*, 10: 187-217.

Scitovsky, T. (1941) "A note on welfare propositions in economics" *Review of Economic Studies,9, 1: 77-88.*

Souto, G (2002) "El descuento social". Hacienda Pública Española / revista de Economía Pública, 165: 99-126.

Sumaila, U. R. and Walters, C. (2005) "Intergeneracional discounting: a new intuitive approach" *Ecological Economics, 52:135-142.*

Treasury (2003) *The Green Book, Appraisal and Evaluation in Central Government.* H.M. Treasury, London.

Triest, R (1990) "The Relationship Between the Marginal Cost of Public Funds and Marginal Excess Burden" *The American Economic Review*, 80, 3: pp. 557-566.

WCED (1987) *Our common future.* Oxford University Press, Oxford.

Weitzman, M. L. (2001) Gamma discounting. *American Economic Review, 91, 1: 260-271.*

Zafonte, M. and Hampton, S. (2007) "Exploring welfare implications of resource equivalency analysis in natural resource damage assessments", *Ecological Economics,61:134-145.*

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