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Preface: Professor Charles Perrings

Dr. Charles Perrings is a Professor of Environmental Economics at the Global Institute for Sustainability at Arizona State University since August 2005. Professor Perrings is a world renowned ecological economist who has made an extensive contribution to our understanding of the economics of biodiversity. His applied research extends to a variety of areas that include biodiversity, water resources, and resilience of coupled ecological-economic systems. Also, he has conducted research on the problem of sustainable development for decades, much of which has concerned Central and Southern Africa.

His earlier academic positions have been as Professor of Environmental Economics and Environmental Management at the University of York; Professor of Economics at the University of California, Riverside; Director of the Biodiversity Program of the Beijer Institute, Stockholm; Professor of Economics at the University of Botswana; and Associate Professor of Economics at the University of Auckland. He has been editor of the Cambridge University Press journal, Environment and Development Economics and several other journals in environmental, resource and ecological economics, and in conservation ecology. He is a Past President of the International Society for Ecological Economics and is currently the Vice-President of the Scientific Committee of *Diversitas*, an international program of biodiversity science. Professor Perrings is an advisor to various governmental, intergovernmental and international non-governmental organizations as well as research funding agencies.

Prof. Perrings has published extensively in leading journals in the area that include *Journal of Marine Science, Ecological* Modeling, T*rends in Ecology and Evolution*, *Ecological Economics, Journal of Environmental Management*, *Conservation Biology*, Philosophical *Transactions of the Royal Society of London, Scottish Journal of Political Economy, American Journal of Agricultural Economics*, *Environment and Resource Economics*, Bulletin of Marine Science*, American Journal of Physical Anthropology*, *Journal of Environmental Economics and Management*, and *Conservation* Ecology*.*

Professor Perrings' applied research is centred on three main issues: biodiversity, water and the resilience of coupled ecological-economic systems. The work on biodiversity relates to identifying the causes of biodiversity change in different ecological and economic systems. These include climatic changes, sea level rise, social factors, market failures associated with the lack of well-defined rights to biological resources. Systems that are being studied in this way include agro-ecosystems (arable and livestock), lakes and semi-arid rangelands. His work also looks at the consequences of biodiversity change in terms of the stock of genetic information, the impact on the flow of economically valuable goods and services, and the capacity of the system to withstand stresses and shocks. This contributes to a greater understanding of the link between stability, resilience and the sustainability of ecological-economic systems.

Professor Perrings' has also contributed to the development of decision-models for dealing with problems that are characterised by sensitivity to initial conditions, path dependence, abrupt if not discontinuous change at threshold values of selected biological resources, fundamental uncertainty and irreversibility. His current work includes models of the decision-problem in the case of invasive pests and pathogens where conventional risk assessment fails due to the fundamental uncertainty of novel events, and the potential irreversibility of the costs of successful invasion. Professor Perrings has been involved in the development of strategies, policies and instruments to address the problem of biodiversity loss at multiple spatial and temporal scales. His collaborative work has led to a re-evaluation of the significance of the local versus the global public good dimensions of biodiversity conservation.

Professor Perrings' work on water involves the development of wetland models. He studies the correlations between economic uses of wetland resources and wetland functions, and the state of the wetlands. The study has focussed on wetlands in Argentina (Ibera) and East Africa (Lake Victoria). The main feature of the work is that it involves spatially explicit models. He further deals with growth functions of stocks in fisheries models that are not spatially explicit. It captures the effect of economic activities on the parameters or structure of the model. Two specific applications, freshwater fisheries in Lake Malawi and Penaeid shrimp fisheries in the Gulf of Paria, Trinidad, exhibit that the approach can capture quite complex environmental effects with fairly minimal data requirements. The Malawi study has explored the problem of changes in the diversity of the fish catch through a bio-economic diversity index (a Simpson's index weighted by the market price of fish). Not only does it substantially improve the fit of the fisheries model, but it turns out to offer some very interesting implications for the effect of different regulatory and property rights regimes on fish biodiversity.

Professor Perrings and his co-workers were the first to identify the relevance of the ecological notion of resilience to the problem of sustainability. They identify an ecological economic system as a stochastic process that will flip from one state (stability domain) to another. This supposes that the system can converge on any one of a number of possible states, depending on initial conditions. For a given set of initial conditions, a given disturbance regime, and a given state of nature, it may be possible to estimate the probability that the system will converge in some finite time to some other state of nature. The connection with sustainability is direct. If the transition probabilities are known, it is possible to estimate either the time the system occupies a particular state (the sustainability of that state), or the time to converge to any other state (the time to sustainability). One may also estimate the robustness of the system under a particular disturbance regime to change in any particular direction.

The work suggests that we can analyse the evolution of an ecological-economic system under different initial conditions. This makes it possible to compare the long-run implications of different initial conditions in terms of the sustainability or resilience of the system under each. Some progress has been made in developing methods to estimate empirically the loss of resilience in managed ecosystems.

The work of Professor Charles Perrings has far reaching implications for economic-ecological balance, sustainable development and resilience of economies specially those in the developing countries. We dedicate this volume to his great contributions.

IJEES appreciatively acknowledge, for valuable information about Professor Perrings and his contributions, the following resources:

http://cdsagenda5.ictp.trieste.it/askArchive.php?base=agenda&categ=a0258&id=a0258s23t5/recording
http://www.ens-newswire.com/ens/jul2006/2006-07-19-01.asp,
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Ecological Economics after the Millennium Assessment

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Abstract

The Millennium Ecosystem Assessment has changed the way that we think about the interaction between social and ecological systems. By connecting ecological functioning, ecosystem processes, ecosystem services and the production of marketed goods and services it has identified ecological change as an economic problem. It has also drawn attention to a new dimension of the environmental sustainability of economic development. The Hartwick rule for the reinvestment of Hotelling rents on exhaustible and renewable natural resources provides one basis for evaluating the sustainability of extraction policies. The MA's focus on the regulating services provides another. The regulating services offered by ecosystems limit the variability of ecosystem functioning, processes and the production of marketed goods and services. They help to conserve the resilience and hence sustainability of ecosystems. This offers both a challenge and an opportunity to ecological economists. The challenge is to understand the linkages between such services and the capacity of economic systems to function over a range of environmental conditions. The opportunity stems from the fact that the field is uniquely placed to meet this challenge.

Mathematics Subject Classification 2000: 91B76, 91B02
JEL Classification: N50, Q50, Q57

Introduction

A number of studies of the evolution of ecological economics have identified several common features of work in the field. These include the perception that the economy is embedded in and constrained by the environment; that the economy and its environment co-evolve through time, and that the coupled system is complex and adaptive, exhibiting path dependence, non-linearity, and sensitivity to initial conditions; that this generates fundamental uncertainty about the future consequences of current actions; and that for any given set of technologies there is a sustainable scale of the economy. Röpke (2005a, 2005b) has also, however, drawn attention to the fact that the field has developed in many different directions. This is partly as a function of the disciplinary background of the people involved, and partly a function of the institutional, cultural and environmental conditions in which they themselves operate. The International Society for Ecological Economics is an organization with a widely distributed membership, organized in a number of regional societies. The research foci of members of the regional organizations in India, Africa and Latin America tend to be very different from those in the USA, Canada, Europe or Australia/New Zealand.

In thinking about where the field is likely to go in the future, I want to focus on a single area – albeit a very important one. This is not the only direction that the field will go. Indeed, the only thing I am sure about is that ecological economics will continue to push the frontiers of knowledge on wider front than most fields of equivalent size, simply because of its transdisciplinarity and the heterogeneous nature of its practitioners. But it is an area where the stakes are extraordinarily high for all of us. It is the impact of environmental change on regulating ecosystem services, and the consequences this has for human well-being. *Ecological Economics* has published (or has in press) a little over a hundred papers on ecosystem services. A number of notes were generated through the policy forum conducted around Costanza et al (1997), and extended papers appeared in special issues of the journal in 1999 and 2002. However, there has been an explosion of interest in the topic since the results of the MA started to appear. One fifth of all of the papers published in the journal on ecosystem services have either appeared or are due to appear in 2006. This is a major growth area within the field.

Why is it important? Ecosystem services include not just the provision of foods, fuels and fibres and well-understood beneficial phenomena such as pollination, watershed protection, habitat provision and so on, but also the mechanisms that regulate the impact of stresses and shocks (Dirzo and Raven, 2003). Amongst these, for example, is disease regulation. The establishment and spread of introduced pests and pathogens, including emergent zoonotic diseases like the ebola virus, HIV, SARS or avian flu, may turn out to have more impact on human wellbeing over timescales that matter than many other environmental threats currently attracting attention (Daszak and Cunningham, 1999, 2000). The severity of the impact of these diseases, however, depends on environmental conditions (e.g. UNAIDS, 2006). The regulating ecosystem services determine the capacity of ecosystems both to regulate the impact of these shocks, and to respond to changes in environmental conditions without losing functionality (Kinzig et al, 2006). This is a dimension of the environmental sustainability that has been largely ignored by economists. It turns out that the regulating services are important wherever there is a distribution of outcomes, and wherever decision-makers care about the properties of that distribution. Both variance and kurtosis matter to risk-averse decision-makers. Like the institutions of market economies, the regulating services of ecosystems affect the distribution of outcomes through both the capacity to respond to perturbations and the severity of those perturbations. It is hard to imagine a more critical set of issues than those surrounding the decline in regulating ecosystem services.

The Millennium Assessment

The Millennium Assessment has fundamentally changed the landscape in ecosystem service research. By switching attention from the underlying ecological processes and functioning to the services that confer benefits or impose costs on people it has brought the analysis of ecosystem services into the domain of economics – and in so doing has created a natural niche for those working in the field of ecological economics. Although ecological economics is much more than the union of ecology and economics, it was the perception that social and ecological processes are integrally linked that originally spawned the field. Many ecological economists are still concerned to understand: (a) how economic activity, ecological functioning and ecological processes are related, (b) what that means for the value of environmental assets where the latter are either public goods/bads or are not subject to well-defined property rights, and (c) what options are available to deal with the resulting challenges to both efficiency and equity. For such people the research agenda created by the MA is an especially attractive one.

So what did the MA do that opens the door to ecological economics? It defined ecosystem services in terms of the benefits yielded by ecosystems (as composite assets), distinguishing between four broad categories of benefit: provisioning services, regulating services, cultural services and supporting services. Of the four categories, the first is most familiar. Provisioning services cover the renewable resources that had been the focus of much work in environmental and resource economics in the last three decades of the 20th century, including foods, fibres, fuels, water, biochemicals, medicines, pharmaceuticals and genetic material. Many of these products are directly consumed, and are subject to reasonably well-defined property rights. They are priced in the market, and even though there may be important externalities in their production or consumption, those prices bear some relation to the scarcity of resources.

The other three categories are less familiar. Cultural services comprise a novel category of services that captures many of the non-use or passive use values of ecological resources, and reflects the fact that the diversity of ecosystems is reflected in the diversity of human cultures. Cultural services include the spiritual, religious, aesthetic and inspirational well-being that people derive from the 'natural' world. They include the sense of place that people have, as well as the cultural importance of landscapes and species. More importantly, they include (traditional and scientific) information, awareness and understanding of ecosystems and their individual components offered by functioning ecosystems. One modern expression of cultural services – ecotourism – involves well-developed markets. Others do not. While intellectual property rights are increasingly well-defined (largely to protect the patent rights of corporations seeking to develop novel products from biochemical and genetic material drawn from ecosystems), most cultural services are still regulated by custom and usage, or by traditional taboos, rights and obligations. Nevertheless, they are directly used by people, and so are amenable to valuation by methods designed to reveal people's preferences.

The category of support services captures the main ecosystem processes that underpin all other services. Examples offered by the MA include soil formation, photosynthesis, primary production, nutrient, carbon and water cycling. These services play out at quite different spatial and temporal scales. For example, nutrient cycling involves the maintenance of the roughly twenty nutrients essential for life, in different concentrations in different parts of the system. It is often localized, and is therefore at least partially captured by the price of the land on which it takes place. Carbon cycling, on the other hand, operates at a global scale, and is very poorly captured in any set of prices. Since these services are, in a sense, embedded in the other services, they are captured in the valuation of those services.

I wish to focus here on the category of regulating services. For the MA, these include the following:

- Air quality regulation involves chemicals contributed to and extracted from the atmosphere, influencing many aspects of air quality.
- Climate regulation stems from the fact that ecosystems influence climate both locally and globally. So, for example, changes in land cover affect both temperature and precipitation at a local scale, while changes in carbon sequestration or greenhouse gas emissions have significant effects at a global scale.
- Water regulation affects runoff, flooding, and aquifer recharge through changes in land cover, and depends on the mix of plant species and soil microorganisms.
- Erosion regulation depends on vegetative cover, and plays an important role in soil retention and the prevention of landslides.

- Water purification and waste treatment services are both positive and negative, and include both water pollution and filtration in inland waters and coastal ecosystems. It also includes the capacity to assimilate and detoxify soil and subsoil compounds.
- Disease regulation services are also both positive and negative, and include change in the abundance of human pathogens, such as cholera, or disease vectors such as mosquitoes.
- Pest regulation involves the role of ecosystems in determining the prevalence of crop and livestock pests and diseases.
- Pollination services depend on the distribution, abundance, and effectiveness of pollinators.
- Natural hazard regulation covers a wide range of buffering functions, particularly in coastal ecosystems where mangroves and coral reefs can reduce the damage caused by hurricanes and storm surges.

In every case, they affect the impact of stresses and shocks to the system. Some – such as climate or disease regulation are global public goods. Many are local public goods (Perrings and Gadgil, 2003). That is, they offer non-exclusive and non-rival benefits to particular communities. The rest of the world may have minimal interest in such benefits. The fact that they are public goods means that if left to the market, there will be too little conservation effort. But there will be some conservation effort. Indeed, the greater the local benefits to conservation, the greater will be the local conservation effort. More importantly, where there are locally or nationally capturable benefits there will also be an incentive to identify to identify those benefits.

The MA's report on changes in the availability of all of these services is somewhat sketchy, reflecting the paucity of knowledge on these things. But it is still striking how little it was able to say about the value of the services being described, despite twenty-five years of valuation studies by economists. Without wanting to re-open old debates, this is largely because most effort in valuation research has gone into understanding of human preferences for environmental characteristics that are directly consumed. Comparatively little effort has gone into understanding the indirect linkages between ecological functioning, ecosystem services and the production and consumption of marketed goods and services. Almost no effort has gone into understanding the value of the role of the environment in either mitigating or exacerbating the risks we face. This is what the regulating services do. The MA has provided us with a clear challenge. By identifying changes in the regulating ecosystem services as amongst the most important environmental consequence of human activities, and by underscoring our inability to track the effect of this on human well-being, it has set a research agenda that ecological economics is better able to meet than any other field.

Understanding ecosystem services

In thinking about this research agenda I want to consider how current research on the valuation of regulating ecosystem services is meeting the challenges raised by the MA, and what remains to be done. But to get to the punch line first, the major challenge facing ecological economics at present is: (a) to understand the consequences of ecological change induced by current economic activity; (b)to understand the distribution of possible outcomes attaching to alternative activities and, where feasible, the probabilities attaching to those outcomes; and (c) to develop appropriate mitigating or adaptive policies. Valuation is a part of this, but it is only a part.

A number of studies prior to the MA drew attention to the changes in ecosystem services and the importance of quantifying the value of these changes to human societies in terrestrial (e.g. Daily et al, 1997; Daily, 1997), marine (e.g. Duarte, 2000) and agroecosystems (Björklund et al, 1999). Within ecological economics there were also attempts both to refine the identification of ecosystem services, and to come up with estimates of their value (Costanza et al, 1997; Bolund and Huhammar, 1999; Norberg, 1999; Limburg and Folke, 1999; Woodward and Wui, 2001). The MA (2005) itself summarized the state of the art on the identification of ecosystem services, but had difficulty in attaching values to the observed changes in physical magnitudes. This reflected the growing concern over the unreliability of valuation estimates.

Three main concerns have been expressed in the literature. One is the fact that most studies of ecosystem services have focused on a single dimension of the problem only. Turner et al, (2003) drew attention to the fact that few studies had considered multiple functions, and fewer still had estimated ecosystem values 'before and after' environmental changes had taken place. Daily et al (1997) had emphasized that most ecosystem services were the result of a complex interaction between natural cycles operating over a wide range of space and time scales. Waste disposal, for example, depends both on highly localized life cycles of bacteria as well as the global cycles of carbon and nitrogen. The same cycles are implicated in the provision of a range of other services. By ignoring multiple services, many valuation studies underestimate the importance of the underlying ecosystem stocks to the economy.

A second concern is that many valuation studies depend on elicitation of the preferences of people who have little conception of the role of ecosystem stocks in the generation of ecosystem services, or of the link between those services and the production of commodities (Winkler, 2006a). The problem here is that ecosystems and the services they provide are, for the most part, intermediate inputs into goods and services that are produced or consumed by economic agents. As with other intermediate inputs, their value derives from the value of those goods and services. To illustrate, consider the following simplified description of the decision-maker's problem.

$$Max_{\mathbf{h}(t)} \int_0^\infty u(\mathbf{q}(\mathbf{x}(\mathbf{s}(t))), \mathbf{h}(t)) e^{-\delta t} \, dt$$

where utility depends on a vector produced goods, $\mathbf{q}$, a vector of marketed inputs, $\mathbf{x}$,
the state of the environment, $\mathbf{s}$, the harvest of ecosystem resources, $\mathbf{h}$, and the discount rate, δ. This is subject to the dynamics of the natural environment, summarized by the equations of motion:

$$\frac{ds_i}{dt} = f_i(\mathbf{s}(t)) - h_i(t), \; i = 1,\ldots,n$$

The value of the n ecosystem stocks in this problem is their social opportunity cost, measured by the shadow price (or costate variable) obtained from the solution to the optimization problem. Specifically, if the costate variables in the solution to the problem are denoted λ_i, then they will evolve as follows:

$$\frac{d\lambda_i}{dt} = \lambda_i(\delta - f_i^{'}) - \sum_j \frac{du}{dq_i}\frac{dq_i}{d\mathbf{x}}\frac{d\mathbf{x}}{ds_i}, \; i = 1,\ldots,n$$

and in the steady state, λ_i takes the value:

$$\lambda_i = \frac{\sum_j \frac{du}{dq_i}\frac{dq_i}{d\mathbf{x}}\frac{d\mathbf{x}}{ds_i}}{\delta - f_i^{'}}, \; i = 1,\ldots,n$$

So the value of the ith ecosystem stock depends (a) on its regeneration rate relative to the yield on produced capital, indicated by the discount rate, and (b) on its marginal impact on the production of the set of marketed outputs, $\mathbf{q}$, through the effect it has both on other ecosystem stocks, $\mathbf{s}(t)$, and on marketed inputs, $\mathbf{x}$.

A third concern is particularly relevant to the problem of the regulating services. It relates to the way in which valuation studies address the problem of uncertainty (Winkler, 2006b).

Since the value of ecosystem stocks is the discounted stream of net benefits they provide, it is sensitive to uncertainty about the environmental and market conditions under which they will be exploited. Most valuation studies simply sidestep the problem. Others address it indirectly through the discount rate. Since uncertainty is typically an increasing function of time, if the future is discounted sufficiently heavily the more uncertain consequences of the use of ecosystem stocks are effectively ignored. Where uncertainty about the future consequences of the use we make of the environment includes the likelihood of severe and irreversible consequences, this is not satisfactory. Since social-ecological systems are complex, coupled and adaptive, the capacity to predict the future consequences of current actions is limited at best. Such systems have the usual properties of non-linearity, path dependence and sensitivity to initial conditions. Any estimate of the value of stocks is conditioned on the capacity to predict those consequences, as is the choice between adaptation to and mitigation of those consequences.

The regulating services affect the distribution of outcomes, and in particular, they affect both variation about the mean response and the likelihood of extreme responses. The next section considers how far this is currently being addressed in attempts to value ecosystem services, at least in the pages of *Ecological Economics*. This is an illustrative exercise only. There are many more papers by ecological economists published in other journals, and these are not surveyed. My interest is more in the way that the problem is being addressed by ecological economists than with the results of the very many valuation studies that continue to be published.

Valuing regulating ecosystem services

The proliferation of studies in *Ecological Economics* of different ecosystem services is evidence that ecological economists are indeed trying to meet the challenge posed by the MA. However, the focus of such studies suggests that there is more to be done. On the plus side, a two-part paper by Winkler (2006a, 2006b) has recently raised concerns about the way that ecosystem services have been evaluated in the past, and has attempted to redress the problem by constructing an integrative model of a coupled social-ecological system under uncertainty. Appropriately, the model seeks both to understand the physical interactions between the elements of the system, and the preferences that govern people's perception of the importance of environmental conservation. Part of the problem with many existing studies is that the valuation of the environmental stocks that underpin the production of ecosystem services is limited by the perceptions of the users of those services.

Consider a recent study of the value of ecosystem services from Opuntia scrublands in Peru (Rodriguez et al, 2006). The authors' own evaluation of the range of potential ecosystem services from the scrublands identifies erosion control, habitat provision, nutrient retention, water regulation and supply as amongst the more important services. However, the study focuses on the resource users' perceptions of the value of the resource, using semi-structured surveys to elicit preferences. It finds, not surprisingly, that the users' own valuation of the resource is wholly dominated by the products it yields – coccineal, fruit, fodder and fuel. Without going further, nothing could be said about the value of other services.

One problem is that some researchers do argue that stated preference methods are appropriate for at least some regulating services (de Groot et al, 2002), which I doubt. A more significant problem is the growing use of value (benefit) transfer techniques in ecosystem service valuation studies. This may be sensible in the case of carbon sequestration services, where the contribution of carbon sequestration to the general circulation system is independent of where it takes place (e.g. Songhen and Brown, 2006). However, it makes less sense where the benefits of ecosystem services depend heavily on local conditions. Viglizzo and Frank (2006), for example, use the 1997 biome values obtained by Costanza et al (1997) in a recent study of the impact of land use changes in the Del Plata Basin in South America. This is unlikely to yield useful results for various well-understood reasons.

Less problematic is the use of stated preference methods to value the outputs of activities for which there are no well-functioning markets, and then to use this to derive the value of regulating and supporting ecosystem services from this. Allen and Loomis (2006), following Goulder et al (1997), use such an approach to derive the value of species at lower trophic levels from the results of surveys of willingness to pay for the conservation of species at higher trophic levels. Specifically, they derive the implicit willingness to pay for the conservation of prey species from direct estimates of willingness to pay for top predators. They refer to this as a form of quasi-benefit transfer. They make the point that it is not necessary for people to understand the trophic structure of an ecosystem, since their willingness to pay for top predators effectively captures their willingness to pay for the whole system. While this ignores any value attaching to the diversity of species or to other ecosystem services other than habitat provision, it is at least a constructive use of stated preference methods.

Where there are prices for the outputs of activities, then derived demand (production function) methods are appropriate- and there are a growing number of studies that use such an approach (e.g. Barbier, 2000; Nunes et al, 2006; Matete and Hassan, 2006). These studies identify values for ecosystem services that represent at least part of the shadow value of those resources. Like the study by Allen and Loomis (2006) they apply knowledge of ecosystem functioning and processes in order to derive the value of supporting and regulating ecosystem services. To this point, however, there are very few studies of the value of regulating services in changing the distribution of outcomes. Studies that derive the value of ecosystem services look for the partial derivative of the production function with respect to the service to be valued, but do not consider the marginal impact of a change in the service on the second (or higher) moments of the distribution of output.

The way ahead

Of course the valuation of ecosystem services is not the only subject that is going to attract the attention of ecological economists over the next few years, but in the aftermath of the MA I suggest that it is going to be an extremely important topic. I believe that it will develop in ways that deepen our understanding both of the interactions between ecosystem functioning, ecosystem processes and the production and consumption activities of economic agents. It will also enable us to begin to evaluate mitigation and adaptation strategies for in a more systematic manner.

When output is measurable and either has a market price or one can be imputed, determining the marginal value of the resource is relatively straightforward, providing that the connection between ecological functioning, ecosystem services and human production processes are well-understood. If output cannot be measured directly, then either a marketed substitute has to be found, or complementarity or substitutability between ecosystem services and one or more marketed inputs has to be specified explicitly. Some ecological economists are exploring these relationships, as the paper by Nunes et al (2006) illustrates, so strengthening understanding of the way that social and natural processes are linked. But the persistence of valuation studies that neglect underlying ecosystem services and the inappropriate use of value transfers suggests that there is much to be done.

There is, however, a great opportunity here for ecological economics. No other field is better placed to explore the interface between ecological and social processes, particularly in managed or heavily impacted ecosystems. This opportunity extends well beyond interactions in agriculture, forestry or fisheries. There are a range of novel scientific questions to be posed about the interdependence between biodiversity, ecosystem functioning, ecosystem services, economic, technical and institutional change at the global scale (Dirzo and Loreau, 2005). New research methodologies are being developed to clarify the linkages between biodiversity change and ecosystem functioning (Loreau et al, 2002; Caldeira et al, 2005; Hooper et al, 2005) and human well-being (Perrings, 2005; Heal et al, 2005).

The main challenge, however, is to develop predictive models of the impact of external forcing functions – such as climate on - on ecosystem services. The application of dynamic bioclimatic-envelope modeling techniques to predict species response to changes in climate has improved the capacity to connect land-use change, biodiversity distributions, and ecological functioning (Pearson and Dawson 2003; Wilson et al 2005; Sutherland 2006). Evaluation of the economic consequences of climate change (Fankhauser and Tol 2005; Tol 2005) has raised important issues about the modeling techniques appropriate to the inherent uncertainties of the problem. Ecological economics is well-placed to exploit these developments to improve our capacity to generate predictive models that will make it possible to evaluate the relative pay-off to adaptation or mitigation of climate (and the associated ecosystem) change.

There is scope for ecological economists to identify the effect of ecosystem change on the capacity of socio-ecological systems to absorb anthropogenic and environmental stresses and shocks without loss of value. This parallels work on the resilience of coupled systems within the Resilience Alliance (Kinzig et al. 2006; Scheffer et al. 2000; Walker et al. 2004; Walker et al. 2006) and is, again, grounded in an analysis of the linkages among biodiversity change, ecological functioning, ecosystem processes, and the provision of valued goods and services. Although it is recognized that ecosystem change has economic implications because of the value it has through insurance against environmental shocks (Balmford et al. 2002), most research still neglects this dimension of the problem. If we are to understand and enhance the resilience of coupled systems we need integrated models of the linkages between biodiversity and ecosystem services (Loreau et al. 2002; Naeem and Wright 2003; Reich et al. 2004; Hooper et al. 2005), and between biodiversity change and human well-being (Kontoleon et al. 2006; Finnoff and Tschirhart 2006; Baumgartner 2006).

In summary, while the Millennium Assessment has brought the analysis of ecosystem services within the domain of economics, and while ecological economics is perfectly placed to exploit the opportunities this brings, we have yet to seize these opportunities. My own view is that ecological economics has an obligation to develop the science needed to understand, model and predict the dynamics of coupled ecological-economic systems. Indeed, it is the raison-d'etre of the field. This is an exciting task. It does involve technical difficulties, but if successfully accomplished it has the potential to significantly improve the capacity of resource managers everywhere to navigate the challenges posed by globalization and climate change.

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Charles Perrings: An Appreciation

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Although I had known Charles by reputation even in the mid 1980s, I met him for the first time in the summer of 1991, when we both became associated with the Beijer International Institute of Ecological Economics, Stockholm - I as Chairman of the Scientific Board of the Institute and Charles as Director of the Institute's inaugural Biodiversity Programme. It is hard to imagine today how little ecologists and economists knew of one another's works at that time. Even though environmental and resource economics was an established field, the models that economists worked with for the most part contained a single resource; moreover, the analysis was frequently conducted in a partial equilibrium setting - meaning that the resource in question was regarded as inessential. As Karl-Goran Maler, the Institute's Chairman, had a contempt for authority, Charles had a free hand in defining his Programme, choosing its participants, and vetting the products that grew out of it. The volumes that emerged (Perrings *et al*., 1994, 1995) are pioneering and have had a great influence on those of us who take Nature's non-linearities seriously.

Charles, I believe, was one of the first economists to appreciate the importance of ecological services in economic life. He was also one of the first economists to recognise the importance of collaboration with ecologists if we are to make progress. His masterly paper with Brian Walker ("Biodiversity Loss and the Economics of Discontinous Change in Semi-arid Rangelands") in Perrings *et al*.

(1995) identified what are now called "tipping points" that economies arrive at when the underlying ecosystems reach points of bifurcation. The paper is technical (I mean mathematically, and not simply in terms of the sophistication in the ecology and economics deployed by the authors). That paper alone provides a compelling reason for regarding Charles as one of the founders of *ecological economics*. Collaboration among ecologists and economists is becoming routine in this new field. Charles has played a major role in making that collaboration happen. (This is reflected well in a more recent publication: Perrings, 2000). That such collaborative research isn't easy is proven by the fact that the other research programme that the then newly reconstituted Beijer Institute initiated in 1991, namely, Complex Systems, was a failure. Charles' subsequent work on the economics of ecosystem resilience (again in collaboration with Brian Walker) has also been pioneering. Their idea has been to arrive at numerical indicators of "resilience" in canonical models in ecology. This work is likely to have far reaching implications in the design of environmental policy.

Charles' growing influence in ecological economics led an international group of ecologists and environmental and development economists to appoint him in 1996 Editor of *Environment and Development Economics* (*EDE*), a quarterly journal (published by Cambridge University Press) in the field of environment and development. As before, his intellectual leadership since the journal's inauguration has been exemplary: the journal has helped to shape the way the subject has developed, with original contributions from economists in South Asia and, more significantly, sub-Saharan Africa. Charles' decision not to submit anything by himself in *EDE* during his tenure reflects his probity, but it has also robbed the rest of us of the pleasure of reading him there.

I don't know of any other economics journal that offers the reader editorial reflections on the articles that it publishes. *EDE* is educational, it has variety, has intellectual clout, and is morally serious. It's also enjoyable to read. For me, it's the most exciting journal in either environmental or development economics.

Charles is an outstanding professor, researcher, leader of research, and expositor. His lectures are calm, reflective, and rigorous. He is generous to a fault toward the works of others. And if he is at ease with others, it is because he is at ease with himself. I cannot imagine a greater virtue.

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Reconsideration of Dynamic Utility Optimisation and Intergenerational Equity in Sustainable Development Studies

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Abstract

Many sustainable development studies have employed intergenerational social welfare functions based on dynamic utility optimisation models (RCK models), which were pioneered by Ramsey and elaborated by Cass and Koopmans. This line of studies, however, rarely scrutinise the relevance of the fundamental assumption that dynamic optimisation in RCK models directly addresses intergenerational equity issues. This paper critically examines this assumption and presents an alternative interpretation: dynamic optimisation in RCK models signifies that an individual's current utility level is determined not only by the activities and environments at this moment but also by future expectations about them. This interpretation leads us to reconsider treatment of intergenerational equity in sustainable development studies. This paper claims that intergenerational equity in sustainable development is better represented as certain "survival conditions" rooted in physical facts instead of sustainability conditions based on value judgements. As one candidate of such survival conditions, an approach based on an ecological resilience concept is illustrated.

Keywords: *sustainable development, social discounting, intergenerational equity, dynamic optimisation*

JEL Classification: *D63, D91, Q01*

Mathematics Subject Classification 2000: 91B76

1. INTRODUCTION

Dynamic utility optimisation models pioneered by Ramsey (1928) and elaborated by Cass (1965) and Koopmans (1965), which this paper refers to as RCK models, have served as an important tool to investigate implications of sustainable development

[1] I would like thank Peter King, Richard Howarth, and two anonymous referees for their valuable comments. All errors are my own. An earlier version of this paper was presented to the Third World Congress of Environmental and Resource Economists, Kyoto, 3-7 July 2006.

(SD). The ability of RCK models to address intertemporal resource allocation decisions doubtlessly demonstrates their congeniality to SD problems. Furthermore, dynamic optimisation in RCK models has often been associated with intergenerational resource allocation, and many authors have discussed intergenerational equity, a core element of SD, based on intergenerational social welfare functions derived from RCK models (e.g. Toman et al. 1995). In fact, most "sustainability criteria" for SD in environmental economics are proposed in the context of this type of intergenerational social welfare functions (Chichilnisky 1996, Heal 1998). In spite of such an almost authoritative status, this line of SD studies rarely scrutinise the relevance of their fundamental assumption that dynamic optimisation in RCK models directly addresses intergenerational equity issues.

This paper critically examines this assumption and raises an alert over the necessity of careful distinctions between (i) intertemporal optimisation and intergenerational optimisation, and (ii) private optimisation and social optimisation, in RCK frameworks.[2] This examination allows us to reconsider the meaning of dynamic utility optimisation in SD studies. This paper derives an alternative interpretation which could potentially reconcile disputes on discounting in SD studies. Moreover, this paper reconsiders treatment of intergenerational equity in SD studies.

2. MEANING OF DYNAMIC OPTIMISATION IN RCK MODELS

The utility function in basic RCK models takes the following form.

$$U = \int_0^\infty e^{-\rho t}\, u(c(t))dt \qquad (1)$$

Note that population is assumed to be constant for simplicity. The conventional interpretation is as follows. *U* is lifetime utility of an immortal dynasty, *u*( ) is an instantaneous utility function in which utility level at time *t* is determined by consumption at time *t* denoted by *c*(*t*). ρ denotes a constant discount rate or more precisely a pure rate of time preference.

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2 Burton (1993) distinguishes between intertemporal discount rates of members of the society and intergenerational equity considerations in his overlapping generations model and shows their implications for the optimal resource harvesting decisions.

In SD literature, it is common to customise this basic form for research purposes. Various customised utility functions can be represented by the following more generalised form.

$$U = \int_0^\infty D(t)u(\mathbf{z}(t))dt \qquad (2)$$

A vector $\mathbf{z}(t)$ represents any activities (e.g. consumption, leisure activities) and environments (e.g. stocks of pollutants) at time *t,* which affect utility level at time *t*.[3] $D(t) \in (0, 1]$ is a general form of discount factor which could be exponential, hyperbolic, or whatever. The discussion in this paper is not affected by this generalisation at all. In either form, the crucial assumptions are (a) the utility level experienced at time *t* (instantaneous utility) is solely determined by the current activities and environments, and (b) the objective function is a net present value of future utilities for infinite time horizon that represents lifetime utility of an immortal dynasty. Although most SD studies take these assumptions as granted, these are in fact strong and arguable assumptions.

Even casual self-examination tells us that our current utility level depends not only upon the current conditions (activities and environments) but also upon our experiences in the past and our expectations in the future. The point in question is not that the assumption (a) is unrealistic, since no model can ever precisely replicate such complexity in a useful manner. The point is whether this assumption is a reasonable first order approximation of reality for our analytical purposes. An alternative assumption could be that the utility level currently experienced is determined not only by the current conditions but also by the discounted sum of expected future “enjoyment” levels.[4] Following Gorman (1955) and Arrow and Kurz (1970) this paper employs the term “felicity” for referring to enjoyment derived only from the current conditions.

The alternative assumption employed by this paper significantly alters the interpretation of Equation 2. $u(\mathbf{z}(t))$ represents the *currently expected* felicity level to be

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[3] See Beltratti (1997) for the discussion on this vector.

[4] As Peter King pointed out, current utility level could be influenced by past experience as well, but it must be reflected by functional forms or parameter values of utility functions. Future expectation is influenced by the current decision, but the past experience is not.

experienced at time *t*. *U*, or more precisely *U*(0), is the utility level experienced at time 0, which is conventionally referred to as instantaneous utility. This interpretation is reminiscent of an influential work by Strotz (1956). He criticised the term instantaneous utility function as a "misnomer" since he acknowledged the possibility of utility experienced at a point of time "depending on the consumption of a later date" (Strotz 1956; Footnote 2; p.167).

Now let's reconsider the assumption (b). Based on the above alternative interpretation, utility level *U* is not associated with an immortal dynasty but with an individual or a household with a finite lifetime. Further, it is plausible that the time horizon of this individual or household is shorter than their lifetime. Then, we must examine the relevance of employing an infinite time horizon in Equation 2, following Aronsson et al. (2004) on this issue. Their argument is based on the fact that optimisation for finite time horizons must include a value function at the terminal time which represents the terminal value of "assets". Inclusion of the terminal time value function in dynamic utility optimisation seems consistent with the observation that people rarely plan to consume their asset completely during their lifetime, and even more so if the time horizon is shorter than the lifetime. Then they argue that for the optimal solution the value function must be a discounted sum of a stream of felicities along the optimal trajectories after the terminal time. Formally, it can be expressed as follows:

$$U(0) = \int_0^T D(t)u(\mathbf{z}(t))dt + V_{\max}(T),\ V_{\max}(T) = \int_T^\infty D(t)u(\hat{\mathbf{z}}(t))dt \qquad (3)$$

In the above equation $V_{max}(T)$ is the optimal value function at time *T* and $\hat{\mathbf{z}}(t)$ is the values of $\mathbf{z}(t)$ along the optimal trajectories. Mathematically, Equation 3 becomes exactly the same as Equation 2 along the optimal trajectories. The remaining question is whether we can correctly specify the optimal value function for future time *T*. The answer is "no" unless we could have perfect foresight, but it seems reasonable to assume that our guess is good enough to approximate the real utility function by Equation 3. This assumption makes it possible to avoid the difficulty in specifying the value function at the terminal period and in setting the length of the time horizon.

3. SOCIAL WELFARE FUNCTIONS AND DISCOUNTING

In his seminal paper, Bergson stated that the social welfare functions must represent prevailing values in the community because "only if the welfare principles are based upon prevailing values, can they be relevant to the activity of the community in question" (Bergson 1938; p.323). This is in fact a tough requirement due to the diversity in values of community members, and it must be solved through political processes. Most SD studies employing the RCK framework have not tackled this challenge explicitly. Either they simply assume the validity of Equation 2 as a social welfare function, or they do not mention that the objective function of their problem is a social welfare function. This casual treatment has caused some confusion in SD literature.

When we assume that social welfare is represented by the unweighted sum of individual member's utility and we normalise the total population as unity, the social welfare function takes the same form as the private utility function, but they represent different things. The private (instantaneous) utility function represents our psychological mechanisms of enjoyment and are empirically determined, while the same equation as a social welfare function represents normative and political value judgements of the society. This distinction provides a clue to reconcile the dispute over discounting.

Suppose that private agents make decisions based on dynamic optimisation of utility levels currently experienced, following my alternative interpretation, and that the social welfare is defined as the unweighted sum of individual utility, Equation 2 can be interpreted as the social welfare function representing social welfare of the current generation. It involves a discount factor but does not involve intergenerational aspects. $D(t)$ simply reflects our psychological facts and it can be observed through our behaviour and decisions.

Now suppose that a society decides to incorporate intergenerational equity issues into the social welfare function by taking into account the discounted sum of each generation's utility level.[5] Then, the social welfare function can be expressed as;

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[5] Burton (1993) employs the same assumption on his social welfare function.

$$SW(0) = \int_0^T G(s)U(s)ds = \int_0^T G(s)\left\{\int_s^\infty D(t)u(\boldsymbol{z}(t))dt\right\} ds \qquad (4)$$

SW(0) is intergenerational social welfare at the current moment, *U*(*s*) is utility level of generation *s*, and $G(s) \in (0, 1]$ is a social discount factor which describes the society's rule of weighting between different generations. *T* indicates a maximum distance of the furthest generations to be considered. In Equation 4, two discount factors *D*(*t*) and *G*(*s*) coexist; the former represents the empirical or positive discounting within a generation and the latter represents the ethical or normative discounting between generations (Burton 1993, Tol 1999, Arrow et al. 1996). This separation of two discount factors resolves the contradiction between normative and positive discounting approaches.

The claim that any adjustments of discounting for ethical considerations must be avoided is doubtlessly relevant for *D*(*t*). It is an empirical discount factor reflecting our psychological facts such as impatience. Hence *D*(*t*) is given and cannot be determined by our will.

Ethical considerations for intergenerational equity, however, can be reflected by *G*(*s*). For example, Ramsey's (1928) famous claim that discounting a future generation's utility is "ethically indefensible" is definitely convincing if it is about *G*(*s*). This intergenerational discount factor can only be determined based on value judgement, and hence it is subject to political decision.

To clarify these points, the intention is not to propose the above intergenerational welfare function (Equation 4) as an analytical tool to address SD issues. Rather, Equation 4 elucidates the inappropriateness of relying on dynamic social welfare optimisation for addressing intergenerational issues (Schelling 1995). As Goulder and Stavins (2002) assert, intergenerational considerations must be done outside cost-benefit analysis that is underpinned by dynamic social welfare optimisation. It does not, however, imply irrelevance of dynamic utility optimisation to SD studies. Private utility optimisation (Equation 3) remains a powerful tool to simulate responses of private agents to policies which have impacts on any activities or the environments denoted by $\boldsymbol{z}(t)$. Such simulations are instrumental in ascertaining the implications of SD policies.

4. INTERGENERATIONAL EQUITY IN SUSTAINABLE DEVELOPMENT

Intergenerational equity in SD is not about efficiency but about conservation of the basis of human survival.

SD as a global political agenda aims at achieving both poverty alleviation and environmental sustainability. An underlying recognition is that poverty alleviation requires economic development while conventional economic development has too often destroyed the basis of human survival such as rainforests, fertile agricultural lands or freshwater ecosystems (World Commission on Environment and Development 1987). In the famous WCED definition of SD, intergenerational equity is expressed as "without compromising the ability of future generation to meet their own needs" (World Commission on Environment and Development 1987; p.43). Rapid destruction of rainforests is regarded as unsustainable not because a decrease in natural capital may result in intergenerational inefficiency but because we intuitively fear that it may result in some irreversible catastrophe.

Suppose we can successfully alleviate poverty without undermining the basis of human survival. Future generations will set their own targets based on their values and preference system and will pursue those targets. They will not bother whether they can inherit less wealth, as a total or as any individual elements, than preceding generations, so long as they can inherit the world free from poverty and fatal environmental threats to human survival. It is the same as the present generation not bothering whether previous generations were wealthier than us or not.

This clarification suggests that intergenerational equity in SD is better represented as certain "survival conditions" rooted in physical facts. A candidate is the maintenance of integrity of ecosystems that underpin the life-support systems such as sound hydrological cycle, nutrients cycles, and soil systems. This is obviously a necessary condition, but not a sufficient condition, to secure the basis of human survival. We know very little about ecosystems' behaviour and operationalising this condition is a real challenge, but an approach based on the concept of ecological resilience appears to be promising (Common and Perrings 1992, Kojima 2005).

Resilience of a system, after Holling (1973), can be defined as the maximum perturbation of the system that does not cause the system to leave its original stability domain (Perrings and Dalmazzone 1997). A candidate of survival conditions based on this concept is that perturbations in life-support ecosystems caused by development should be less than the ecosystems' resilience.[6] It must be possible to set certain safe minimum standards based on the precautionary principle for this purpose, although we currently have very limited ability to understand behaviours of ecosystems due to non-linearity, path-dependence, discontinuity, and uncertainty associated with ecosystems (Perrings et al. 1995). It is well known that many indigenous peoples have shaped their life style so that they can avoid losing ecosystem resilience. Needless to say, the more scientific knowledge we have about ecosystem behaviour, the less strict safe minimum standards we can adopt.

A good example of this approach would be the Kyoto Protocol. Its thrust was the fear of irreversible catastrophe caused by global warming such as cessation of ocean current or irreversible changes in ecosystems, in other words, loss of ecosystem resilience. The Kyoto Protocol demonstrates the ability of the international community to establish political consensus on safe minimum standards of GHGs emissions for avoiding such loss of resilience in spite of limited scientific knowledge of climate change mechanisms and ecosystems' behaviour under the potential global warming.

There are many other threats to life-supporting ecosystems including desertification, soil degradation, excessive groundwater exploitation, and so on. Sustainable development requires establishment of safe minimum standards to maintain resilience of these ecosystems in parallel with accumulating knowledge about them.

5. CONCLUSIONS

Through examination of basic assumptions underlying intergenerational social welfare functions based on RCK models, this paper presents an alternative interpretation that dynamic optimisation in RCK models simply means that our utility level at this moment is determined not only by current activities and environments but also by future expectations.

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[6] For more discussion on this approach, see Kojima (2005).

According to this interpretation, the intergenerational social welfare function based on RCK models involves a double integral (or a double summation) and consequently it can accommodate two discount factors; one is a private discount factor representing our psychological tendency to devalue distant future events, and the other is a social discount factor representing a weighting rule of intergenerational welfare comparisons which must be determined by value judgements and through political processes. If both ethical and empirical discount factors coexist, the seemingly unbridgeable dispute over choice of discount factor between ethical (normative) and empirical (positive) approaches disappears. Separation of private and social discount factors makes it clear that there is no convincing ground to devalue the utility of future generations (Ramsey 1928).

The above discussion prompts reconsideration of adequate treatment of intergenerational equity in SD studies. If SD primarily aims at achieving both poverty alleviation and environmental sustainability, as the Brundtland Report (World Commission on Environment and Development 1987) claimed, intergenerational equity in SD is not about efficiency but rather about conservation of the basis of human survival. Discounting matters only if efficiency is the main concern. This paper claims that intergenerational equity in SD is better represented as certain “survival conditions” rooted in physical facts instead of sustainability conditions based on a value judgement such as discounting. An approach based on an ecological resilience concept is illustrated as one candidate of such survival conditions.

The alternative treatment of dynamic utility optimisation and intergenerational equity proposed by this paper is expected to sort out some confusion in sustainable development literature. For instance it elucidates why there is no convincing way to incorporate intergenerational equity into cost benefit analysis, which is based on RCK framework, of long-term environmental problems such as global warming. Large time scale of global warming is required because of its potential long-run effects on ecosystem resilience, and such long-term concern can be accommodated by survival conditions but not by cost benefit based sustainability conditions. Equation 4 and the proposed ecological resilience approach corroborate the claim that intergenerational considerations must be done outside cost-benefit analysis (Schelling 1995, Goulder and Stavins 2002).

As a final conclusion of this paper, it must be emphasised that the mainstream definition of SD in ecological/environmental economics as eternally non-declining well-being has an appeal only for the richer portion of the world population. It is high time for ecological/environmental economics as a discipline to reconsider the original objectives of SD if it means to address intra- and inter-generational equity issues responding to the global concern. Although our knowledge about ecosystem behaviour is limited, the ecological resilience approach illustrated by this paper can be implemented with the current level of knowledge by setting safe minimum standards based on the precautionary principle as exemplified by the Kyoto Protocol.

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Solid Waste Market Distortions and Recycling

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ABSTRACT

Solid waste management is an important topic in environmental economics, and there is a need for providing better incentives to further optimize the chain of materials and waste. We investigate market distortions caused by flat fee pricing in the solid waste market and we show how flat fee pricing influences households in their decisions to recycle, separate or dispose of rest waste. We develop and apply a general equilibrium model for the solid waste market and describe in detail how market distortions, as a result of flat fee pricing, can be analyzed in an applied general equilibrium framework. A numerical example demonstrates the effects of flat fee pricing on both waste generation and recycling. In the presence of flat fee pricing, the cost of recycling has no impact on the behavior of households, and thus households are not responsive to recycling subsidies. The results show that introducing a unit-based pricing scheme for waste collection can stimulate recycling far more effectively than subsidies and improve the effectiveness of recycling policies.

KEYWORDS: Applied General Equilibrium Modeling; Market distortions, Policies; Recycling; Waste management

JEL classification: D58; H21; Q28

Mathematics Subject Classification 2000: 91B76, 91B32

INTRODUCTION

Governments still fail to achieve a decoupling between waste generation and economic growth. For example, since 1950, the quantity of waste generated in the Netherlands has more than tripled, from about 17 Mtonnes in 1950 to about 67 Mtonnes in 2000 (WMC, 2003). The European Environment Agency (EAA, 2000) has demonstrated that waste generation in the European Union is still coupled with economic growth, making it impossible

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to pursue economic growth without creating increasingly serious waste management problems.

Several studies have shown that failure to achieve a decoupling between waste generation and economic growth can be attributed to flat fee pricing for waste collection (see for example Miedema, 1983)[1]. Most studies on the effects of distortions in the solid waste market have used a partial equilibrium approach. Wertz (1976) was the first to analyze the effects of a user charge on solid waste disposal. Miedema (1983) analyzed the effects of other distortionary characteristics of the solid waste market, like virgin material-biased tax policies, virgin material-biased regulations, and indirect subsidization of virgin materials. Other empirical studies include Jenkins (1993), Hong *et al.* (1993), Miranda *et al.* (1994), Morris and Holthausen (1994), Sterner and Bartelings (1999), and Kinnaman and Fullerton (2000). The overall conclusion of these studies is that solid waste generation is sensitive to user fees; the introduction of user fees can cause a substantial reduction in solid waste generation, especially if they are combined with programs that enlarge the public awareness for the solid waste problem.

However, thoughtless construction of waste handling tariffs might not have the desired effect and can encourage illicit dumping, burning, or other improper disposal. Fullerton and Kinnaman (1996) estimate that about 28% of the decrease in waste generation may be caused by increased illegal disposal. Empirical studies, like Jenkins (1993) and Miranda and Aldy (1998), also report instances of increased illegal dumping. These results, however, are contradicted by other empirical studies. For example, Miranda *et al.* (1994), Strahman *et al.* (1995), Nestor and Podolsky (1998), Podolsky and Spiegel (1998), Sterner and Bartelings (1999) and Linderhof *et al.* (2001) found no significant evidence of increased illegal disposal.

Despite the risk of illegal disposal, the unit-based price is one of the most effective policy options to provide an incentive to increase prevention and home composting. For instance, Calcott and Walls (2002) find that a modest disposal charge will always be part of the set of optimal policy instruments. Shinkuma (2003) argues that even if illegal disposal is an option, the unit-based pricing system will still provide a second best optimum as long as the price of recycled material is positive.

In this paper we extend the analysis of the impact of different pricing systems on waste collection and recycling. In contrast to previous research, we build a general equilibrium model focusing on the complete product chain from extraction to production to consumption and finally to recycling and waste treatment. By developing this more complex general equilibrium model, we can analyze whether policy changes in the end phase of a product also has an impact on the extraction sector, and production phase of a product, and consequently on the prices of and demand for consumption goods. Thus we can show the *direct* effects of waste policy changes on recycling behavior and solid waste generation and the *indirect* effects through price changes and changes in consumption patterns. Inspection of the indirect effects can also shed light on the importance of using a general equilibrium approach in comparison to a partial equilibrium framework.

The paper evaluates the effectiveness of unit-based pricing versus a curbside recycling program, which stimulates recycling by subsidizing recycling efforts. We analyze whether policies aimed at promoting recycling and waste reduction are more effective under a flat fee or under unit-based pricing. Policymakers prefer recycling to incineration or landfilling since recycling will in general have less environmental effects and will help in closing the material cycle. This is in line with the idea of the waste hierarchy[2]. Although some studies have shown that recycling may not be preferred in some cases (see for example Barrett and Lawlor, 1997), we will not address this issue.

The paper is structured as follows: Section 2 describes the model and shows how both flat fee pricing and unit-based pricing can be included in an applied general equilibrium model. Section 3 presents a numerical example in which the consequences of flat fee pricing versus unit based pricing mechanisms and the possibilities to promote recycling are analyzed. Data used in this example are based on the Netherlands in 1996. The model is constructed, however, in such a way that results can be generalized to other countries. Section 4 concludes.

DESCRIPTION OF THE MODEL

1.1 General description of the AGE model

For studying the various waste management options we develop an applied general equilibrium model for a national economy, including a solid waste market. Consumers maximize utility and producers maximize profit. An applied general equilibrium model can be solved in various ways, also called formats[3]. In this paper the Negishi-format (as proposed by Negishi, 1972) is chosen as the preferred tool for building an applied general equilibrium model. One of the advantages of the Negishi format is the relative ease with which non-convexities, such as a flat fee-pricing scheme, can be implemented (see also Ginsburgh and Keyzer, 1997).

1.2 General introduction to the model structure

In the model, two types of actors are distinguished: consumers and firms. Consumers buy goods and services and supply production factors; firms produce goods and services and use both production factors and intermediate goods as inputs to production. We distinguish two types of consumers -private households and the government- and eight types of producers, each producing one unique good. These producers are: (1) a producer of extraction services producing virgin material; producers of (2) agricultural goods, (3) industrial goods and (4) commercial services; (5) a producer of waste recycling services; (6) a producer of waste collection services and producers of (7) waste incineration services and (8) waste landfilling services. The life cycle of materials for the hypothetical economy is shown in Figure 1.

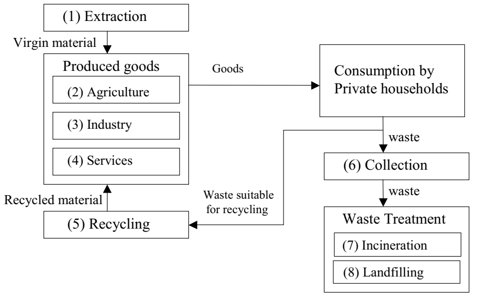

Figure 1 Schematic representation of the economy

Private households consume agricultural goods, industrial goods, and commercial services, such as transport, distribution, retailing, and sale of goods, but also insurance and bank services; the government consumes only commercial services. Only private households generate waste[2]. The government does not produce waste. Waste generation is triggered by the consumption of agricultural and industrial goods and not by the consumption of commercial services[3]. Waste will have to be either recycled or collected by the municipality and treated by the waste treatment sector.

In the benchmark data set, private households pay a flat fee for collection of waste. According to such a pricing scheme, the marginal costs of waste collection equal zero. Thus the equilibrium prices for waste collection of rest waste equal zero. Modeling zero marginal prices in a general equilibrium model presents some problems. Since demand is determined by marginal prices, a zero marginal price could lead to infinite demand. Therefore, to implement this in the Negishi format, a subsidy-*cum*-tax scheme is used. In the subsidy-*cum*-tax scheme, as illustrated in Figure 2, households pay the equilibrium price for waste collection. The government, however, reimburses the households with exactly the same amount in the form of a subsidy, thus the price of waste disposal as perceived by the consumer equals zero. The government will finance the costs of the subsidy by demanding a direct tax from the private households for waste collection[4]. One should keep in mind, however, that the direct tax does not necessarily has to be as high as the costs of the subsidy. In fact as described in the next section in the Netherlands the direct tax only covers

[2] For simplicity we completely focus on domestic waste and we have assumed that the firms do not generate waste. Although this limits the analysis we feel that it is justified, because there is almost no connection between treating municipal solid waste or industrial waste. Therefore waste treatment of industrial waste will not directly affect the price of waste treatment of municipal solid waste.

about 95% of the costs of waste collection. Implicitly this means that the government is subsidizing part of the costs of waste collection.

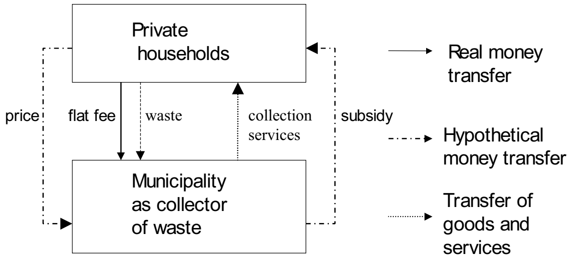

Figure 2 The subsidy-cum-tax scheme

The model used in this paper describes a closed economy. Allowing import and export of waste and goods will hardly affect the results because we are focusing on the effects policy measures have on the generation of household waste. Households in this case make the decision whether or not to recycle waste. Both recycled waste and rest waste are collected by the municipality. The households do not have the option of transporting waste to a treatment center themselves. Modeling an open economy will only have an impact due to a larger supply of waste extensive products. However, it can be expected that these effects will be minimal as the expected waste prevention due to the introduction of unit-based pricing or recycling subsidies is minimal. Furthermore, international trade in waste is still rather uncommon.

1.3 Description of the model

In the Negishi format, total welfare is maximized subject to the relevant balance constraints and production possibilities (Ginsburgh and Keyzer, 1997). The total welfare function is shown in equation 2.1. Total welfare *(TWF)* is determined by the sum of the weighted utilities (u_i) of all consumers ($i=1,...,I$) . Consumers derive utility from consumption of produced goods (x_i^g) where $g=1,...,G$ denotes the produced goods. The utility of each consumer is weighted by a factor α_i, the so-called Negishi weights[5].

$$TWF(\alpha) = \max \sum_i \alpha_i u_i(x_{i,g}) + \xi \; TW \tag{2.1}$$

$$x_{i,g} \geq 0 \;, \; all\; i \;\;, \;\; y_j \;\; all\; j$$

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[3] *Production* of commercial services could lead to generation of waste. *Consumption of these services*, however, will not lead to substantial amounts of waste generation due to the specific nature of commercial services; therefore, we have omitted this from the model.

To implement the subsidy-*cum*-tax scheme as discussed in the previous paragraph, we need to calculate two prices for waste collection: the equilibrium price, which equals the marginal production costs and the perceived price, which equals zero. The proper way of modeling this is by adding a subsidy term (ξTW) to the welfare function (see equation 2.1) where TW stand for the total amount of non-recyclable waste generated and ξ for the cost of the subsidy per unit of waste[6].

The subsidy wedge (ξ) is defined as the difference between the equilibrium price for waste collection (p_{cs}) and the perceived price ($p_{cs,sub}$). In the flat fee case the perceived price of waste collection equals zero, so the subsidy wedge equals the social costs of waste collection. In the unit-based price case, the perceived price of waste collection equals the marginal costs of waste collection so the subsidy wedge equals zero.

The total benefits of the subsidy are only added to the social welfare function to change the perceived price of waste collection. It does not imply that introducing subsidies would positively influence social welfare[7]. Prices for waste collection are determined by the balance equation for waste collection services:

$$TW \le y_{cs} \qquad \perp p^{w} \tag{2.2}$$

$$\sum_i x_{i,cs} \le TW \qquad \perp p_{cs,sub} \equiv p_{cs} - \xi \tag{2.3}$$

Where y^w equals the total production of waste collection services.

The equilibrium price of each commodity is determined by taking the marginal value of the corresponding balance equation (this is symbolized by $\perp$p). In the first balance constraint (2.2) the shadow price of waste collection is calculated. This price equals marginal production costs. In the second balance constraint (2.3) the shadow price of waste collection, as the households perceive it, is calculated. This price equals the equilibrium price minus the subsidy.

Generation of waste is triggered by consumption of agricultural and industrial goods. The most straightforward way to model waste generation is to assume the amount of waste generated after consumption of a good is equal to the original material inputs to produce that same good. Waste generation, however, has a dynamic dimension; not all products become waste immediately after consumption. To include this dynamic aspect of waste generation in a comparative static model, instead of linking waste generation directly to material input, we assume that waste generation is equal to a certain fraction of the produced good[8]. Waste generated per consumer (w_i) is equal to a fixed percentage (β) of consumption.

$$w_i = \sum_g \beta_g x_{i,g} \tag{2.4}$$

It is assumed that all waste generated is either collected *(x_{cs})* or recycled *(x_r)*, the model does not take into account illegal disposal of any kind. ´Environmental awareness´ may affect the choices of households regarding recycling, but this is not explicitly considered in the model, due to a lack of empirical information.

$$x_{i,r} + x_{i,cs} = w_i \tag{2.5}$$

All firms produce commodities y_j within their given production set Y_j. The production set for each of the firms in the model is given by a nested CES production function that depends on the input of capital *(k)*, labor *(l)*, virgin material *(q_v)*, recycled material *(q_r)* and intermediate inputs *(q_g)*.

Finally, a common requirement in applied general equilibrium models is that demand should equal supply for each commodity. This is ensured in various balance constraints[9].

A NUMERICAL EXAMPLE

The model discussed above is applied in a numerical example with stylized data for the Netherlands. The economic data used in the numerical example are based on the economy in the Netherlands in 1996 (Statistics Netherlands, 1998). We have chosen to use data from 1996 because at that time the Netherlands had hardly started with modern waste management policies. After 1996, they introduced landfilling taxes, landfilling bans, waste separation, and municipalities started experimenting with unit-based pricing. By using data from 1996 we do not have to separate the effects of these policies on the choice between recycling, landfilling, and incineration from the effects of new policies we want to examine.

1.4 Parameter values used in numerical example

Parameter values are based on the accounting matrix displayed in Table 1, which describes the initial equilibrium. Supply, *i.e.* producer outputs and consumer endowments, are given positive values; demand, *i.e.* producer inputs and consumption, are given negative values[10].

All prices are normalized to unity except the price of waste collection. As explained in section 2, waste collection basically has two prices: a perceived price and a social price. The perceived price equals the total fee divided by the total demand for waste collection. The social price equals the total fee plus the total amount paid by the government for waste collection divided by the total demand. We have chosen to normalize the perceived price for waste collection, which means that the social price for waste collection (which is shown in Table 1) is higher than unity.

Government expenditure is kept constant in the model at the benchmark level to avoid modeling problems concerning public expenditure incidence. When the income of the government changes in the model due to policy measures, the model simulates the private households compensating the government through a lump-sum transfer.

Table 1 *Benchmark social accounting matrix (expenditures in Billion Euros) and standardized prices (last column).*

	Agri	Indu	Serv	Extr	Rm	Rec	Col	Incin	Land	Cons	Gov	C-sum	Price
Agri	18.18	-8.18	-0.45	0	0	0	0	0	0	-9.55	0	0	1.00
Indu	-5.45	99.55	-21.82	0	0	0	0	0	0	-72.27	0	0	1.00
Serv	-1.82	-25.91	239.55	0	0	0	0	0	0	-202.73	-9.09	0	1.00
Extr	-0.45	-14.45	-0.91	15.82	0	0	0	0	0	0	0	0	1.00
Rm	0	-1.82	0	0	0.18	0	0	0	0	0	0	0	1.00
Rec	0	0	0	0	0	0.11	0	0	0	-0.11	0	0	1.00
Rw	0	0	0	0	-0.11	0.11	0	0	0	0	0	0	1.00
Col	0	0	0	0	0	0	0.43	0	0	-0.43	0	0	1.05
Incin	0	0	0	0	0	0	-0.27	0.27	0	0	0	0	1.00
Land	0	0	0	0	0	0	-0.09	0	0.09	0	0	0	1.00
K	-8.18	-21.27	-89.55	-12.41	-0.02	-0.18	-0.05	-0.23	-0.07	122.73	9.11	0	1.00
L	-2.27	-29.55	-126.82	-3.41	-0.05	-0.05	-0.05	-0.05	-0.02	162.36	0	0	1.00
Fee	0	0	0	0	0	0	0	0	0	-0.43	0.43	0	1.00
Subsidy	0	0	0	0	0	0	0	0	0	0.45	-0.45	0	1.00
R-sum	0	0	0	0	0	0	0	0	0	0	0	0	

Note: 'Agri', 'Indu' and 'Serv' stands for the three producer sectors of consumer goods (Agriculture, Industry and Commercial services); 'Extr' stands for the extraction sector; 'Rm' indicates the production sector of recycled material; 'Rec' indicates the production sector of recycling services; 'Rw' stands for waste suitable for recycling; 'Col' is the collection sector; 'Incin' indicates the production sector of incineration services; 'Land' indicates the production sector of landfilling services; K and L stand for the primary production factors capital and labor; Fee is the flat fee households pay to the government for waste collection; Subsidy stands for the total amount of money the government gives for waste collection as a subsidy to the households; 'Price' gives the prices of all commodities; 'R-sum' is the sum of a column; 'C-sum' is the sum of each row.

Source: Statistics Netherlands, 1998

Substitution elasticities for the CES production functions for the different sectors are given in appendix A. The private utility function is of the Cobb-Douglas type and depends on consumption of agricultural goods, industrial goods, and commercial services. The government only consumes commercial services. The initial Negishi weights are determined on the basis of the initial income (received from selling capital and labor).

In the benchmark dataset about 12 million tons of solid waste is generated per year. The waste percentage of agricultural goods (β=46%) is smaller than the waste percentage of industrial goods (β=69%)[11]. Of the waste generated about 20% is recycled and 80% is collected for waste treatment (either landfilling or incineration). Most of the waste collected is incinerated (75%); the rest is landfilled. Private households in the Netherlands pay about 0.43 billion Euro per year for municipal solid waste collection in the form of a flat fee, this is equivalent to 0.1% percent of GDP. This is slightly lower than the real cost of waste collection, which equals 0.45 billion Euro per year (the costs coverage rate equals 95%, WMC, 1997).

1.5 Different scenarios

To analyze the effects of unit-based pricing and recycling subsidies on the generation of waste we will examine three different scenarios: (i) pro-recycling scenario, (ii) unit-based price scenario, (iii) pro-recycling plus unit-based price scenario. It should be noted that these scenarios are not particularly meant as policy recommendations, but they are used to show how distortions caused by flat fee pricing influence household behavior. The results of each of these scenarios are compared to the results of the benchmark case. The benchmark case is described by the data presented in section 1.4 and contains the flat fee pricing scheme.

In the first scenario, recycling is promoted by introducing a subsidy on recycling. Due to the introduction of a subsidy, recycling will be 30% less expensive. The flat fee-pricing scheme is not changed. This policy is labeled ‘pro-recycling scenario’.

In the second scenario the flat fee-pricing scheme is replaced by a unit-based pricing scheme. This policy is labeled ‘unit-based price scenario’. In this scenario the households pay the equilibrium price for waste collection equal to the costs of producing waste collection services[12].

In the third scenario both the variable pricing scheme and the recycling subsidy are introduced. By comparing these results with the results of scenario 1, we will show how the effectiveness of the recycling subsidy is affected by the flat fee scheme. This scenario is labeled ‘pro-recycling plus unit-based price scenario’.

1.6 Results

1.6.1 Pro-recycling scenario

In the first policy scenario, the price of recycling is reduced by introducing a subsidy on recycling efforts.

Table 2 *The main variables for the ‘Pro-recycling scenario’ as compared to the ‘Benchmark case’ (expenditures in Billion Euro) and the percentage change*

	Benchmark	Pro-recycling	% change
Demand by private households			
Agricultural good	9.55	9.55	0.0%
Industrial good	72.27	72.27	0.0%
Commercial services	202.73	202.73	0.0%
Recycling services	0.11	0.11	0.0%
Waste collection	0.43	0.43	0.0%
Recycled material	0.18	0.18	0.0%
Prices			
Agricultural good	1.00	1.00	0.0%
Industrial good	1.00	1.00	0.0%
Commercial services	1.00	1.00	0.0%
Recycling services	1.00	1.00	0.0%
Waste collection	1.05	1.05	0.0%
Recycled material	1.00	1.00	0.0%
Utility private households	140.81	140.81	0.0%

Table 2 shows the changes in the demand for goods and services and the demand for recycling services and waste collection services. The demand for recycling services is not affected by the lower price for recycling services. This is an expected result because if households have the choice between collection and recycling services, the rational consumer will chose collection, which is free. So a lower price for recycling services will not affect the demand for these services as long as this price is larger than zero.

Since the expenditure of the government is kept constant at benchmark level within the model (see section 3.1), the government receives a small lump-sum transfer from the private households to compensate for the extra costs of the subsidy. Therefore the policy measure has no impact on the income of the households and consumption and utility are not affected.

1.6.2 Unit-based price scenario

The second scenario simulates the introduction of a unit-based pricing scheme for waste collection. Households now pay the equilibrium price for waste collection in which more generation of waste means more costs to private households for waste collection.

The results of this scenario are shown in Table 3. In the benchmark scenario the flat fee covered about 95% of the costs of waste collection, the government financed the remaining 5%. In this scenario the private households bear the full costs of waste collection, thus without intervention the expenditure of the government would increase. However, to avoid problems such as public expenditure incidence, we assume that government expenditure is constant. Thus, the government gives a positive lump-sum transfer to the private households (equal to 5% of the costs of waste collection). Private households now bear the full cost of waste collection, but because of the positive lump-sum transfer there is compensating change in the income they can spend on consumer goods. Therefore social welfare does not increase in this scenario.

Because recycling is now slightly less expensive then waste collection, households start to recycle more waste. This increases the demand for recycling services, which in turn causes the price of recycling services to rise. In the equilibrium solution the price of recycling services is again equal to the price of collection services. Recycling increases slightly (6.6%) and thus the price of recycled material decreases slightly (-3%). This changes the choice producers make between the use of recycled material versus virgin material, and the demand for recycled material increases with 5.5 percent.

The results suggest that introducing a unit-based price may give private households both an incentive to prevent generation of waste and to recycle waste. Since the costs of waste collection have increased, households have an incentive to prevent waste generation by substituting the waste intensive goods (agricultural and industrial goods) for the waste extensive good (commercial services). Households may also have an incentive to recycle more waste, since recycling in our specification is somewhat cheaper than waste collection.

***Table 3** Changes in the main variables for the 'Unit-based price scenario' as compared to the 'Benchmark case' (expenditures in Billion Euro) and the percentage change*

	Benchmark	Unit-based price	% change
Demand by private households			
Agricultural good	9.55	9.52	-0.3%
Industrial good	72.27	71.90	-0.5%
Commercial services	202.73	203.13	0.2%
Recycling services	0.11	0.12	6.6%
Waste collection	0.43	0.42	-2.4%
Recycled material	0.18	0.19	5.5%
Prices			
Agricultural good	1.00	1.00	0.0%
Industrial good	1.00	1.00	0.0%
Commercial services	1.00	1.00	0.0%
Recycling services	1.00	1.05	5.0%
Waste collection	1.05	1.05	0.0%
Recycled material	1.00	0.97	-3.0%
Utility private households	140.81	140.81	0.0%

Comparing the unit-based price scenario with the pro-recycling scenario suggests that introducing a unit-based price for waste collection is a more effective tool to promote recycling than subsidizing recycling. As long as waste collection is cheaper than recycling, which is the case in the flat fee-pricing scheme, households do not have a price incentive to start recycling and therefore may not recycle substantial amounts of waste.

1.6.3 Unit-based price plus pro-recycling scenario

In the third scenario both the variable costs for waste collection (scenario 2) and the subsidy for recycling services (scenario 1) are introduced simultaneously.

Now, households have a strong price incentive to increase recycling and demand less waste collection services (see Table 4). Consequently, demand for recycling services increases rapidly, and the quantity of waste collection services decreases by more than 30%.

Since households recycle more waste, the price of recycled material declines and industries start to use more of the relatively cheap recycled material. Thus the production costs decline slightly. The effects, however, are so small that the price of consumption goods hardly changes and consumption is therefore not affected by these indirect effects of recycling.

Scenario one and scenario three show the impacts of policies aimed at promoting recycling under different pricing schemes. Under the flat fee-pricing scheme, promoting recycling is not effective. Since waste collection has a marginal price of zero, subsidizing recycling can only have effect if the subsidy equals the total costs of recycling. In scenario three however, the amount of waste generated goes down. More waste is recycled and less waste is collected, incinerated, or landfilled. Comparing these scenarios shows that in our simplified economy the flat fee for waste collection causes a serious market distortion. Because of the flat fee pricing system, the price of recycling has no impact on the behavior of households and thus households are not responsive to recycling subsidies.

Table 4 *Changes in the main variables for the 'Unit-based price and pro-recycling scenario' as compared to the 'Benchmark case' (expenditures in Billion Euro) and the percentage change.*

	Benchmark	Pro-recycling & unit-based price	% change
Demand by private households			
Agricultural good	9.55	9.53	-0.2%
Industrial good	72.27	71.94	-0.5%
Commercial services	202.73	203.06	0.2%
Recycling services	0.11	0.24	121.1%
Waste collection	0.43	0.29	-32.4 %
Recycled material	0.18	0.39	116.7%
Prices			
Agricultural good	1.00	1.00	0.0%
Industrial good	1.00	1.00	0.0%
Commercial services	1.00	1.00	0.0%
Recycling services	1.00	1.05	5.0%
Waste collection	1.05	1.05	0.0%
Recycled material	1.00	0.68	-32.0%
Utility private households	140.81	140.91	0.1%

DISCUSSION

We have demonstrated how an applied general equilibrium model simulating the solid waste market and incorporating market distortions can be built. Since one of the characteristics of the waste market is the flat fee-pricing scheme for waste collection, it is important to realize that the actual price for waste collection that the households perceive is equal to zero. Special attention, therefore, has been given to modeling goods with a zero price. Such a market distortion has strong effects on the results of the model. This was shown in the application of the model in a numerical example using 1996 data for the Netherlands.

By modeling the complete product chain we can investigate not only the direct effects of unit-based pricing and recycling subsidies, but also the indirect effects through the prices of recycled material. One should keep in mind, however, that unit-based pricing may increase illegal dumping of waste (cf. Fullerton and Kinnaman, 1996). We abstracted from the costs associated with dumping as empirical studies suggest that this is a minor problem; in principle, social costs associated with illegal dumping can and should be accounted for in the unit-based price of waste collection. Furthermore, our approach abstracts from any impact of environmental awareness or other links between waste generation and welfare. These links are complex and virtually impossible to specify in an applied model such as ours.

Looking at the scenario where the unit-based pricing is combined with the recycling subsidy, we observe that due to a price decrease of recycled material, producers start to use far more recycled material. The decrease of production costs is, however, only slight and therefore we do not see any decrease in the prices of consumption goods and commercial services due to increased recycling efforts. Thus there are no significant indirect effects on the consumption pattern. This result implies that for this specific policy analysis, partial equilibrium analysis can provide relatively good approximations of the economic impacts of the policy.

The direct effects of unit-based pricing and recycling subsidies on household behavior are more pronounced. The modeling results suggest that a flat fee-pricing scheme for waste collection reduces the economic incentives to recycle. As long as a flat fee-pricing scheme is used, private households do not have economic incentives to reduce solid waste generation and for that reason will show little tendency to recycle. Even without taking into account the environmental gain of less waste it is effective to introduce unit-based pricing as it does not negatively affect the welfare of the society. Making recycling more attractive by subsiding recycling efforts will not result in substantially less solid waste generation as long as the price of recycling are greater than zero. Only if policies for promoting recycling are combined with a unit-based pricing scheme for waste collection will these policies be effective.

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[1] In a flat fee pricing system municipalities charge a fixed price for waste collection independent of the amount of waste actually generated.

[2] The waste hierarchy is a qualitative ranking of waste management or disposal practices. According to the waste hierarchy, the optimal waste treatment option is prevention followed by recycling and re-use, composting, incineration and landfilling, which is the least preferred way of dealing with waste.

[3] The formats are: 1) Computable general equilibrium format, 2) Negishi format, 3) Open economy format, and 4) Full format. Each of these formats has its strengths and weaknesses, for more information see Ginsburgh and Keyzer (1997). It must be stressed however that a format is just a way to solve a general equilibrium model. Each of the four formats will find the same equilibrium solution for the defined problem.

[4] A direct tax influences the income of the consumer but does not influence the consumption pattern.

[5] In the Negishi format, the equilibrium solution is found with the help of an iterative process. Given initial values for the Negishi-weights based on the income of a consumer, the model is solved and prices for each commodity are calculated as shadow prices. Subsequently, the budget constraint for each consumer is checked. If one or more consumers in the model spend more or less than their

income, the Negishi weight for that consumer is adjusted. The model is then solved again with the adjusted Negishi-weights. The process continues until the budget constraints of all consumers hold. See for more information Negishi (1972) or Ginsburgh and Keyzer (1997).

[6] See Ginsburgh and Keyzer (1997) for more details on this procedure.

[7] If the model were written in another format, the subsidy would not have to be made explicit in the welfare function. Note that optimal prices and quantities calculated by the model do not depend on the format chosen.

[8] Implicitly this means that part of the used material will accumulate in the stock of durable goods. Therefore at any given moment of time the material inflow need not be equal to the material outflow.

[9] A more extensive description of the model can be found in Bartelings (2003).

[10] The column of each producer sums to zero to ensure that the zero profit condition holds (value of input equals value of output). The column of each consumer sums to zero to ensure that the budget constraint holds. Each row must sum to zero to ensure that the market clears (total demand must equal total supply for each commodity).

[11] Waste percentage β is defined as kg waste present in kg good.

[12] Note that the households in this scenario pay the full price for waste collection, while in the benchmark case the flat fee covers only 95% of the costs (see section 3.1). The two scenarios are comparable, however, because we assume that government expenditure is constant. Therefore the government will be compensated for the differences between the flat fee revenues and the real costs of waste collection through a lump-sum transfer.

APPENDIX A SUBSTITUTION ELASTICITIES

Table A-1 Substitution elasticities for the production sectors

		Agri	Indu	Serv	Extr	Rm	Rec	Col	Incin	Land
σ^{kl}	substitution elasticity between capital and labor	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8
σ^{pi}	substitution elasticity between primary and intermediate inputs	0.0	0.0	0.0						
σ^{wm}	substitution elasticity between materials and other intermediate inputs	1.0	1.0	1.0						
σ^{rv}	substitution elasticity between recycled material and virgin material		∞							
σ^{pw}	substitution elasticity between primary factors and recycled waste					0.125				
σ^{li}	substitution elasticity between landfilling and incineration							0.2		

Note: 'Agri', 'Indu' and 'Serv' stands for respectively the three sectors of produced goods (Agriculture, Industry and Commercial services); 'Extr' stands for the extraction sector, 'Rm' indicates the production sector of recycled material; 'Rec' indicates the production sector of recycling services; 'Rw' stands for waste suitable for recycling; 'Col' is the collection sector, 'Incin' indicates the production sector of incineration services and 'Land' indicates the production sector of landfilling services.

APPENDIX B DEFINITION OF INDICES, PARAMETERS, AND VARIABLES

Indices

Label	Entries	Description
g	1,...,3	produced goods (agriculture, industry, commercial services)
i	1,2	consumers (households, government)
j	1,...,8	produced goods and services
z	1,...,10	commodities (including produced goods, services, capital and labor)

Parameters in model specification

Symbol	Description
α	Negishi weight
β	waste percentage
σ^{kl}	substitution elasticity between labor and capital
σ^{qm}	substitution elasticity between materials and other intermediate inputs
σ^{rv}	substitution elasticity between recycled material and virgin material
σ^{pr}	substitution elasticity between primary factors and recycled waste
σ^{il}	substitution elasticity between landfilling and incineration
ξ	subsidy wedge
F	flat fee for waste collection
LST	lump-sum transfer to keep income of government constant
$\overline{K}$	endowment of capital
$\overline{L}$	endowment of labor
P	price
p_t	price including subsidy
M^0	initial income

Variables in model specification

Symbol	Description
k	capital use
l	labor use
y	production
TW	total demand for waste collection services
TWF	total welfare
u	utility
W	total generation of waste
x	consumption
q	Use intermediate goods
X_r	total production of recyclable waste
M	Income

Appendix C Model specification

In the Negishi format total welfare is maximized given constraints of production sets and balance equations. The total welfare function depends on the weighted utility of the individual consumers:

$$TWF(\alpha) = \max \sum_i \alpha_i u_i(x_{i,g}) + \xi TW \tag{C.1}$$

$$x_{i,g} \geq 0,\ w_i \geq 0,\ r_i \geq 0 \ all\ i,\ y_j\ all\ j$$

Subject to production function of goods and collection services:

$$y_j \in Y_j \qquad for\ all\ j \tag{C.2}$$

$$Y_j = A_j \left[\min\{CES(k_j, l_j; \sigma^{kl}), CES(q_{j,g}, CES(q_{j,v}, q_{j,r}; \sigma^{w}); \sigma^{qm})\} \right] \tag{C.3}$$

$$for\ j = agri,\ indu,\ serv$$

$$Y_j = A_j \left[CES\{CES(k_j, l_j; \sigma^{kl}), X_r; \sigma^{pr}\} \right] \qquad for\ j = rm \tag{C.4}$$

$$Y_j = A_j \left[\min\{CES(k_j, l_j; \sigma^{kl}), CES(q_{j,is}, w_{j,ls}; \sigma^{il})\} \right] \tag{C.5}$$

$$for\ j = cs$$

$$Y_j = A_j \left[CES(k_j, l_j; \sigma^{kl}) \right] \qquad for\ j = extr, incin, land \tag{C.6}$$

And balance constraints:

$$\sum_i x_{i,j} + \sum_{j1} q_{j1,j} \leq y_j \qquad \perp p_j \qquad for\ all\ j\ except\ col \quad j1 \neq j \tag{C.7}$$

$$\sum_j k^j \leq \sum_i \overline{K_i} \qquad \perp p^k \tag{C.8}$$

$$\sum_j l^j \leq \sum_i \overline{L_i} \qquad \perp p^l \tag{C.9}$$

$$\sum_i x_{i,cs} \leq TW \qquad \perp p_{cs,sub} \tag{C.10}$$

$$TW \leq y_{cs} \qquad \perp p_{cs} \tag{C.11}$$

Equations waste generation :

$$w_i = \sum_g \beta_g x_{i,g} \tag{C.12}$$

$$x_i^r + x_i^w = w_i \tag{C.13}$$

Negishi weights are determined such that the budget constraint holds for every consumer:

$$\sum_g p_g x_{i,g} + p_{rs} x_{i,rs} + (p_{cs} - \xi) x_{i,cs} + F_i + \frac{M_i}{\sum_i M_i} LST = p^k \overline{K_i} + p^l \overline{L_i} \tag{C.14}$$

$$for\ i{=}cons$$

$$\sum_g p_g x_{i,g} + \xi TW = p^k \overline{K_i} + F_i + LST \qquad for\ i{=}gov \tag{C.15}$$

$$LST = M^0_{gov} - (p^k \overline{K_{gov}} + \sum_i F_i - \xi \sum_i x_{i,cs}) \tag{C.16}$$

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How extreme events can affect a seemingly stabilized population: a stochastic rendition of Ricker's model

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Abstract

Our paper computationally explores Ricker's predator satiation model with the objective of studying how the extinction dynamics of an animal species having a two-stage life-cycle is affected by a sudden spike in mortality due to an extraneous extreme event. Our simulation model has been designed and implemented using sockeye salmon population data based on a stochastic version of Ricker's model; with the shock size being reflected by a sudden reduction in the carrying capacity of the environment for this species. Our results show that even for a relatively marginal increase in the negative impact of an extreme event on the carrying capacity of the environment, a species with an otherwise stable population may be driven close to extinction

Key words: Ricker's model, extinction dynamics, extreme event, Monte Carlo simulation

Mathematics Subject Classification 2000: 65C05, 92D25

JEL classification: C15, Q59

Background and research objective

PVA approaches do not normally consider the risk of catastrophic extreme events under the pretext that no population size can be large enough to guarantee survival of a species in the event of a large-scale natural catastrophe. [1] Nevertheless, it is only very intuitive that some species are more "delicate" than others; and although not presently under any clearly observed threat, could become threatened with extinction very quickly if an *extreme event* was to occur even on a low-to-moderate scale. The term "extreme event" is preferred to "catastrophe" because catastrophe usually implies a natural event whereas; quite clearly; the chance of man-caused extreme events poses a much greater threat at present to a number of animal species as compared to any large-scale natural catastrophe.

An animal has a two-stage life cycle when; in the first stage, newborns become immature youths and in the second stage; the immature youths become mature adults. Therefore, in terms of the stage-specific approach, if Y_t denotes the number of immature young in stage t and A_t denotes the number of mature adults, then the number of adults in year t + 1 will be some proportion of the young, specifically those that survive to the next (reproductive) stage. Then the formal relationship between the number of mature adults in the next stage and the number of immature youths at present may be written as follows:

$$A_{t+1} = \lambda Y_t$$

Here λ is the survival probability, i.e. it is the probability of survival of a youth to maturity. The number of young next year will depend on the number of adults in t:

$$Y_{t+1} = f(A_t)$$

Here f describes the reproduction relation between mature adults and next year's young.

This is a straightforward system of simultaneous difference equations which may be analytically solved using a variation of the *cobwebbing approach*. [2] The solution process begins with an initial point (Y_1, A_1) and iteratively determines the next point (Y_2, A_2). If *predator satiation* is built into the process, then we simply end up with Ricker's model:

$$Y_{t+1} = \alpha A_t e^{-At/K}$$

Here α is the maximum reproduction rate (for an initial small population) and K is the population size at which the reproduction rate is approximately half its maximum [3]. Putting $\beta = {}^{1}/_{K}$ we can re-write Ricker's equation as follows:

$$Y_{t+1} = \alpha A_t e^{-\beta At}$$

It has been shown that if (Y_0, A_0) lies within the first of three possible ranges, (Y_n, A_n) approaches (0, 0) in successive years and the population becomes extinct. If (Y_0, A_0) lies within the third range then (Y_n, A_n) equilibrate to a steady-state value of (Y^*, A^*). Populations that begin with (Y_0, A_0) within the second range oscillate between $(Y^*, 0)$ and $(0, A^*)$. Such alternating behavior indicates one of the year classes, or cohorts, become extinct while the other persists i.e. adult breeding stock appear only every other year. Thus the model reveals that three quite different results occur depending initially only on the starting sizes of the population and its distribution among the two stages. [4]

We use the same basic model in our research but instead of analytically solving the system of difference equations, we use the same to simulate the population dynamics as a stochastic process implemented on an MS-Excel spreadsheet. Rather than using a closed-form equation like Ricker's model to represent the functional relationship between Y_{t+1} and A_t, we use a Monte Carlo method to simulate the stage-transition process within Ricker's framework; introducing a massive perturbation with a very small probability in order to emulate a catastrophic event.[5]

Conceptual framework

We have a formulated a stochastic population growth model with an inbuilt capacity to generate an extreme event based on a theoretical probability distribution. The non-stochastic part of the model corresponds to Ricker's relationship between Y_{t+1} and A_t. The stochastic part has to do with whether or not an extreme event occurs at a particular time point. The gamma distribution has been chosen to make the probability distribution for the extreme event a skewed one as it is likely to be in reality. Instead of analytically solving the system of simultaneous difference equations iteratively in some variation of the cobwebbing method, we have used them in a spreadsheet model to simulate the population growth over a span of ten time periods.

We apply a computational methodology whereby the initial number of immature young is hypothesized to either attain the expected number predicted by Ricker's model or drastically fall below that number at the end of every stage, depending on whether an extraneous extreme event does not occur or actually occurs. The mortalities as a result of an extreme event at any time point is expressed as a percentage of the pristine population size for a clearer comparative view.

Model building

Among various faunal species, the population dynamics of the sockeye salmon (*oncorhynchus nerka*) has been most extensively studied using Rickert's model. Salmon are unique in that they breed in particular fresh water systems before they die. Their offspring migrates to the ocean and upon reproductive maturity, they are guided by a hitherto unaccounted instinctive drive to swim back to the very same fresh waters where they were born to spawn their own offspring and perish. Salmon populations thus are very sensitive to habitat changes and human activities that have a negative impact on riparian ecosystems that serve as breeding grounds for salmon can adversely affect the peculiar life-cycle of the salmon. Many of the ancient salmon runs (notably those in California river systems) have now gone extinct and it is our hypothesis that an even seemingly stabilized population can be rapidly driven to extinction due to the effect of an extraneous (quite possibly man-made) extreme event with the capacity to cause mass mortality. The following table shows the four-year averages of the sockeye salmon population in the Skeena river system in British Columbia in the first half of the twentieth century.

Year	**Population (in thousands)**
1908	1,098
1912	740
1916	714
1920	615
1924	706
1928	510
1932	278
1936	448
1940	528
1944	639
1948	523

(***Source***: *http://www-rohan.sdsu.edu/~jmahaffy/courses/s00/math121/lectures/product_rule/product.html#Ricker's Model*)

A non-linear least squares best-fit to Ricker's model is obtained for the above set of data is obtained as follows:

Minimize $\varepsilon^2 = \sum_{t=1}^{n}[d_t - \{\alpha A_t e^{-\beta A_t}\}]^2$, where d_t is the actual population size in year t.

The necessary conditions to the above least squares best-fit problem is obtained as follows:

$$\partial(\varepsilon^2)/\partial\alpha = \partial(\varepsilon^2)/\partial\beta = 0; \text{ whereby we get } \alpha^* \approx 1.54 \text{ and } \beta^* \approx 7.8 \times 10^{-4}$$

Plugging these parameters into Ricker's model indeed yields a fairly good approximation of the salmon population stabilization in the Skeena river system in the first half of the previous century.

As the probability distribution of an extraneous extreme event is likely to be a highly skewed one, we have generated our random variables from the cumulative distribution function (*cdf*) of the gamma distribution rather than the normal distribution. The distribution boundaries are fixed by generating random integers in the range 1 to 100 and using these random integers to define the shape and scale parameters of the gamma distribution. The gamma distribution performs better than the normal distribution when the distribution to be matched is highly right-skewed; as is desired in our model. The combination of a large variance and a lower limit at zero makes fitting a normal distribution rather unsuitable in such cases.[6] The probability density function of the gamma distribution is given as follows:

$$f(x, a, b) = \{b^a\Gamma(a)\}^{-1} x^{a-1} e^{-x/b} \text{ for } x > 0$$

Here α > 0 is the shape parameter and β > 0 is the scale parameter of the gamma distribution. The cumulative distribution function may be expressed in terms of the *incomplete gamma function* as follows:

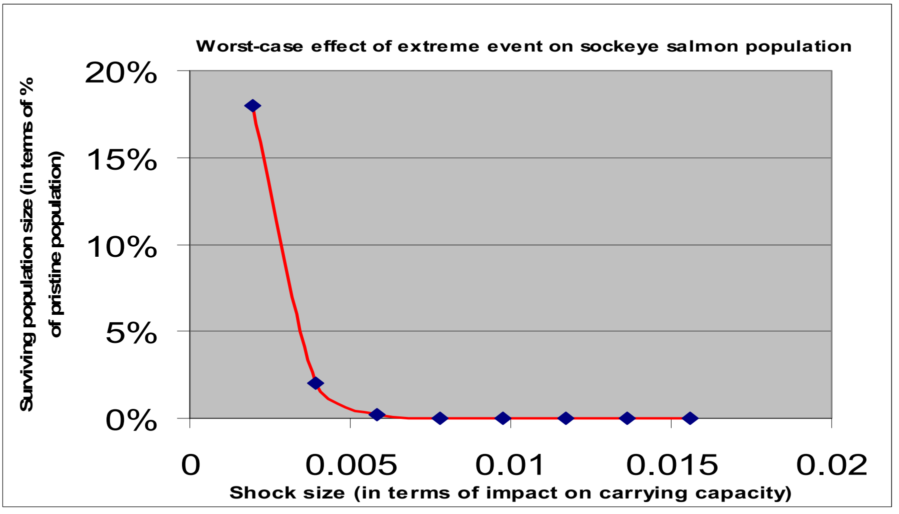
$$F(x, a, b) = \int_0^x f(u)du = \gamma(a, x/b)/\Gamma(a)$$

In our spreadsheet model, we have F (*R*, *R*/2, 2) as our *cdf* of the gamma distribution. Here *R* is an integer randomly sampled from the range 1 to 100. An interesting statistical result of having these values for x, α and β is that the cumulative gamma distribution value becomes equalized with the value [1 - χ^2 (*R*)] having *R* degrees of freedom, thus allowing χ^2 *goodness-of-fit* tests. [7]

Our model is specifically designed to simulate the extinction dynamics of sockeye salmon population using a stochastic version of Ricker's model; with the shock size being based on a sudden reduction in the parameter K i.e. the carrying capacity of the environment for this species. The model parameters are same as those of Ricker's model i.e. α and β (which is the reciprocal of K). We have kept α constant at all times at 1.54, which was the least squares best-fit value obtained for that parameter. We have kept a β of 0.00078 (i.e. the best-fit value) when no extreme event occurs and have varied the β between 0.00195 and 0.0156 (i.e. between 2.5 times to 20 times the best-fit value) for cases where an extreme event occurred. We have a third parameter *c* which is basically a 'switching constant' that determines whether an extreme event occurs or not. The switch is turned on triggering an extreme event when a random draw from a cumulative gamma distribution yields a value less than or equal to *c*. Using F (*R*, *R*/2, 2) as our *cdf* of the gamma distribution where *R* is a randomly drawn integer in the range (1, 100) means that the cumulative gamma function will randomly select from the approximate interval 0.518 ~ 0.683. By fixing the value of *c* at 0.5189 in our model we have effectively reduced the probability of occurrence of an extreme event to a miniscule magnitude relative to that of an extreme event not occurring. We have used the sockeye salmon population data from the table presented earlier For each level of the β parameter, we simulated the system and observed the maximum possible number of mortalities from an extreme event at that level of β. The results are reported below in Fig 1.

We made 100 independent simulation runs for each of the eight levels of β. The low probability of extreme event assigned in our study yielded a mean of 1.375 for the number of observed worst-case scenarios (i.e. situations of maximum mortality) with a standard deviation of approximately 0.92. The worst-case scenarios for our choice of parameters necessarily occur if the extreme event occurs in the first time point when the species population is at its maximum size. Our model shows that in worst-case scenarios, the size of surviving population after an extreme event that could seed the ultimate recovery of the species to pre-catastrophe numbers (staying within the broad framework of Ricker's model) drops from about 18% of the pristine population size for a shock size corresponding to 2.5 times the best-fit β; to only about 0.000005% of the pristine population size for a shock size corresponding to 20 times the best-fit β.

Fig. 1: Results obtained from the simulation model



Therefore, if the minimum required size of the surviving population is at least say 20% of the pristine population in order to survive and recover to pre-catastrophe numbers, the species could go extinct if an extreme event caused a little more than two-fold decrease in the environmental carrying capacity! Even if the minimum required size for recovery was relatively low at say around 2% of the pristine population, an extreme event that caused a five-fold decrease in the environmental carrying capacity could very easily force the species to the brink of extinction. An immediate course of future extension of our work would be allowing the fecundity parameter α to be affected by extreme events as is very likely in case of say a large-scale chemical contamination of an ecosystem due to a faulty industrial waste-treatment facility.

Conclusion

Our study has shown that even for a relatively marginal 2.5-fold decrease in the environmental carrying capacity due to an extreme event, a worst-case scenario could mean a mortality figure well above 80% of the pristine population. As a guide for future PVA studies we may suggest that one should not be deterred simply by the notion that extreme events are uncontrollable and hence outside the purview of computational modeling. Indeed the effect of an extreme event can almost always prove to be fatal for a species but nevertheless, as our study shows, there is ample scope and justification for future scientific enquiries into the relationship between survival probability of a species and the adverse impact of an extreme event on ecological sustainability.

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Low-income Farmers' Behavior toward Land Degradation: The Effects of Perceptions, Awareness, Attitude, and Land Use

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ABSTRACT

A structural equation modeling approach is used to examine the relationship between farmers' perceptions of land degradation and their self-reported environmental behavior. Influence of crop dependency, land tenure, and demographic pressure on perceptions and behavior is also explored. Results showed that perceived susceptibility and severity of land degradation strongly influences farmers' awareness of, and attitude toward environmental problems. Awareness of land resource depletion is likely to foster behavioral changes that will mitigate the problems. Greater financial and economic dependence on crop production causes greater perceived susceptibility, severity, benefits, and more positive attitude toward land degradation. Greater dependence on crops is associated with less perceived barriers to behavioral change. Direct ownership of the land is negatively related to perceived severity, and awareness, but positively associated with barriers and behavior.

Keywords: Structural; equation; perceptions; behavior; land degradation

JEL Classification: 62H, 62J

Mathematics Subject Classification 2000: 91B76, 91B82

1. INTRODUCTION

With one of the fastest growing populations in the western world, Haitians depend directly or indirectly on agriculture for their livelihood. Agricultural lands have been under enormous pressure to produce food, fiber, and firewood for a growing population (annual average growth rate of 1.67 percent [CIA World Fact Book, 2003]) for several decades. As the population increases, more and more marginal lands are brought into production to satisfy market demand for food. The intensive cultivation of those lands, most of which are unsuitable for agricultural production, results in severe soil loss due to erosion, landslides, and finally a complete change of landscape. This situation of declining soil fertility and loss of agricultural lands keeps the Haitian economy in a chronic state of underdevelopment and exposure to long-term climatic changes (Lundahl, 1996).

Farmers have partially responded to land resource depletion and the concomitant declining agricultural productivity by adopting indigenous techniques, such as contour trash barriers and contour bunds to retard the erosion process and improve soil fertility (White and Jickling, 1995). In the face of farmers' limited capacity and laxity to reduce the land degradation problem, government and international institutions have encouraged programs aimed at retarding the process and improving the environment. Results of such programs have been less than satisfying because of farmers' reluctance to change their environmental attitude and behavior.

Early studies (Saint-Dic, 1981) posit simplistic theories to explain farmers' lack of interest into changing behaviors and practices that may result in environmental improvement. They point out farmers' ignorance and conservatism in the face of change. Other studies have been less critical, arguing that farmers' environmental attitude and behavior are related to the multiple constraints they face (Jean-Pierre, 1984; Murray, 1979). Recent studies (White and Quinn, 1992; Bayard, 2000; Bannister, 2001) suggest that social, demographic, and economic factors are the primary determinants of farmers' environmental behavior.

Although several studies indicated that socio-economic factors may influence farmers' environmental behavior, little is understood about how their perceptions, beliefs and attitude toward the environment may result in behavioral change. Even more noticeable is that few attempts have been made to understand farmers' perceptions and awareness of the environmental problem, and the underlying socio-economic mechanism that may trigger the modification of poor farmers' attitudes toward the environment. Furthermore, analysis of the effect of farmers' economic and financial dependence on the land on their perceptions, attitude, and behavior has not been conducted. This study investigates limited, resource Haitian farmers' knowledge and perceptions of land degradation.

2. PREVIOUS RESEARCH

A number of studies worldwide have evaluated determinants of farmers' environmental behavior. Some researchers (Gould et al., 1989; Bultena and Hoiberg, 1983; Norris and Batie, 1987; Burton et al., 1999; Featherstone and Goodwin, 1993) showed that socio-economic factors, including age and education, significantly shape farmers' environmental attitude and behavior. Financial factors, such as short-term profitability and long-term asset value are important in explaining farmers' attitudes (McConnell, 1983; Barbier, 1990; Napier, 1991; Sain and Barreto, 1996).

Economic factors including farm income, off-farm revenue, and risk aversion are found to influence soil conservation adoption decisions (Shields et al., 1993; Lasley et al., 1990). Higher income is likely to increase adoption of land management technologies (Lynne and Rola, 1988; Shields et al., 1993; Luzar and Diagne, 1999) while off-farm activities may inhibit this decision (Lasley et al., 1990). Institutional factors such as land tenure arrangements, membership in environmental and local organizations, and technical assistance also play a significant role in adoption of soil conservation techniques (Francis, 1986; Lee and Stewart, 1983; Sureshwaran et al., 1996; Burton et al., 1999; Soule et al., 2000). Other studies indicated that farm size, length and degree of slope, and soil erodibility are the primary determinants of conservation behavior (Rahm and Huffman, 1984; Barbier, 1990; Huszar and Cochrane, 1990; Sureshwaran et al., 1996).

Results of the studies reported above were rather inconclusive and/or conflicting. Studies by Ervin and Ervin (1982) and Rogers (1995) suggest that perceptions and awareness of environmental problems are likely to influence individuals' behavior. According to those researchers, farmers perceive land degradation problems before they decide whether to adopt or not to adopt a conservation measure (Ervin and Ervin, 1982; Gould et al., 1989; Traoré et al., 1998; Bultena and Hoiberg, 1983; Napier and Brown, 1993). They found farmers' behavior to be significantly related to their levels of perception of environmental problems.

Farm level studies, recognizing the importance of perceptions, emphasize attitude as the prime determinant of environmental behavior. In the United States, Lynne and Rola (1988), Luzar and Diagne (1999), and Bourke and Luloff (1994) reported that farmers' environmental attitude was significantly related to participation in environmental programs.

In Europe, Willock et al. (1999a, 1999b), explored the correlations between farmers' attitude and objectives and environmental behaviors. Their results showed significant positive correlations between environmentally oriented behaviors and attitudes. Behaviors were also significantly correlated to farmers' sustainability objectives. Their results also indicated that attitudes and farmers' goals have significant positive correlations.

Pouta and Rekola (2001) found significant relationships between attitude and behavioral intentions for abatement of forest regeneration in Finland. Attitudes are in turn influenced by beliefs held about the outcomes of behavior. In Bangkok, Daniere and Takahashi (1999) found that attitude and value variables are highly significant predictors of environmental behavior. Other studies (Carr and Tait, 1991; Hines et al., 1990; Kantola et al., 1982) observed significant correlations between attitude and behavior. In one study, Kantola et al. (1982) found a positive effect of attitude and social norms on intentions to conserve water. In another set of results, Kantola et al. (1983) argued neither increasing severity of water pollution, nor efficacy influenced farmers' behavioral intentions in promoting water conservation.

In a study conducted in Austria, Vogel (1996) found that farmers' behavior is strongly related to their attitudes toward environmental problems in general, and other issues related to their spheres of activity. In short, attitude may be a significant determinant of environmental behavior. However, in some cases, attitude may be a weak determinant of environmental behavior. In this study, the structure of farmers' environmental attitude and behavior is analyzed, and the influence of the level of economic dependence on crop production, land tenure, and demographic pressure on farmers' perceptions, awareness, attitude, and behavior toward land degradation is also examined.

3. CONCEPTUAL FRAMEWORK

The theory of reasoned action (TRA) developed by Ajzen and Fishbein (1977), and its extended version, the theory of planned behavior (TPB) (Ajzen and Madden, 1986; Ajzen, 1991) connect attitude and behavior in a logical framework. According to the TRA, intention is the most important determinant of action.

Behavioral intention refers to a person's intention to perform various behaviors. Intention in the TRA is specified as a function of attitude and subjective norms. Attitude refers to the degree to which a person has a favorable or unfavorable evaluation of an object. Attitude is influenced by a person's beliefs that performing a behavior will lead to a desired outcome.

In this study, we use the TRA framework and results from the empirical studies reviewed above to examine the attitude-behavior relationship among low-income farmers in Haiti. The study is based on the premise that farmers, who primarily depend on the land for their livelihood, will develop an attachment to the land and a strong concern for the environment, and perceive themselves as environmentally responsible. However, the level of dependency on crop production for their survival may complicate the development of a positive attitude toward the environment. As indicated by Vogel (1996), farmers living in difficult conditions may possess a higher perception of environmental problems. All categories of individuals may be concerned about the degradation of the environment. Differences in attitudes and behavior may result from different opportunities, constraints, and perceptions.

The importance of the land in the Haitian context makes its improvement desirable. To achieve a certain level of improvement in the environment, not only are technological changes important, but also changes in attitudes and behaviors of those involved in agriculture are necessary. Nevertheless, farmers' beliefs may enhance or inhibit confidence in their ability to perform a particular task.

We assess farmers' perceptions of susceptibility, severity of land degradation, benefits of environmental improvement, and social and technological barriers to change. Influence of those psychological constructs on environmental awareness and attitude is examined. Susceptibility refers to one's perception of being affected by environmental degradation. Perceived seriousness of land degradation is the consciousness of the problem. Benefits refer to the perceived economic and social benefits of environmental improvement; and barriers are the potential negative consequences or factors that may hinder positive actions.

A structural model examining the nature of farmers' environmental behavior is developed in figure 1. Typically, the model postulates that perceived susceptibility to land degradation, perceived seriousness of the problem, perceived benefit of environmental improvement, and perceived barrier to behavioral change influence farmers' awareness of, and attitude toward environmental problems. Attitude and awareness, in turn, affect self-reported behavior, suggesting a mediating role between the perception factors and behavior. Finally, the model examines the effect of crop dependency, land tenure, and demographic pressure on the perceptual variables, awareness, attitude, and behavior. To determine the predictors of farmers' environmental behavior, we propose the following hypotheses:

Hypothesis 1: Farmers' perceptions about environmental degradation will influence their awareness of the problem and their attitude toward adopting behaviors that will mitigate the problem.

Hypothesis 1a: The greater the perceived susceptibility to environmental degradation, the more farmers become aware, and the more positive attitude they will develop toward the environment.

Hypothesis 1b: The perception of seriousness of land degradation enhances farmers' level of awareness, and their attitude toward the environment.

Hypothesis 1c: The greater perception of physical, environmental, financial, social, and economic benefits, the greater awareness, and the more positive attitude will farmers have toward environmental improvement.

Hypothesis 1d: The greater the perception of barriers to actions towards reducing environmental degradation, the weaker farmers' awareness of and attitude toward the environmental problem.

An important aspect of the study is the investigation of the relationship between attitude and behavior. It is assumed that attitude is directly related to behavior, and perceptions of land degradation, along with the level of dependence on the land, will strengthen this relationship. Therefore, given the importance of the land in all aspects of the farmer's lives, we examine the following hypotheses:

Hypothesis 2: Farmers' awareness of, and attitude toward the environment will positively influence their behavior.

In the Haitian peasant economy, land is an important factor as most Haitians are culturally tied to this natural resource, and economically and financially dependent on it for their survival.

Hypothesis 3a: Farmers' economic and financial dependency on crop production positively influences their perceptions of the susceptibility, seriousness of environmental degradation, and perceived benefits of environmental improvement, and negatively influences their perception of the barriers to behavioral change.

Hypothesis 3b: Farmers' economic and financial dependency on crops positively influences their awareness of land degradation, and their attitudes and behavior toward the problem.

Hypothesis 3c: Farmers who directly own the land they operate are less likely to be concerned about environmental degradation, but more likely to change their behavior toward the problem.

4. METHOD

4.1. Study area

The study was conducted in two regions in Haiti: the South and the Southeast. These regions were selected because of the observed levels of land degradation, because of farmers' exposure to soil and water conservation projects, and previous participation by villagers in soil management practices that are likely to raise their awareness of the problems. In the South, field surveys were conducted in Gaita and Bannate, two villages within the community of Camp-Perrin, where farmers have been exposed to soil conservation projects conducted by development agencies in collaboration with local organizations.

Cultivated lands in the area of Gaita and Bannate are on elevations of 100 to 300 meters above sea level. The average annual rainfall is usually between 1,500 and 2,000 millimeters. Rainfall in the region, occurring from February to November, is highly variable with a bimodal seasonal distribution. The first rainy season lasts from February to May, and the second from July to November. This regional rainfall distribution pattern allows two planting seasons per year.

In the Southeastern region, data were collected in three locations including Cap-Rouge, Cayes-Jacmel, and Marigot. For the last three decades, several development projects conducted activities related to coffee production, planting and grafting of fruit trees, and diffusion of other soil management practices (Macroscope, 1997). The research site in this region varies in elevation from 200 to 500 meters. The average annual rainfall in this region varies from 1,000 to 1,500 millimeters.

In both regions, the farming systems are characterized by the production of various annual crops. The major food crops, which include corn, sorghum, beans, cassava, yams, and sweet potato, occupied steeply sloping lands that are classified as more appropriate for forest uses. The slopes of cultivated plots in the regions can be more than 60 percent. Some farmers produce perennial crops, such as coffee and cocoa, that provide them with substantial cash earnings. Hillside farming in these regions is especially intensive. The farming system is dominated by a rotation of maize/beans and maize/sorghum/pigeon peas with minimal levels of purchased inputs and intensive use of family labor. Both regions suffer severe soil erosion problems due to the agro-climatic conditions on one hand, and the lack of soil protection on the other. Coupled with a short fallow period of one to two years, the degradation of the soil causes the decline of the fertility level, and consequently reduces crop yields. For several decades, the Haitian government and development agencies have promoted soil conservation projects in an attempt to reduce land degradation. Nevertheless, farmers have failed to extensively adopt the techniques promoted which include rock walls, contour canal, and hedgerows.

Unlike many basically subsistence-oriented farmers, agricultural producers in the study areas are highly integrated into a market economy. While the farmers' primary goal is food self-sufficiency, a large proportion of agricultural surplus is sold on the market. Production and market risks often threaten the farmers' well being, and force them to develop survival strategies, including agricultural diversification, off-farm activities, and selective migration.

4.2. Data collection

Interviews with farmers in both regions were carried out in two successive phases. In the first phase, a random sample of 240 farm operators from the Southern area, and 360 from the Southeastern region was taken, and data were collected through personal interviews between July and August 2000. A second set of interviews was carried out between January and March 2002 to collect additional information not included in the previous survey. Upon completion of the field survey, six interviews were discarded for incomplete information. Thus, 594 observations remained for the final analysis, constituting 99 percent usable questionnaires.

4.3. Survey instrument

Two survey instruments were developed for the purpose of the interviews. The first questionnaire which was used in the first phase gathered information on farm structure and operators' characteristics. Information was also collected on farmers' awareness of, and attitude toward soil erosion and land degradation, their perception of susceptibility, their appraised severity of environmental degradation, the benefits of conservation and the perceived barriers to change. The survey instrument used in the second phase gathered information on farmers' goals, opinions on policy formation, and environmental behavior. Questions related to perceptions, awareness, attitude and behavior were measured on a five-point scale in terms of how strongly the respondents felt about a set of statements. The responses were weighted 1 to 5 with lower values indicating greater agreement (see Bayard, 2003 for more details). Prior to the field survey, pretests of the questionnaires were conducted with a group of farmers in both regions.

4.4. Variable measurement and model estimation

The variables representing perceived susceptibility, perceived severity, benefits, barriers, awareness and attitude were recorded by asking farmers to scale a set of questions that expressed their beliefs about each issue. These questions evaluated farmers' attitudes toward ecological, social, and economic problems related to environmental degradation. Behavior was recorded by asking farmers a set of questions that indicated actions they have taken, or intend to take to improve the environment.

Exploratory factor analyses were conducted on the data to extract the items that measure each construct using the SAS software system (Hatcher, 1994). Each set of items defining a particular construct was submitted separately to factor analysis. The scree test was used to determine the number of meaningful factors retained for interpretation, and an orthogonal varimax rotation was used. A reliability assessment (Cronbach's alpha) was used to check for internal consistency of each factor.

Structural equation modeling was used to test the hypotheses developed in the study using Maximum Likelihood Estimation procedures. We used a system of simultaneous equations to accommodate the structural model developed in Figure 1. Equations 1 and 2 depict relationships between observed indicators and latent variables for exogenous and endogenous variables, respectively.

$$x = \Lambda_x \xi + \delta \qquad (1)$$

$$y = \Lambda_y \eta + \varepsilon \qquad (2)$$

where x and y are the $q \times 1$ and $p \times 1$ vectors of observed variables; Λ_x is the $q \times m$ matrix of regression coefficients of x on ξ; Λ_y is the $p \times m$ matrix of coefficients of the regression of y on η; ξ is the $n \times 1$ matrix of random vector of the latent exogenous variables representing perceived susceptibility (ξ_1), perceived seriousness (ξ_2), perceived benefits (ξ_3), and perceived barriers (ξ_4); η is the $m \times 1$ matrix of random vector of latent endogenous variables representing awareness (η_1), attitude (η_2), and behavior (η_3); and δ and ε are $q \times 1$ and $p \times 1$ vectors of measurement errors in x and y, respectively. The following equations were used to estimate the hypothesized causal relationships:

$$\eta_1 = \gamma_{11}\xi_1 + \gamma_{12}\xi_2 + \gamma_{13}\xi_3 + \gamma_{14}\xi_4 + \gamma_{15}CD + \gamma_{16}LT + \gamma_{17}DP + \zeta_1 \quad (3)$$

$$\eta_2 = \gamma_{21}\xi_1 + \gamma_{22}\xi_2 + \gamma_{23}\xi_3 + \gamma_{24}\xi_4 + \gamma_{25}CD + \gamma_{26}LT + \gamma_{27}DP + \zeta_2 \quad (4)$$

$$\eta_3 = \beta_{31}\eta_1 + \beta_{32}\eta_2 + \beta_{33}CD + \beta_{34}LT + \beta_{35}DP + \zeta_3 \quad (5)$$

$$\xi_1 = \alpha_{11}CD + \alpha_{12}LT + \alpha_{13}DP + \zeta_4 \quad (6)$$

$$\xi_2 = \alpha_{21}CD + \alpha_{22}LT + \alpha_{23}DP + \zeta_5 \quad (7)$$

$$\xi_3 = \alpha_{31}CD + \alpha_{32}LT + \alpha_{33}DP + \zeta_6 \quad (8)$$

$$\xi_4 = \alpha_{41}CD + \alpha_{42}LT + \alpha_{43}DP + \zeta_7 \quad (9)$$

where CD, LT, and DP are manifest exogenous variables representing crop dependency, land tenure, and demographic pressure, respectively; β is the $m \times m$ matrix of coefficients relating the latent endogenous variables, γ is the $m \times n$ matrix of the ξ-variables in the structural relationship between η and ξ, and α is the coefficient for the effects of land dependency on ξ. It is assumed that $E(\zeta) = 0$, $E(\xi\zeta\ ) = 0$, $E(\zeta\zeta\ ) = \Psi$. Crop dependency is calculated as the share of crop earnings in total household income. Land tenure represents an index of the ration of the size of land directly owned by farm size. Demographic pressure is measured as the number of people dependent on the household divided by the size of the farm. The analyses were conducted using Lisrel 8.54 (du Toit and du Toit, 2001; Jöreskog and Sörbom, 2001). In the line of Anderson and Gerbing's (1988) two-stage approach, the measurement model was first evaluated, then, the structural paths were estimated. Model fit was assessed using the normed-fit and goodness-of-fit indices, as well as the comparative fit index.

5. EMPIRICAL RESULTS

5.1. Demographic profile of the respondents

Of the 594 respondents, male farmers made up 85 percent of the total sample, whereas females represented 15 percent. The average age of surveyed farmers was 48 years, ranging from 22 to 86 years. Approximately half of the respondents were between 35 and 50 years of age.

The survey reveals that 63 percent of the respondents had some primary school level of education. Thirty-two percent had no formal education, and only 5 percent of the respondents had continued their studies beyond primary level. Fifty-nine percent of surveyed farmers declared that they pursued some type of educational training in soil conservation. Sixty-seven percent of the farmers interviewed are members of local groups, while 33 percent do not belong to any local organizations. Twenty-one percent of those who belong to a group had assumed, at one time or the other, a leadership position.

The average size of households in the study areas is 6.33 ranging from one to 18 individuals. Given the small size of plots operated by farmers, the average density of a farm household is about seven people per hectare (ha). Forty-seven percent of the farm households had a population density of less than five people. Households with five to 10 and more than 10 people per hectare made up 35 percent and 18 percent of the sample, respectively. These reveal that a relatively high level of pressure is being exerted on household resources in these regions.

5.2. Farm structure

The typical Haitian farm enterprise is composed of numerous plots for which land tenure arrangements vary. The various plots that a farm household may manage are often located in different agro-ecological areas, allowing farmers to grow a variety of crops including cereals, vegetables, and fruits. This land distribution pattern allows farmers to spread food availability over the entire year; therefore, limiting the risk of starvation.

Farmers in the study areas cultivate an average of 5.23 plots ranging from one to 18. The majority of the farms (52 percent) are composed of five to 10 plots. Despite the relatively large number of plots cultivated, the size of a farm in the study areas is small averaging 1.48 hectares. Forty-seven percent of the farms are between one and three ha. The plots are cultivated under various land tenure arrangements including purchase, inheritance, share-cropping, renting, and temporary use of family plots. Nevertheless, direct ownership via purchase is the most important source of access to land in the research sites. Direct ownership represents 56 percent of all cultivated land in the research areas.

Farm household income is generated from a multitude of activities in which family members are involved. Agriculture is the primary source of earnings for the numerous people in the study areas. However, household members are involved in a number of off-farm activities, including forest exploitation, petty trade, and off-farm jobs that bring additional income in the household on a regular basis. The estimated per capita income per year for a household is on average 1,871 gourdes (1gourde = U.S.$ 0.05) ranging from 19 to 15,790 gourdes. Most farmers (61 percent) have an annual per capita income of over 1,000 gourdes. Revenues from selling crops grown on the farm represent 74 percent of the household earnings. Share of animal production represents 18 percent, whereas off-farm earnings make up 8 percent of total family income. Although farmers are involved in a multitude of economic activities that bring revenue to the household, they are highly dependent on the land for their survival.

5.3. Farmers' perceptions of environmental problems

Numerous indicators were used to assess farmers' perceptions of environmental problems. The indicators describe their susceptibility towards, and their perceived severity of the degradation of the environment, their perceived benefits of environmental improvement, and the self-imposed barriers that may hinder actions to improve the environment. Table 1 shows the factor analysis results for the items defining the perception factors.

Perceived susceptibility suggests that respondents do not perceive themselves as being affected by erosion problems because they have taken conservation measures to prevent them. Four items, with loading greater than 0.40, were retained. Cronbach's alpha coefficient for the susceptibility factor was 0.80 indicating a relatively good internal consistency. The seriousness of the problem shows how land degradation may affect the farmer, the community, and country's welfare. In terms of perceived seriousness of environmental problems, five items with loadings greater than 0.40 were retained by the factor analysis. The items mainly reflect the perception of the damages caused by erosion at the farm level. The reliability coefficient for this factor is estimated at 0.83.

Five indicators define farmers' perceived benefits of environmental improvements (Table 1). The items reflect farmers' decisions to adopt conservation measures that are in themselves beneficial and can improve the land because they result in positive outcomes. Reliability coefficient for the benefit factor was 0.89.

Factor analysis suggests seven items that measure perceived barriers to environmental improvement. The items deal with various types of issues including social and physical barriers. These items have high loadings on the perceived barrier factor. The coefficient alpha for barrier factor was 0.84 suggesting that the scale measuring the items is reliable.

5.4. Farmers' environmental awareness, attitude, and behavior

Awareness of land degradation is an important step toward undertaking remedial actions. Farmers must be aware of erosion problems, of their consequences, and measures that halt environmental degradation before they can engage in any conservation behavior. Awareness of the problem should generate greater willingness to change agricultural practices that will engender environmental improvement. To assess farmers' awareness of environmental degradation in Haiti, farmers were asked to indicate their agreement with the knowledge of various environmental issues. Table 2 shows the items used to define the awareness factor.

Seven items, with loadings greater than 0.40, define the awareness factor. The scales of these variables demonstrate a high level of reliability with an estimate of 0.78. Items that load heavily on the awareness factor cover a wide range of issues. The items of the awareness factor are related to knowledge of the existence of erosion problems at the national, local, and farm levels. Two of the variables capture the impact of erosion on soil nutrients and crop yields. The two final items demonstrate the effect of farming practices and tree cutting on the process of erosion. It appears from these results that items directly related to activities on the farm have the highest loadings on the awareness factor.

The items dealing with farmers' environmental attitudes reflect the global effects of soil erosion and the individuals' responsibility in the process. The attitudinal factor consists of seven items with loadings greater than 0.40. The attitudinal factor seems to have a relatively good reliability with an alpha coefficient of 0.83.

To assess individuals' environmental behavior, a number of statements were included in the questionnaire that elicited the reasons why farmers adopted measures to retard environmental degradation. Table 2 reports the selected items that load on the behavioral construct. All the items, but two had loadings greater than 0.70. The behavior factor had a coefficient of reliability of 0.72, indicating a reasonable internal consistency.

5.5. Determinants of environmental perceptions, awareness, attitude, and behavior

We used a structural model in Lisrel (Jöreskog and Sörbom, 2001) to examine the determinants of the structure of farmers' environmental behavior. The maximum likelihood estimation results are reported in Table 3. Chi-square statistics is often used to assess the fit of structural equation models. However, sample sizes tend to inflate this statistic (Vaske and Kobrin, 2001; Byrne, 1998). Consequently, it is suggested that the chi-square should be evaluated in relation to the model's degree of freedom (Marsh and Hocevar, 1985; Vaske and Kobrin, 2001; Jöreskog and Sörbom, 2001). A χ^2/df ratio along with other indicators, such as the Normed Fit Index (NFI), the Goodness of Fit Index (GFI), and Root Mean Square Error Approximation (RMSEA) are suggested as measures of fit. A χ^2/df ratio between 2:1 and 5:1 indicates an acceptable fit (Marsh and Hocevar, 1985). Byrne (1998) and Bentler (1990) also suggested the use of the Comparative Fit Index (CFI) to assess the fit of a structural model. A value of 0.90 and above for NFI, CFI, and GFI is considered good fit.

The overall fit of the model was assessed using the NFI, GFI, and CFI. Values for NFI, GFI, and CFI were 0.95, 0.92, and 0.96, respectively. Hence, the indices suggest that the overall fit of the model is acceptable.

Among the perceptual factors, perceived susceptibility and severity of environmental degradation were found to influence farmers' awareness of the problem. The standardized path coefficients for perceived susceptibility and severity factors are 0.24 (t= 2.36) and 0.35 (t=5.24), respectively. The coefficients were significant at 5 percent level supporting the hypotheses that perceived susceptibility and severity of environmental degradation were positively related to awareness. This result is interesting in the sense that soil erosion is a widespread phenomenon throughout Haiti. Several indicators dealing with the effects of land degradation on the agricultural production process define the severity factor. The degradation of the environment impacts individuals' lives at different levels. Farmers on the hillside continuously observe the physical aspect of the erosion process and experience its effects on agricultural production. Given their direct experience with the problem, farmers have developed a good understanding of the negative effects of erosion on soil nutrients, crop yields, and of other environmental damages. Therefore, the more severe they perceived the damage and the more susceptible they feel about it, the more they become aware of the extent of environmental degradation in Haiti. It is evident that the impact of soil erosion would be greater in the more eroded areas than it is in less eroded ones. Hence, farmers in the most affected areas may be more aware of erosion problems than those in less eroded areas.

Results of the model also support the hypotheses that perceived susceptibility and perceived severity influence farmers' attitude toward the environment. The perception of susceptibility was positively related to attitude toward the environment. As shown in Table 3, the standardized coefficient for the susceptibility factor was 0.44 ($t = 4.65$), suggesting a significant influence on attitude. The results suggest that farmers who feel susceptible to land degradation are more likely to develop a positive attitude toward the environment. The perceived severity factor was positively related to the attitude variable. The standardized path coefficient for this variable was 0.22, with a t-value of 4.18. Increasing perceived severity of environmental degradation tends to promote a positive attitude of Haitian farmers toward the environment.

Results of the model show that awareness of environmental degradation has a significant causal effect on farmers' environmental behavior. As hypothesized, greater awareness of the degradation of the environment leads farmers to take measures to reduce the problem. The standardized coefficient of awareness on behavior is 0.30 ($t = 4.32$). It appears that perceived susceptibility and severity of the degradation of the environment are the most important factors influencing a change of environmental behavior. The influence of these variables on behavior is mediated by the awareness factor.

One of the hypotheses in this study is that farmers' dependence on the land for crop production will have a significant effect on their perceptions, awareness, attitude, and behavior. As mentioned earlier in this study, Haitian farmers depend heavily on the land for their survival. Not only that the staples produced are primarily used for home consumption, but also, a fraction of the production is sold on the market, allowing farmers to participate in commercial transactions. The results support the hypothesis that crop dependency influences farmers' perceptions of environmental problems. Dependency on crop production significantly affects perceptions of susceptibility, severity of land depletion, benefit of and barriers to environmental improvement. The coefficients of crop dependency on susceptibility, severity, benefit, and barriers are 0.36 ($t = 8.25$), 0.20 ($t= 4.32$), 0.30 ($t= 6.97$), and -0.16 ($t=-3.61$), respectively. Attitude toward environmental degradation is also affected by the level of economic and financial dependence on crops produced. The results show that attitude toward land degradation increases with the level of dependence on crops. The standardized coefficient of land dependency on attitude is 0.10 ($t = 2.32$). This result supports the hypothesis that greater dependence on crop engenders a more positive attitude toward environmental problems among farmers. In general, Haitians living in rural areas depend on the land for their survival. Some farmers have off-farm jobs that compete with farming activities. Others have no other alternative than farming. Therefore, their only means to make a living is to keep exploiting the limited land they operate.

Land tenure arrangements are often considered important factors in land management decisions. Results of this study show that ownership of the land is negatively related to awareness of environmental degradation ($\alpha_{16}= -0.13$), and perceived severity of the problems ($\alpha_{22}= -0.13$), but positively associated with barriers ($\alpha_{32}=0.12$) and behavior ($\beta_{34}= 0.09$).

6. DISCUSSIONS AND CONCLUSIONS

This study focuses on the influence of perceptual variables on low-income farmers' awareness of environmental degradation, their attitude and behavior toward the problem, and the role played by land use in shaping their perceptions. The present results suggest that Haitian farmers are aware of the degradation of the environment in their country. Their strong agreement with statements underlying awareness showed that their understanding and knowledge of the problem are lucid.

An important finding of this research is the effect of farmers' perceived severity of land degradation on their awareness. Results of this research also show that perception of the severity of land degradation is influential in determining farmers' attitude toward the environment. Perception of the severity of environmental degradation appears to play a significant role in raising farmers' awareness and shaping their attitude. This result appears to corroborate findings from previous research (Gould et al., 1989, Bultena and Hoiberg, 1983). Land degradation affects people's lives at various levels. Farmers who are dealing with this problem on a daily basis may have observed the phenomenon of land degradation on their farms and surrounding areas. In fact, the items defining the perceived severity of land degradation deal with issues directly related to farmers' well-being. They include the physical damage caused by erosion on plots, the negative impacts of erosion on soil nutrients and crop yields, and the threat to food security. All these issues denote farmers' evaluations of the impacts that land degradation may have on their ability to produce food for their family and to generate income for their welfare. Hence, greater perceptions of the severity of land degradation will cause greater awareness, and ultimately a greater inclination to embrace change.

Beside perceived severity of environmental damage, farmers' awareness of and attitude toward the environment was greatly affected by their perceived susceptibility to the problem. These results suggest that farmers will be more aware and inclined to develop a positive attitude toward environmental sustainability if they feel directly exposed to the problem. As indicated by Leventhal et al. (1965), individuals facing a particular threat consider their susceptibility to, and the severity of the problem. Immediate threats of soil erosion, for instance, will cause a more positive attitude toward the environment. The implication of these findings is that policy makers need to develop strategies to point out the importance of the seriousness of land degradation to the farming population. Not only is it important to stress the extensiveness of environmental damage, but also the consequences of not taking appropriate and immediate actions.

An important question raised is the extent to which farmers' attitude and awareness influence their behavior. Findings of this research support the hypothesis that awareness of environmental degradation significantly influences behavior. According to this result, supported by Napier and Napier (1991), greater awareness of environmental degradation is likely to cause farmers to change their behavior. This finding elucidates the mediating role of awareness in explaining the relationships between farmers' perception of the susceptibility to and severity of land degradation and their environmental behavior.

A critical finding of this research is the influence of crop dependency on the perceptual variables, and farmers' environmental attitude. Dependence on crop production directly influences farmers' perceived susceptibility, severity, benefits, barriers, and attitude toward environmental problems. These findings suggest that decisions should be made to improve agricultural production in order to increase earnings from that sector. Particular attention should be given to individuals who show great attachment to the land.

Another issue addressed in the study is the importance of land tenure in shaping individual's perceptions of land degradation. It appears that farmers who have more security on the land they operate are less aware of environmental damage, perceive the problem less severe, and impose greater barriers to change than those who have little control over their land. However, individuals who directly control their land are more likely to change their behavior. This situation is typical in areas where soil management projects are implemented. Farmers who own their land seem to be indifferent to land degradation because they are often economically well-off. They are ready to embrace change in order to obtain benefits from project interventions. It is apparent from the paths coefficients that dependency on crops has a stronger effect on their perceptions than does land tenure.

This study represents an attempt to develop a structural model to explain low-income farmers' environmental behavior. The causal effects of psychological and socio-economic factors on farmers' environmental behavior were evaluated. The study provides some useful insights on the determinants of farmers' environmental behavior structures, but it has one caveat. The nature of the statements and questions and farmers' levels of understanding may have increased respondents' biases. However, particular attention was paid in the design of the survey instrument and during training sessions to reducing enumerator's bias. Nevertheless, this study makes a significant contribution to the stock of knowledge of resource poor farmers' environmental behavior. The information generated in the study is particularly useful in addressing policy issues in terms of resource allocation to programs aimed at coping with behavior modification to reduce environmental degradation in Haiti.

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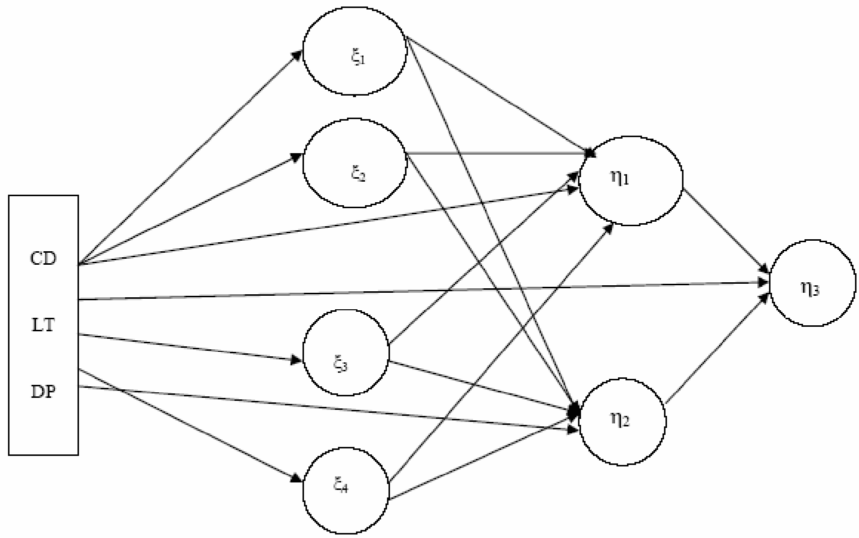

FIGURE 1
HYPOTHESIZED MODEL OF ENVIRONMENTAL BEHAVIOR

CD= crop dependency, LT= land tenure, DP= demographic pressure, ξ_1= susceptibility, ξ_2= severity, ξ_3=barrier, ξ_4=benefit, η_1 = awareness, η_2 =attitude, η_3 = behavior

TABLE 1

FACTOR LOADINGS ON FARMERS' BELIEFS OF ENVIRONMENTAL DEGRADATION

Items	Factor loadings	α
Perceived susceptibility		**0.80**
I am aware of erosion problems; hence I cannot be affected	0.58	
I use soil conservation techniques on my plots to limit erosion	0.86	
I maintain soil conservation structures to prevent erosion	0.87	
I plant trees to prevent erosion	0.74	
Perceived severity		**0.83**
Erosion can cause damage on my plots	0.81	
Erosion can reduce soil nutrients	0.86	
Erosion can reduce crop yields	0.78	
Erosion can cause famine in Haiti	0.65	
Erosion can cause damage on all plots	0.58	
Perceived benefits		**0.89**
I monitor my plots to detect erosion problems	0.72	
I always install erosion barriers on my plots	0.90	
I take some conservation measures while planting	0.80	
I look for technical assistance before planting in order to prevent erosion	0.85	
I encourage other farmers to establish soil conservation structures on their plots	0.81	
Perceived barriers		**0.84**
I don't protect my land because there is no erosion on my plots	0.64	
I don't search for technical assistance to protect my land because I can do it myself	0.45	
I don't look for aid because other people would think I am poor	0.87	
I don't look for aid because I don't like the technicians in the projects	0.86	
I don't search for aid to protect my land because project intervention is far from my zone	0.81	
I don't search for aid because I have no connections	0.71	
I don't protect my soil to avoid neighbors' hatred	0.51	

TABLE 2

FACTOR LOADINGS ON FARMERS' AWARENESS, ATTITUDE, AND BEHAVIOR FACTORS

Items	Factor loadings	α
Awareness		0.78
I am aware of erosion problems in Haiti	0.47	
I am aware of erosion problems in the zone	0.55	
I am aware of erosion problems on my plots	0.53	
I am aware that the farming practices in Haiti increases soil erosion	0.68	
I am aware that soil erosion reduces soil nutrients	0.76	
I am aware that soil erosion reduces plot yields	0.77	
Tree cutting is responsible for erosion	0.68	
Attitude		0.83
The environment in Haiti is in danger because the soil is washing away	0.68	
The soil in Haiti is eroded because of forest destruction	0.65	
Uphill practices affect downhill areas	0.66	
Erosion plays a role in diseases usually found in the zone	0.68	
Erosion causes water shortage in the country	0.59	
Every citizen is responsible for erosion problems in Haiti	0.63	
Local inhabitants are responsible for erosion problems in Haiti	0.75	
Behavior		0.72
Soil conservation is the best way to guarantee my family survival	0.50	
It is my responsibility to encourage my neighbors to adopt soil conservation techniques	0.79	
I have made financial efforts to protect the environment last year	0.73	
I have made major efforts to adopt conservation practices last year	0.76	
I have encouraged my neighbors to adopt conservation practices in the past year	0.73	
It is my responsibility to seek knowledge to solve environmental problems in Haiti	0.46	

TABLE 3

PREDICTORS OF ENVIRONMENTAL PERCEPTIONS, AWARENESS, ATTITUDE, AND BEHAVIOR

Path	Parameters	Standardized coefficient	t-value
crop dependence ∣ susceptibility	α_{11}	0.36*	8.25
land ownership ∣ susceptibility	α_{12}	0.08	1.79
demographic pressure ∣ susceptibility	α_{13}	0.02	0.38
crop dependence ∣ severity	α_{21}	0.20*	4.32
land ownership ∣ severity	α_{22}	-0.13*	-2.92
demographic pressure ∣ severity	α_{23}	0.05	1.01
crop dependence ∣ barrier	α_{31}	-0.16*	-3.61
land ownership ∣ barrier	α_{32}	0.12*	2.74
demographic pressure ∣ barrier	α_{33}	-0.08	-1.85
crop dependence ∣ benefit	α_{41}	0.30*	6.97
land ownership ∣ benefit	α_{42}	-0.04	-1.04
demographic pressure ∣ benefit	α_{43}	0.05	1.07
susceptibility → awareness	γ_{11}	0.24*	2.36
severity ∣ awareness	γ_{12}	0.35*	5.24
barriers ∣ awareness	γ_{13}	-0.001	-0.068
benefits ∣ awareness	γ_{14}	0.06	0.65
crop dependence ∣ awareness	γ_{15}	0.04	0.78
land ownership ∣ awareness	γ_{16}	-0.13*	-2.80
demographic pressure ∣ awareness	γ_{17}	-0.07	-1.65
susceptibility ∣ attitude	γ_{21}	0.44*	4.65
severity ∣ attitude	γ_{22}	0.22*	4.18
barriers ∣ attitude	γ_{23}	-0.04	-0.83
benefits ∣ attitude	γ_{24}	0.18	1.94
crop dependence ∣ attitude	γ_{25}	0.10*	2.32
land ownership ∣ attitude	γ_{26}	-0.01	-0.17
demographic pressure ∣ attitude	γ_{27}	-0.06	-1.57
awareness ∣ behavior	β_{31}	0.30*	4.32
attitude ∣ behavior	β_{32}	0.01	0.096
crop dependence ∣ behavior	β_{33}	0.05	0.99
land ownership ∣ behavior	β_{34}	0.09*	2.01
demographic pressure ∣ behavior	β_{35}	0.04	1.25

Note: *significant at $\alpha=0.05$, $\chi 2=669.41$, df=265; NFI=0.94, GFI=0.92, CFI=0.96

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Pollution in a Spatial Model: Is Zoning a Policy Response?

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Abstract

In this paper a Hotelling spatial model is developed which includes a single firm, that through its production process, creates a pollution externality on all consumers in a linear market. The social optimum location of the firm is derived, taking into account the pollution externality as well as transportation costs. One of the theoretical results is that spatial separation of industrial and residential land uses can be socially optimal, motivating an efficient zoning regulation. This theoretical hypothesis is tested empirically with spatial data on zoning location for a region in Maryland.

Keywords: Hotelling spatial model, pollution externalities, zoning, probit estimation

JEL Classification: Q2, R0, R3

Mathematics Subject Classification 2000: 91B76, 91B02

1. INTRODUCTION

One of the original motivations for local land use zoning ordinances was to separate non-conforming land uses, such as separating residential areas from the externalities associated with polluting industrial land uses (Pogodzinski and Sass, 1990). From an economic perspective, this non-market solution has the potential to be inefficient. However, Baumol and Oates (1988) have demonstrated that in a constant returns to scale linear model of two firms, where one firm increases the costs of production for the second firm through a pollution externality, if that externality is strong enough to cause a nonconvexity in the production set, then the social optimum will be a spatial separation of the two firms. This separation can be achieved through municipal zoning (Crone, 1983; Fischel, 1994) and this result suggests that a zoning regulation can be the socially efficient outcome. For example, Crone (1983) shows that zoning can be efficient by deriving conditions on relative land prices to indicate externalities strong enough to create non convexities in production sets. In a trade model, Copeland and Taylor (1999) demonstrate similar results for the spatial separation of polluting and nonpolluting industries.

This paper follows Baumol and Oates by developing a Hotelling model (1929), where producers and consumers are distributed along a linear market. Several authors such as Coase (1960), Wellisz (1964), Nutter (1968), Tybout (1972), and Marchand and Russel (1973) have used a scenario with a price taking firm imposing a negative externality on another firm, in their discussions of the Coase theorem. Among the few authors who have dealt with externalities in an oligopolistic setting, Levin (1985) analyzes an externality affecting the demand side of the market and Veendorp and Zambaras (1992) consider Cournot-type oligopolists that generate an externality that affects all of the firms in the market. Germain (1989) investigates one duopolist who imposes an externality on another, but the competition among the duopolists is still of the Cournot type.

The theoretical model presented in this paper is an extension of Gupt and Veendorp (1997a), where pollution externalities are incorporated into a Hotelling setting with a polluting upstream duopolist that imposed an externality, in the form of additional clean up cost on the downstream competitor. Within fixed-price variable-location (Gupt and Veendorp 1997a) and variable-price variable-location models (Gupt and Veendorp, 1997b), firm location decisions are modeled. The firms enter the market in the manner described by Prescott and Visscher (1977). There is a uniform distribution of consumers along the market who have an inelastic demand for the good produced by the firms, with the consumers bearing the transportation costs associated with consuming the good. One of the results for the fixed-price model is that the firms locate asymmetrically under different liability regimes and internalization of the externality through taxation is not an effective policy option to ensure the socially optimal location. Therefore, site regulation is suggested as a policy option.

The model for the present paper continues in this Hotelling setting, but with one firm that creates pollution as a result of its production technology that affects all downstream consumers. While Gupt and Veendorp (1997a, b) limited the firms to locate at specific points in and just outside the market, the firm in this model is allowed to locate at any point along the market. The pollution and transportation costs incurred by consumers are calculated for uniform and triangular consumer distributions for different parameter values using a pollution cost function similar to that used by Gupt and Veendorp (1997a, b). The social optimum location of the firm is derived, taking into account the pollution externality as well as transportation costs. One of the theoretical results is that spatial separation of industrial and residential land uses can be socially optimal, motivating an efficient zoning regulation. This theoretical hypothesis is tested empirically with spatial data on zoning location for a region in Maryland.

2. THE THEORETICAL MODEL

There is a single polluting firm that creates a pollution externality for the consumers located downstream in the linear market. The consumer distribution ($f(x)$) is assumed to take two forms: uniform and triangular. A triangular distribution affords a simple way to capture consumer concentration. In the characteristic space, the consumers could be clustered around some popular brand of a product, while in an urban setting, consumers could be concentrated around a central or regional business district. Tabuchi and Thisse (1995) use a symmetric triangular distribution with consumers concentrated in the center of the market. When the regional business district lies outside the market, the consumers can be expected to concentrate at the end of the market closer to the business district. A triangular distribution with a peak at one end of the market is more representative of this consumer distribution. The model in this paper considers a triangular consumer distribution that peaks at the upstream end of the market. The total population is normalized to one.

If the firm is located at x_A, the pollution cost (PC) for a consumer located at x is:

$$PC = \gamma \frac{q_A}{(k + x - x_A)}; x \geq x_A \text{ and } k > 0 \tag{1}$$

The consumers have a perfectly inelastic demand for the product and since there is only one firm in the market, it supplies the entire market (i.e. $q_A = 1$). The firm incurs a zero marginal cost of production. γ is the pollution cost coefficient which measures the intensity of pollution and takes on values between zero and one[1]. The constant k is included to guarantee that a consumer at the firm's location[2] does not have to face an infinite level of pollution. Hence, for a consumer at the location of the firm, ($x = x_A$) the pollution cost is $\frac{\gamma}{k}$. A larger value of k would decrease the pollution cost borne by a consumer located close to the firm.

The transportation cost per unit for each consumer is c multiplied by the distance between the consumer and the firm. The total transportation cost for all the consumers located along the market is obtained by integrating the product of c and this distance over the length of the market. In the next section, an analysis of the impact of the pollution intensity, the transportation cost, and the type of consumer distribution on the social cost minimizing location of the firm is performed. Two cases involving a uniform and a triangular distribution of consumers are considered.

2.1. Uniform Consumer Distribution: f(x) =1

With a uniform consumer distribution, the total transportation cost (TC) for all consumers is:

$$TC = \int_0^{x_A} c(x_A - x)dx + \int_{x_A}^{1} c(x - x_A)dx \tag{2}$$

$$= cx_A^2 + c(\tfrac{1}{2} - x_A) \tag{3}$$

---

[1] Lower values of γ can describe firms that pollute less or use cleaner technologies, while larger values (closer to one) would correspond to dirty technologies and bigger polluters.

[2] This is possible when the firm and the consumer are located opposite each other on two banks of a river.

The total social cost (C) is the sum of the pollution cost (PC) and the transportation cost (TC):

$$C = \int_{x_A}^{1} \frac{\gamma}{k + x - x_A} dx + c\, x_A^2 + c(\tfrac{1}{2} - x_A) \tag{4}$$

or

$$C = \gamma \ln(k + 1 - x_A) - \gamma \ln(k) + c x_A^2 + c(\tfrac{1}{2} - x_A) \tag{5}$$

The social cost minimizing first order condition with respect to the location of the firm requires:

$$\frac{dC}{dx_A} = \frac{(-\gamma)}{(k + 1 - x_A)} + c(2x_A - 1) = 0 \tag{6}$$

Therefore:

$$c(2x_A - 1) = \frac{\gamma}{k + 1 - x_A} \tag{7}$$

The second order condition requires:

$$\frac{d^2C}{dx_A^2} = 2c - \frac{\gamma}{(k + 1 - x_A)^2} > 0 \tag{8}$$

or

$$(k + 1 - x_A)^2 > \frac{\gamma}{2c} \tag{9}$$

Figure 1 illustrates the effect of various parameters on the social cost minimizing location of the firm. The value of x_A is below (or above) the optimal location whenever:

$$\frac{dTC}{dx_A} < 0 \ (> 0) \tag{10}$$

or

$$c(2x_A - 1) < \frac{\gamma}{k + 1 - x_A} \tag{11}$$

In Figure 1, the curve corresponds to the right hand side of Equation 7, while the lines c_S and c_L represent the left hand side of Equation 7 for small and large values of *c*, respectively. These lines exhibit small and large transportation costs by pivoting around the point (1/2, 0) because the transportation cost is minimum when the firm is located at the center of the market. As *c*, the transportation cost, increases (*ceteris paribus*), the point of intersection between the c_L line and the curve will move to the left and x_A will decrease. As *c* approaches ∞, x_A approaches one half, so as transportation costs increase, the firm should locate closer to the center of the market. On the other hand, with small transportation cost, the intersection point between the line c_S and the curve could be closer to the downstream end point of the market. For *c* values at which the curve and the straight line do not intersect, the optimal location for the firm would be just to the right of the point x_A = 1, .*i.e.* right outside of the market. Positions further downstream will not reduce pollution cost any further and will only increase transportation cost. As *k* increases, the curve shifts down and x_A decreases. This implies that as the pollution cost decreases with *k*, the social cost minimizing location of the firm can move towards the upstream end of the market. When *γ*, the pollution cost coefficient increases, the curve shifts upwards. The intersection point between the line and the curve will consequently move to the right (implying a larger value for x_A). Therefore, as the pollution cost increases, the social cost minimizing position for the firm would be close to or at the downstream end of the market.

2.2 Triangular Consumer Distribution: f(x) = 1 + b(1/2 - x) ; 0 < b < 2

The specific shape of this distribution is determined by the slope *b* (as shown in Figure 2). When *b* is zero, the distribution reduces to the uniform distribution. For non-zero values of *b*, the distribution pivots around the point (1/2, 1). Given this shape of their distribution, the pollution cost (*PC*) for all the consumers is:

$$PC = \int_{x_A}^{1} \frac{\gamma}{(k + x - x_A)} (1 + b(\tfrac{1}{2} - x))dx \tag{12}$$

$$= [1 - b(x_A - k - \tfrac{1}{2})]\gamma \ln \frac{k + 1 - x_A}{k} - \gamma b(1 - x_A) \quad (13)$$

When b=0, the lower extreme, the pollution cost is similar to the case where the consumer distribution is uniform:

$$PC = \gamma \ln \frac{k + 1 - x_A}{k} \quad (14)$$

When b=2, the upper extreme, the pollution cost is:

$$PC = 2(1 - x_A + k)\gamma \ln \frac{k + 1 - x_A}{k} - 2\gamma(1 - x_A) \quad (15)$$

The pollution cost is at a minimum (zero) when the firm locates at the downstream end of the market (i.e., at x_A=1) irrespective of the value of b or any other parameter. The sum of the transportation cost (TC) over all consumers is:

$$TC = \int_0^{x_A} c(x_A - x)(1 + b(\tfrac{1}{2} - x))dx + \int_{x_A}^{1} c(x - x_A)(1 + b(\tfrac{1}{2} - x))dx \quad (16)$$

$$= c\{-\tfrac{b}{3} x_A^3 + (1 + \tfrac{b}{2})x_A^2 - x_A + \tfrac{1}{2} - \tfrac{b}{12}\} \quad (17)$$

When b=0, the total transportation cost incurred by the consumers is:

$$TC = cx_A^2 + c(\tfrac{1}{2} - x_A) \quad (18)$$

For this type of uniform consumer distribution, the transportation cost minimizing location of the firm will be the center of the market, i.e. where x_A =1/2. As the value of b approaches 2, the transportation cost is minimized at firm locations closer to the upstream end of the market. When b =2 and the consumer distribution has a peak at the upstream end of the market, the total transportation cost is minimum when the firm's location, x_A, is 0.29.

The total social cost *C* is:

$$[1-b(x_A-k-\tfrac{1}{2})]\gamma\ln(\frac{k+1-x_A}{k})-\gamma b(1-x_A)+c\{-\tfrac{b}{3}x_A^3+(1+\tfrac{b}{2})x_A^2-x_A+\tfrac{1}{2}-\tfrac{b}{12}\} \qquad (19)$$

The first order condition to minimize this cost with respect to the location of the firm is:

$$\frac{\delta C}{\delta x_A}=\gamma b(1-\ln\frac{k+1-x_A}{k})+\frac{\gamma}{k+1-x_A}[b(x_A-k-\tfrac{1}{2})-1]+c\{-bx_A^2+(2+b)x_A-1\}=0 \qquad (20)$$

The second order condition for an interior minimum requires:

$$\frac{\delta^2 C}{\delta x_A{}^2}=[b(x_A-k-\tfrac{1}{2})-1]\frac{\gamma}{(k+1-x_A)^2}+\frac{2\gamma b}{k+1-x_A}+c\{-2bx_A+2+b\}>0 \qquad (21)$$

From the first order condition, when *k*=1 and *b*=0

$$\gamma\{\frac{1}{2-x_A}\}=c\{2x_A-1\} \qquad (22)$$

This result is a special case of that shown in Figure 1. The other possible extreme is when *b*=2 and *k*=1:

$$\gamma\{2\ln(2-x_A)\}=c\{4x_A-2x_A^2-1\} \qquad (23)$$

Figure 3 shows the effect of different values of the parameters on the social cost minimizing location of the firm from Equation 23. In the special case when *c* and γ have a unit value, which is shown in Figure 3, (with the *c*=1 curve as the right hand side of Equation 23, and γ=1 represents the left hand side) the line and the curve intersect to the right of the center of the market, near x_A = 0.6. For this combination of the parameter values, the firm can be located closer to the center of the market to minimize the total cost to society. When the transportation cost increases (*ceteris paribus*), the curve c = 1 will pivot upwards around the point x_A = 0.29, and the intersection point between the curves will move to the left[3]. The social cost minimizing location of the firm will be closer to the upstream end of the market. On the other hand, the curve γ = 1 will pivot upwards around x_A = 1 and the intersection point will move to the right when the pollution intensity increases. This means that when the pollution cost increases, the firm should be located closer to the downstream end of the market in order to minimize total social cost.

As can be seen from Figure 1, the socially optimal location for the polluting firm will be close to or at the downstream end of the market, for certain combinations of the parameter values γ, k, c and b, whether the model assumes a uniform or a triangular consumer distribution. These results are consistent with the observation that communities have an incentive to adopt a "beggar-thy-neighbor" policy and locate dirty industries close to their borders or downwind/downstream to save themselves from the imminent pollution (Fischel, 1985). In this way, local regulators minimize the social cost to their constituents, by passing on the pollution to neighboring localities. In the following section, tests of the hypothesis if local regulators are more likely to permit industrial land uses closer to their borders are performed, using spatially-explicit zoning and other data in a geographical information system (GIS) framework.

3. Econometrics

3.1. Data

The spatially-explicit zoning data are for seven counties in Maryland: Anne Arundel, Calvert, Charles, Howard, Montgomery, Prince George's, and St. Mary's. This geographical region covers about one thousand square miles. Approximately 50 percent of this area is covered by natural vegetation, 30 percent is occupied by agricultural land uses, 15 percent by residential land uses and industrial land uses

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[3] Which is one of the roots to the quadratic equation of the right hand side of Equation 23. The other root is x_A = 1.7, which is not considered here as it is outside the length of the market.

occupy about 5 percent of the area (Bockstael, 1996). The study area borders Washington D.C. and includes Annapolis, the state capital, located in Anne Arundel county. The zoning data are from the Maryland Office of Planning and reflect the zoning for these counties in 1995. Zoning is regulated at the county level in Maryland. In addition, other spatially-explicit data include information on the road network and data from the 1990 Population Census.

The dependent variable for the econometric model was created from these digital zoning maps, aggregated into two categories: “dirty” (industrial) and “clean” (residential, commercial, agricultural, forestry). To standardize the unit of observation, a grid of 50m by 50m was drawn over the zoning map. To reduce the number of observations, the data were *resampled* to a coarser resolution of 150m by 150m. The explanatory variables from the Census and other digital maps were then linked to the 150m observation through GIS.

3.2. Hypotheses

The results of the theoretical section of this paper suggest that local planners have an incentive to locate dirty land uses close to their boundaries. To test this hypothesis if planners are more likely to zone industrial land uses closer to their borders, controlling for other features, a probit model was estimated with “dirty” and “clean” zoning as the dependent variable. The distance for each observation to the land boundary (*landist)* and water boundary (*watdist*) of the county was calculated, to use in a test of if distant to land boundary decreases the probability of an observation being zoned “dirty”. A distinction is made between the land and water boundaries to account for possible efforts made by planners to maintain the pristine quality of water bodies. These efforts could be motivated by their concern for environmental preservation or to maintain the residential land value of expensive water front property.

Other controls included in the model that are expected to affect the probability of being zoned “dirty” are: distance to nearest major road (*rdist*); distance to the central business district of Washington D.C. (*wdcdist*); population density in the block group that the observation fall in (*popden*). The *wdcdist* and *rdist* variables are proxies for the effect of the transportation cost coefficient (*c*) on zoning decisions from the theoretical model, and *popden* captures the effect of the pollution intensity (γ). The *a priori* hypotheses on the control variables are that *rdist* and *wdcdist* will each have a negative effect on the probability of being zoned "dirty", as these capture the transportation costs of doing business, and *popden* is also hypothesized to negatively influence the probability of being zoned "dirty", as planners often keep non-conforming land uses apart.

The type of neighborhood an observation is located in could potentially also affect its probability of being zoned "dirty" or not, for example due to agglomeration effects. Industrial firms might wish to locate near each other or planners may want to concentrate polluting industries in a specific area to keep the other parts of their jurisdiction "clean." In order to capture this effect a neighborhood variable was calculated (*nbhd*), where a neighborhood was considered "dirty" if a majority of the surrounding nearest observations were also zoned "dirty". Landis and Zhang (1998) use a similar measure of neighborhood effects in their study of the determinants of land-use change in a discrete choice model. Case (1992) finds that ignoring such neighborhood effects may bias the estimation of parameters of interest.

3.3. Empirical Specification

It is assumed that there is a linear function of the explanatory variables that is an index of industrial potential *I* of that observations(Greene,1997; Gujarati,1995):

$$I_i = \alpha_i + \beta' x_i + \mu_i \tag{24}$$

Where the x_i are explanatory variables, and the α and β are parameters to be estimated, and μ is an unobserved error term. I_i is unobservable, however y is observed where:

$$\begin{aligned} &y = 1 \text{ if } I_i > I_i^* \\ &y = 0 \text{ otherwise} \end{aligned} \tag{25}$$

That is, if the index of industrial potential for an observation exceeds a certain critical value I_i^*, the observation will be observed to be zoned "dirty" and otherwise it will be zoned "clean". If the errors are assumed to be independently and identically distributed normally, then the specification becomes a probit model. In the model estimated below, each explanatory variable is interacted with a county dummy to allow for the effects to differ by county, as zoning regulations are set in Maryland at the county level. County dummies are also included to control for unobserved variation in the quality of public goods, tax rates, crime rates, etc. that vary at the county level. The variables and their summary statistics can be found in Table 1.

3.4. Estimation and Results

The estimated coefficients and their significance levels from the econometric model are found in Table 2. There are mixed results for the main variable of interest, the distance to the land border of the county (*landist*). For three out of the seven counties (Calvert, Charles, Howard), the estimated coefficient is negative and statistically significant, implying that holding all else constant, an increase in the distance from the land border for these counties decreases the probability that the observation will be zoned industrial, the result that was hypothesized from the theoretical mode. For three of the counties (Anne Arundel, Montgomery, Prince George's), this variable is not statistically different from zero and for one county (St Mary's) the estimated coefficient was positive and significant. Perhaps these results are because Anne Arundel, Montgomery, and Prince George's were developed much earlier than the other counties, so that much of their current industrial zones are grandfathered land uses.

Turning to the control variables, most of the empirical results are consistent with the *a priori* expectations. For two of the counties (Anne Arundel, Prince George's), the distance to the water boundary (*watdist*) was positive and significant, so that these counties appear to keep industrial zones away from their water boundaries, while for the remaining counties, the estimated coefficients were not statistically different from zero. The distance to roads (*rdist*) and distance to Washington D.C. (*wdcdist*) variables act as proxies for the transportation cost coefficient from the theoretical model. The proximity to major roads plays an important role in determining the probability of a pixel of land being zoned industrial in all the counties except St. Mary's. The *rdist* variable has a negative and significant effect on this probability in six of the counties. However, the estimated coefficients on the *wdcdist* variable are more mixed. It has a positive and significant effect for Anne Arundel and Calvert and a negative and significant for Charles, Howard, Prince George's, and St. Mary's, and is statistically insignificant for Montgomery. Therefore, for four out of the seven counties, the *a priori* expectation was met that the further from the central business district, all else being equal, the lower the probability of an observation being zoned industrial. Perhaps the counter-intuitive effect for Anne Arundel is due to the state capital, Annapolis, is in this county which is not controlled for in the model, while a duo-centric model might capture this effect.

The population density variable (*popden*) attempts to control for the much more heterogeneous distribution of consumers than is possible in the theoretical model and acts as a proxy for the pollution cost variable. The results here are mixed as well. A negative and statistically significant coefficient resulted for three of the counties (Anne Arundel, Calvert, Prince George's), as expected, however, there is a positive and statistically significant coefficient for one county (St. Mary's) while for the remaining counties, the coefficient is statistically insignificant (Charles, Howard, Montgomery). The neighborhood (*nbhd*) and county dummy variables act as controls for unobserved variation. The neighborhood variable controls for unobserved factors that are spatially autocorrelated while the county dummies control for all unobserved variation that differs by county. This latter variation could arise from different tax rates, crime rates, schools and other public facilities. The results on the neighborhood variable are all positive and statistically significant, demonstrating the potential for some unexplained agglomeration in industrial zoning, holding all else equal. That is, the more industrial an area, the greater the probability that an individual observation will be zoned industrial. For the county dummies, the excluded county is Prince George's, and the estimated coefficients are all statistically significant: Anne Arundel, Calvert and Montgomery are less likely than Prince George's to zone industrial, while Charles, Howard and St. Mary's are more likely than Prince George's to zone industrial.

4. Conclusions

When a single firm in a Hotelling-type linear market, such as a firm located along a river, disposed of waste in that river, it imposes an external pollution cost on downstream consumers. The total cost to society is the sum of the pollution cost and the transportation cost associated with the location of the firm. The results derived from the theoretical model in this paper indicate that the transportation cost minimizing location of the firm is at the center of the market when a uniform distribution of consumers is considered and moves closer to the upstream end of the market as the distribution becomes triangular (with a peak at the upstream end of the market). The pollution cost is minimized when the firm locates close to or at the downstream end of the market irrespective of the shape of the consumer distribution or the value of any other parameters. The total social cost in the presence of the externality can be minimized by locating the polluting firm close to the downstream end of the market for certain combinations of the parameters.

This theoretical result that suggests that planners might deliberately zone industrial land uses close to their boundaries in an effort to avoid having the externality affect their residents. This hypothesis was tested using spatially-explicit zoning data for seven counties in Maryland; specifically to test if the distance from the county's land boundary had a negative and statistically significant effect on the probability that an observation is zoned industrial. In many instances, the empirical results supported this hypothesis. The other control variables in the econometric model, for the majority of the cases also met *a priori* expectations. The empirical observations support the results of the theoretical model that suggest that local planners might have an incentive to locate dirty land uses close to their boundaries.

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Table 1: Summary Statistics of the Variables

Variable	Mean	Standard Deviation	Variable Description
zones	0.03387	0.18089	Type of zoning 0 = non industrial 1 = industrial
landist	9856.913	9922.522	Distance between center of a pixel and the nearest land boundary in meters
watdist	6663.542	7392.727	Distance between center of a pixel and the nearest water boundary in meters
rdist	1823.271	1849.293	Distance between center of a pixel and the nearest major road in meters
wdcdist	33648.54	18980.97	Distance between center of a pixel and Washington D.C. in meters
popden	1.341902	2.712811	Population density (people per acre) of the census block group to which the pixel belongs
nbhd	0.022764	0.149149	Neighborhood 0 = clean 1 = industrial
aalandist	978.7167	2864.579	Distance to land boundary for Anne Arundel county in meters
calandist	1499.401	6114.864	Distance to land boundary for Calvert county in meters
chlandist	1679.375	4789.909	Distance to land boundary for Charles county in meters
holandist	327.8764	1264.225	Distance to land boundary for Howard county in meters
molandist	1475.314	3882.903	Distance to land boundary for Montgomery county in meters
pglandist	866.006	2251.625	Distance to land boundary for Prince George’s county in meters
smlandist	3030.224	9332.05	Distance to land boundary for St. Mary’s county in meters
aawatdist	635.692	2216.456	Distance to water boundary for Anne Arundel county in meters
cawatdist	206.526	877.595	Distance to water boundary for Calvert county in meters
chwatdist	947.222	2823.17	Distance to water boundary for Charles county in meters
mowatdist	2293.774	5891.524	Distance to water boundary for Montgomery county in meters
pgwatdist	2265.789	6133.268	Distance to water boundary for Prince George’s county in meters
smwatdist	314.540	1112.218	Distance to water boundary for St. Mary’s county in meters
aardist	193.693	629.310	Distance to major road for Anne Arundel county in meters
cardist	186.378	818.087	Distance to major road for Calvert county in meters
chrdist	463.747	1378.76	Distance to major road for Charles county in meters

Contd ---Table 1

hordist	106.355	439.057	Distance to major road for Howard county in meters
mordist	292.254	1022.126	Distance to major road for Montgomery county in meters
pgrdist	257.111	843.778	Distance to major road for Prince George's county in meters
smrdist	323.734	1128.5	Distance to major road for St. Mary's county in meters
aawdcdist	4854.848	11770.77	Distance to D.C. for Anne Arundel county in meters
cawdcdist	3832.198	13450.94	Distance to D.C. for Calvert county in meters
chwdcdist	6101.299	13975.96	Distance to D.C. for Charles county in meters
howdcdist	2950.116	9348.598	Distance to D.C. for Howard county in meters
mowdcdist	4193.635	10051.67	Distance to D.C. for Montgomery county in meters
pgwdcdist	2621.618	6558.94	Distance to D.C. for Prince George's county in meters
smwdcdist	9094.829	23556.04	Distance to D.C. for St. Mary's county in meters
aapopden	0.24210	1.10686	Population density (people per acre) for Anne Arundel county
capopden	0.02980	0.14038	Population density (people per acre) for Calvert county
chpopden	0.05837	0.39630	Population density (people per acre) for Charles county
hopopden	0.10887	0.67418	Population density (people per acre) for Howard county
mopopden	0.43604	1.84948	Population density (people per acre) for Montgomery county
pgpopden	0.42261	1.84283	Population density (people per acre) for Prince George's county
smpopden	0.04412	0.20398	Population density (people per acre) for St. Mary's county
aa	0.15277	0.35977	County dummy for Anne Arundel
ca	0.08016	0.27154	County dummy for Calvert
ch	0.17119	0.37668	County dummy for Charles
ho	0.09368	0.29139	County dummy for Howard
mo	0.18446	0.38786	County dummy for Montgomery
pg	0.18181	0.38569	County dummy for Prince George's
sm	0.13593	0.34271	County dummy for St. Mary's

Table 2: Econometric Results, Probit model, dependent variable, zoning (industrial or not) n=309,884 Pseudo R^2 = 0.7444

Variable Name	Estimated Coefficient	t-statistic	Variable Name	Estimated Coefficient	t- statistic
aanbhd	3.428*	70.942	hordist	-4.96e-04*	-9.51
canbhd	4.467*	26.973	mordist	-1.89e-04*	-6.66
chnbhd	4.212*	44.414	pgrdist	-2.17e-04*	-9.95
honbhd	4.082*	48.891	smrdist	2.67e-05	1.46
monbhd	4.244*	57.231	aawdcdist	6.56e-05*	19.37
pgnbhd	4.149*	83.975	cawdcdist	7.88e-05*	4.92
smnbhd	4.862*	28.019	chwdcdist	-2.61e-05*	-7.20
aalandist	-1.04e-05	-1.95	howdcdist	-2.09e-05*	-5.13
calandist	-7.82e-05*	-4.73	mowdcdist	-2.39e-06	-0.97
chlandist	-1.82e-05*	-3.60	pgwdcdist	-1.76e-05*	-6.81
holandist	-2.14e-04*	-11.34	smwdcdist	-1.39e-04*	-4.30
molandist	7.16e-06	1.69	aapopden	-0.102*	-8.68
pglandist	-1.24e-06	-0.23	capopden	-0.756*	-3.80
smlandist	1.47e-04*	4.45	chpopden	-0.026	-1.20
aawatdist	9.15e-05*	14.25	hopopden	0.006	0.35
cawatdist	1.11e-05	0.36	mopopden	-0.006	-0.90
chwatdist	5.87e-06	1.11	pgpopden	-0.034*	-5.98
mowatdist	-3.85e-06	-1.12	smpopden	0.181*	6.57
pgwatdist	8.53e-06*	4.64	aa	-2.671*	-16.02
smwatdist	2.1e-05	0.83	ca	-2.371*	-4.31
aardist	-6.6e-05*	-4.48	ch	0.695*	4.43
cardist	-4.98e-04*	-7.21	ho	0.986*	6.38
chrdist	-2.62e-04*	-12.03	mo	-0.474*	-4.37
constant	-1.887*	-29.66	sm	4.785*	3.32

*** significant at the 5% level**

Figure 1 The relationship between the pollution cost, the transportation cost and the social cost minimizing location of the firm.

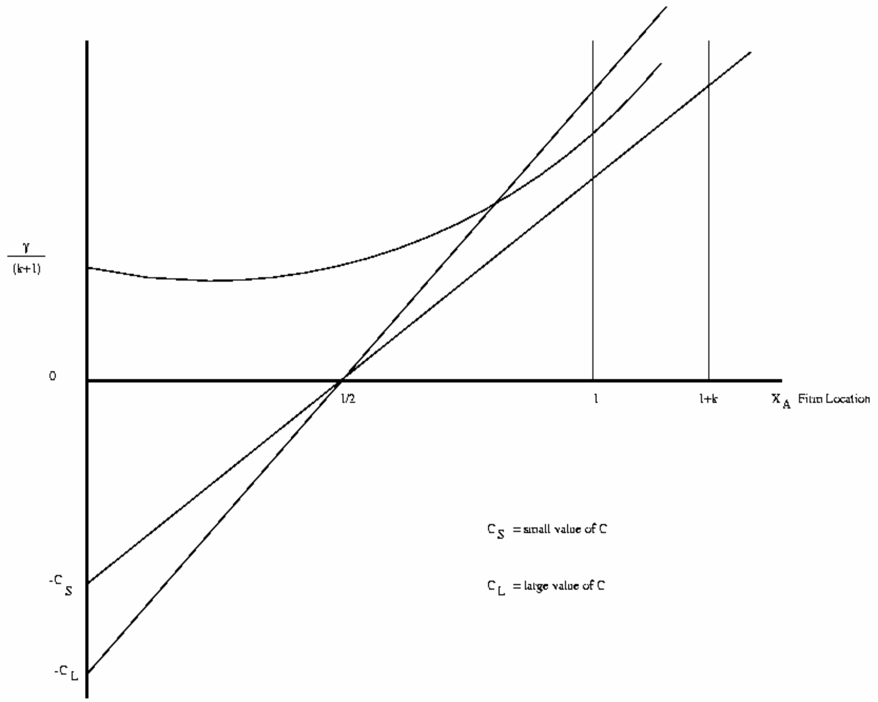


Figure 2 The triangular consumer distribution

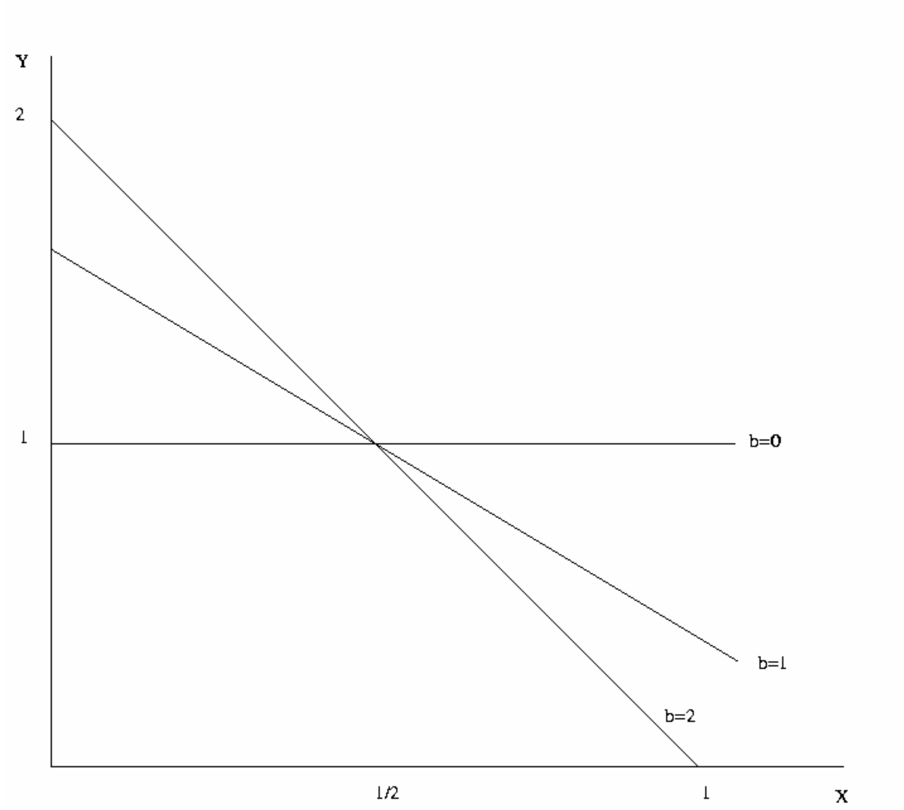
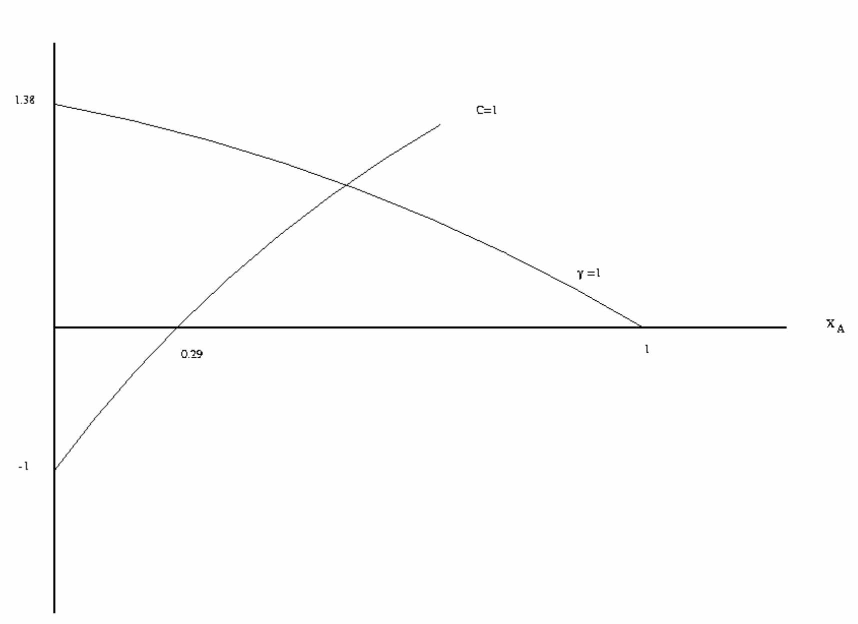


Figure 3 The effect of c and γ on the social cost minimizing location of the firm.
c=γ =1



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