Environmental and economic efficiency in recycling of household waste, pollution control and land-use changes

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Preface

This dissertation was prepared when I was a PhD student at the Department of Economics and the Industrial Ecology Programme at the Norwegian University of Science and Technology. My supervisors have been Professor Anders Skonhoft and Senior Researcher Kjell Arne Brekke.

The issues addressed in this dissertation are related to the link between environmental degradation and economic activities. The dissertation is presented in five papers which can be read separately. The first paper is an introductory to the dissertation which summarizes and discusses the subsequent four papers. In the next paper the issue of whether household waste should be recycled into new products or incinerated in order to produce district heating is discussed. The theoretical model presented in this paper is then used in the third paper as a basis for an empirical analysis of plastic packaging waste generated in households in the city of Trondheim in year 2000. Paper 4 analyzes pollution control under uncertainty in a dynamic model where the decision problem is to allocate resources between production of goods and pollution abatement. In paper 5 the focus is on how the change in wilderness land can be related to the level of economic activity and economic growth in Norway in the years 1988-1994.

Trondheim, March 2004

Håvard Solem

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In 2002, I spent six months at the University of California at Berkeley where Professor David Zilberman helped me sort out problems related to what have become Papers 2 and 4 in my dissertation. During my stay in Berkeley I also got the chance to audit a course in ecological economics lectured by Professor Richard B. Norgaard, which opened my mind to new and intriguing issues related to the relationship between environmental problems and economics. I would also like to thank Professor Ragnar Torvik at NTNU and his family for making my family's stay in California something that we will always remember as a good time.

My wife Ingebjørg and my daughter Marie have not seen as much of their husband and father as we all would have wanted during the past four years. Even though I tended to be somewhat preoccupied with my work when I was at home, my little family never stopped supporting me in my efforts to earn a doctoral degree. I am eternally grateful to both of you.

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Paper 1

Introduction: including ecological principles in economic analyses

> Advances in fundamental science have made it possible to take advantage of the uniformity of matter/energy, a uniformity that makes it feasible without preassignable limit, to escape the quantitative constraints imposed by the character of the earth's crust... Nature imposes particular scarcities, not an inescapable general scarcity.
>
> (Barnett and Morse 1963, p. 11)

> One must have a very erroneous view of the economic process as a whole not to see that there are no material factors other than natural resources. To maintain further that 'the world can, in effect, get along without natural resources' is to ignore the difference between the actual world and the Garden of Eden.
>
> (Georgescu-Roegen 1975, p. 361)

1. INTRODUCTION

This dissertation includes four essays on problems related to the relationship between economic activity and environmental degradation. Two of the most important physical laws that govern this relationship are the first and second laws of thermodynamics. The first law refers to the principle of conservation of matter and energy, namely that matter and energy cannot be created or destroyed, only transformed. The second law deals with energy transformations and the concepts of energy quality. Every time useful energy is converted or transformed from one state to another there is always less useful energy available in the second state than there was in the first (Ruth 1993). All of the essays in this dissertation are anchored to the first, and to some extent the second laws of thermodynamics. Papers 2 and 3 deal with the problem of handling the waste flow that is generated by household consumption, both theoretically and empirically, paper 4 focuses on environmental problems in the form of stocks of some given hazardous material, and paper 5 is an empirical analysis of the relationship between the amount of wilderness land and economic activity and growth in Norway.

What has inspired the problems that are addressed in this dissertation is that we find ourselves in a world in which the human population has risen dramatically over the past century and that is likely to double during the next (World Resources 1992). This development brings an increase in consumption and the production of goods and services. A reduction in the level of required material input and energy use per unit GDP resulting from technological development is counterbalanced by the growth in both the population and in the requirements and needs of the current population. Consequently, a boost in matter and energy exchanges between the environment and the economy is seen, which contrasts the world's limited resource base which contains a complex and interrelated set of ecosystems that show signs of fragility. An important question that follows is how far analyses by economists have allowed for the limitations inherent in the physical laws governing the economy-environment interactions.

Way back in the 3rd century B.C. Epicurus stated that "Nothing is created out of that which does not exist: for if it were, everything would be created out of everything with no need of seeds. And again, if that which disappears were destroyed into that which did not exist, all things would have perished, since that into which they were dissolved would not exist" (Bailey 1926, p. 21). Out of this came the laws of thermodynamics that are loosely interpreted into the economist's notion of "no free lunch". The discovery of the laws of thermodynamics in the early nineteenth century contributed to a clear understanding of the physical laws governing the transformations of matter and energy. Contrary to what one would think the degree to which the discipline of economics has incorporated insight from the natural sciences into its models varies a good deal. A strand of economists, sometimes loosely termed ecological economists are trying to remedy this aspect, which many consider a weak point in the discipline of economics.

Issues on the biophysical foundations of economics were later explored using the works of Georgescu-Roegen (1971). Together with Georgescu-Roegen, Herman Daly and Kenneth Boulding argued that the limits to economic growth could no longer be explained exclusively on the basis of the possibility of running out of conventional resources – the traditional Malthusian approach (Boulding 1966; Daly 1974). The arguments were extended to include complementarity of production factors, restricted regenerative and assimilative capacity of the natural environment and that technology could not be viewed as the ultimate means of circumventing ecological limits. Lately, many of the ideas underlying ecological economics have transfused into neoclassical environmental and resource economics as we often see concepts such as material balance, entropy law, limits to absorptive capacity of the natural environment, and carrying capacity frequently being used.

However, many mainstream economists argue that natural resources are not becoming increasingly scarce and the arguments are in fact based on the possibilities for substitution between capital and natural resources, technological progress and that the prices of increasingly scarce resources will increase and encourage more effort to be put into locating new sources of the resource or finding substitutes (Barnett and Morse 1963; Simon and Kahn 1984). The most orthodox economists would argue that the majority of environmental problems can be solved by economic instruments, i.e. if we just "get the prices right". Bluntly formulated, efficiency is obtained by identifying the nature of the environmental externality and implementing the corresponding Pigouvian tax. Compared to the neoclassical approach, the ecological economic perspective appears to be based more on a precautionary approach:

> The problem of ecological sustainability needs to be solved at the level of preferences or technology, not at the level of optimal prices. Only if the preferences and production possibility sets informing economic behavior are ecologically sustainable can the corresponding set of optimal and intertemporally efficient prices be ecologically sustainable. Thus the principle of "consumer sovereignty" on which most conventional economic solutions are based, is only acceptable to the extent that consumer interests do not threaten the overall system – and through this the welfare of future generations.
>
> (Costanza et al. 1997: xv)

The cautious approach results from looking at biophysical limits from a broader context. Moreover, the human economy is viewed as nothing but a small, albeit important, subset of the natural ecosystem. The two systems are considered interdependent, which is why ecological economists focus on understanding the linkages and interactions between economic and ecological systems. Resources, including assimilative capacity, exist in a finite amount, and they are complements in production processes. Given this perspective, the scale of human activities becomes an important issue. The concern is therefore that the population size and aggregate consumption of resources are already approaching the limits of the finite natural world (Ayres 1978; Pearce 1987).

The perception that there exists a sustainability problem was also pronounced in the book, *The Limits to Growth* (Meadows et al. 1972), which was widely understood to claim that environmental limits would cause the collapse of the world economic system in the middle of the twenty-first century. Basically the book reported the results obtained from a computer model of the world system, which predicted that the limits to growth would be reached within 100 years with a sudden and uncontrollable decline in both population and industrial capacity. Most economists were critical about the conclusions of this report and the main criticism was that the feedback loops in the model were poorly specified in that they failed to take account of behavioral adjustments operating through the price mechanism (see e.g. Beckerman 1972; Nordhaus 1973). It was conceded by some economists that this argument was weakened by the fact that markets did not exist for many environmental resources. However, the counterargument was that part of this problem could be remedied by proper policy responses.

The disagreement clearly has some important implications for how we analyze and model economic processes that include some kind of environmental aspect. The principle linking the essays of this dissertation together is the attempt to take the implications following from the laws of thermodynamics into consideration when performing my economic analyses. The way in which the papers reflect the laws of thermodynamics varies from explicit use of material balances in the analysis of solid waste handling systems (papers 2 and 3), and to a somewhat less extent in my analysis of controlling abatement efforts in a dynamic setting (paper 4). It is least explicit in the analysis of the relationship between economic growth and the change in wilderness land (paper 5), but is nevertheless a core aspect underlying the problem at hand. In order to explore the relationship between economics and thermodynamics I will give a brief outline of the development of economic methodology before we investigate the impacts from natural science and the criticism from ecological economists.

1.1. Economic methodology

The methodology of the nineteenth-century British economists like Smith, Ricardo and Malthus focused attention on theory and was defensive about the validity of verification of economic predictions (Blaug 1980, chapter 3). The grounds for theory were introspection or casual observations constituting *a priori* truths, which were true *a posteriori* only in the absence of disturbing causes[1]. In this way the purpose of verifying implications was to determine the extent of applicability of economic theory and not to test its validity. There was hence no accepting of the symmetry thesis saying that explaining is equivalent to prediction. This means that economists of that time were

[1] This has led to the characterization of economic rules as being tendency laws. In order to control for contradicting effects a tradition for applying *ceteris paribus* assumptions has been echoed in modern economics. It is widely believed that this assumption is much rarer in the natural sciences, which some claim is not true (Blaug 1980).

verificationists rather than falsificationists because they took a defensive position in order to secure the young science against attacks. Robbins sums up the economic methodology of the nineteenth century by stating that the validity of a theory is dependent on its logical derivation from the general assumptions, but its applicability to the real world is depending on whether it is encompassing the forces operating in that given real-world situation (Robbins 1935).

As a reaction to the economic methodology of the nineteenth century which was a system of pure deductions from a set of postulates derived from introspection that are not open to external verification, Hutchison (1965) proposed a prescription that economic methodology should be confined to statements which are empirically testable. Machlup calls this ultraempiricism and argues that it implies a system based on observations, which is not applicable for a social science like economics. Unlike the natural sciences the data of observations are themselves interpretations of human actions by human actors (Machlup 1978). Arguments similar to Hutchison's are found in Friedman (1953) in the defense of his irrelevance of assumptions thesis. He argues that theory should be judged by its predictive power for the phenomena that it is intending to explain. If predictions of a hypothesis have survived many opportunities to be contradicted we will have great confidence in it, despite the lack of realism that we find in the assumptions that the theory is based on. Friedman does not refer to Popper but his arguments are similar to a great extent when he finds that a desirable criterion of good theory is simplicity (which increases the possibility of falsification) and hence isolates the economic mechanisms from disturbing effects. Economics are later held to be a box of tools, and empirical testing is not so much a tool for rejecting or accepting a theory, but to see the degree to which a theory is applicable in a given situation. As current mainstream methodology of economics has been labeled as innocuous falsificationism because of its highly protective "rules of the game", "...almost any model will do provided it is rigorously formulated, elegantly constructed, and promising of potential relevance of real-world situations." (Blaug 1980, p. 128).

At the core of real-world economic situations we find the production process, i.e. the transformation of matter and energy. This is the main link between the economic system and nature. Here it is important to consider the influence that the natural sciences have had on mainstream economics.

1.2. Economic methodology and its relations to the natural sciences

The emergence of economic methodology as we know it has been influenced by the natural sciences, in a very distinct way. Since the success of Newton's contributions to the development of natural science, the use of mathematical methods has become an ideal for other branches of natural science and a source for perceiving of the social sciences as inferior to the natural sciences because of the former's lack of formalization. Even before Newton's *Principa*, there were a number of attempts to construct the social sciences on the principles of mathematics and the natural sciences as early as in the seventeenth century. These efforts considered the accomplishments of among others Descartes, Galileo, and Kepler. Neo-classical economists like Jevons, Pareto, and Fisher declared physics to be a model for economics as a discipline in order to reach a goal of making economics a "true" science (Cohen 1994). It is later argued that economists "boldly copied the reigning physical theories." (Mirowski 1989, p. 454) because mathematical physics was the part of an exact science with the highest standing and would show that their subject shared the features of an exact science. It is also argued that economics in the mid-nineteenth century proved to be well adapted to the application of mathematical techniques as can be seen in the theories constructed by economists like Edgeworth, Jevons, and Walras (Cohen 1994)[2].

However, the adoption of, and the heavy reliance on mathematical methods has been proposed as the reason for making economics *less* scientific (Payson 1997). It is argued that it is not sufficient to obtain and manipulate data, one must also understand what the data actually means. Payson (1997) pleads that the wide use of abstractions from reality is a sign of lack of scientific practice, and that this is the main problem for the discipline. The same argument is put forward by Ravetz, who states that "Perhaps the economists were the victims of the doctrines they learned from the philosophy of science, which concentrated on abstract problems of validation of theories while ignoring the principles of measurement." (Ravetz 1995, p. 174). Keeports and Morier (1994) argue along the same lines, and label methodology based on Friedman's irrelevance of assumptions thesis, pseudoscientific belief. This is because sound reasoning using a false hypothesis may lead to true predictions. According to these authors, the reason for economists having a different perspective on science is that the natural scientist is devoted to having a purely objective view of reality (including any assumptions), while the economist is only interested in the conformity of a hypothesis with empirical observations. An explanation of this is offered by Payson (1997) who says that economics is less interdisciplinary than science. Knowledge in other disciplines can lead economists to "assume things away" not because of convenience,

[2] Whether the social sciences are suitable for application of a particular technique depends on the state of development reached by the subject. It is also depending on the degree of development of the natural sciences and whether this permits application. An example of such an application is the use of comparative statics in economics.

but because they can base their assumptions on an approach where the effect of given factors are negligible.

While it can be argued that the logic and technique do not differ much, the division between the natural sciences and economics can be traced back to the difference in the objects that are studied. Machlup (1978) has labeled this difference "silent nature, talking man" which is based on Schutz (1953) who argues that the facts, events, and data the social scientist refers to are founded in a context in which he or she is a player, while the observational field of the natural scientist does not mean anything to the objects themselves. This difference is an argument in favor of those that do not consider the methodology of physics to be the ideal and the standard for what is to be called science. In economics the scientist has the possibility of accessing data for inner experience that is not available to the natural scientist. Such introspection and communication with the players on the social scene makes the source of information larger in the social sciences than in the natural sciences.

However, this difference in observations and data between the disciplines can be seen both as an advantage and as a disadvantage. It can be argued that it is the personal experience that enables us to understand mechanisms within observations and hence formulate more accurate hypotheses about the scientific object. Theorizing *a priori* is necessary to understand and be able to interpret the observations made by the scientist. This is as important in explaining action as explaining intentional refraining from action.

Mainstream neoclassical economic theory explains human behavior by self-interest and rationality. It is assumed that people take into account whatever information they need to assess the consequences of each alternative and choose the one that maximizes their net utility (see for example Gravelle and Reese 1992, chapter 1). However, other social sciences question the degree to which economic theory is able to reflect and explain behavior affected by social aspects such as culture, altruism, laws and social norms (Elster 1989). Baland and Platteau (1996) offer two ways of addressing the issue of social forces in economic models. These may be reflected as a binding constraint on the choices of the self-interested utility maximizing individual or they can be included in the preference structure of the individual agents.

An area where the above discussion is relevant is the problem concerning the assessment of utility and costs related to the effort made by households in municipal waste handling systems. This hotly debated issue is considered in papers 2 and 3 of this dissertation. Attempts to identify the relationship between household effort and the sorting level have been carried out but most of the studies focus on finding out what motivates or facilitates recycling (see Tasaday 1991; Hornik et al. 1995). What these studies implicitly assume is that there are no costs associated with the effort that households put into recycling activities, meaning that the net benefits increase with the effort. Contrasting this is the observation that less than 100% of the potentially recyclable amount of waste is sorted from the rest of the waste fractions in the households, which indicates the existence of barriers within the sorting activity (Kinnaman and Fullerton 1999; Eik et al. 2002). Thus, in the absence of legal requirements and possibilities of sanctions against violators, the level of sorting may be seen as the result of some kind of utility maximization. Papers 2 and 3 in the dissertation do not reveal anything new about the microeconomics of household sorting, but rather add insight to the important role of the households in recycling systems. This shows how they influence the solutions, demonstrates how they can hinder the implementation of the efficient solution and consequently, why it is important for policy makers to be aware of this issue.

It is evident from this discussion that part of the challenge facing the economist is rooted in the fact that the social sciences make "subjective" or psychological inventions of hypotheses from objective occurrences. This is opposed to the natural sciences where they do not have first-hand experience, but make hypotheses from observations "without knowing what it is like to be a rapidly moving molecule." (Nagel 1961, p. 484). This first-hand information can be misleading as well as helpful because the classes of social phenomena are poorly bounded compared to the more well-bounded class of physical phenomena, which creates differences in the possibilities of plausible generalizations. The chance of experiencing observations that contradict theories is larger when the observations are subject to interpretations by the object itself or by the scientist[3]. The question of whether the observations in the social sciences are conscious or unconscious reactions must also be taken into account. These two elements have to be considered by the social scientist and can be a source of new knowledge, which can be valuable for modifying existing theories.

The main difference is that the natural scientist does not communicate with the molecules, but if he or she had to, this scientist would have to deal with the help or challenges this would imply. So the impression is that the main difference between economics and the natural sciences lies more in the focus of the analysis and in the objects of investigation rather than in the methodology and logic of the disciplines. Let

[3] Philosophers of science are now fully agreed that almost all disciplines include a core of general propositions, independent of time and space, which can be applied to concrete situations or particular cases (Machlup 1978).

us now turn to the arguments for a closer integration of economics and ecology put forward by ecological economists.

1.3. Inputs from ecology

From the previous section we were left with an impression that the main difference between economics and the natural sciences was based more in the objects of investigation rather than in the methodology and logic of the disciplines. However, from an ecological economics perspective an equally important argument is that there is a discrepancy about the relevance of the laws of thermodynamics for the performance of economic systems between economists and natural scientists (Ruth 1993; Daly 1987; Georgescu-Roegen 1971). In its purest form economic theory suggests that under ideal conditions economic agents consider all relevant future costs associated with the use of matter and energy and act rationally such that their choices of actions are in accordance with a complete set of current and future markets. Prices include all relevant information regarding the availability of materials and energy, direct their optimal allocation and induce the introduction of substitutes and the development of new technologies. Since substitution between input factors is assumed to always be possible, the scarcity of energy and materials is just a relative matter. Hence, the decision maker's problem is merely to adjust the optimal allocation of resources. Here, I have not forgotten that during the last few decades economists have put great efforts into the relaxation of assumptions that are necessary to describe and analyze economy-environment interactions. Nevertheless, physical interdependencies only receive attention if they are associated with prices and costs, which are based on certain assumptions about the interdependency of the economy and the environment. The different assumptions about the flow of energy and materials across the boundaries for the systems under investigation are recognized differently among disciplines. From an ecological economics point of view the problem arises where the different disciplines overlap but where relevant insight of other disciplines are not recognized.

Production is essentially a transformation of matter and energy and is viewed as a starting point in economic activity (Ayres 1978). Following from this, the natural system is the ultimate source of all material inputs for the economic subsystem. An important issue within ecological economics is therefore the recognition of the limits on both the regenerative and the assimilative capacities of the natural ecosystem arising from the physical laws that govern the energy and matter transformation (Georgescu-Roegen 1993). The implication is that natural resources cannot be conceived as boundless. In addition, since all transformations require energy for which there is no substitute ecological economics elevate the importance of energy resources to the economic processes and the ecosystem as a whole (Odum and Odum 1976; Costanza 1980; Mirowski 1988).

Instead of focusing on the substitution of factors, ecological economics stresses that the depletion of natural resources cannot be resolved through endless substitutions of labor and capital for natural resources. This means that the 'optimistic' technological assumptions often applied in neoclassical models are challenged by the complementarity of factors and the laws of thermodynamics. Ecological economists are therefore concerned with the scale of the human economic system relative to the global natural ecosystem (Daly 1992). Biophysical limits to economic growth must be recognized since growth in the economic system is bounded by a non-growing and finite ecological sphere (Daly 1996). It follows from this that three specific elements should be considered in ecological economic studies. First, the performance of an economy should not be evaluated by efficiency considerations alone. Both distributional and ethical concerns of both intertemporal and intergenerational varieties must be included (Daly 1973). Second, since the economic and ecological systems are interwoven, economic problems should be analyzed using a system framework with an interdisciplinary focus in contrast to the standard static or comparative static equilibrium analyses (Norgaard 1989; Costanza et al. 1993). Third, long-term economic assessments of environmental and resource problems must consider the existence of uncertainty because irreversible processes are found in the interactions of complex systems (Arrow et al. 1995). This warrants precaution in introducing technology and species, pollution control measures and protection for threatened or endangered ecosystems and habitats.

Decisions on resource use concern the future as well as the present, and we cannot know the future with certainty. Equally important the idiosyncrasies and complexities within human-made and natural systems together with heterogeneous individuals and surroundings make it necessary that assessments of environmental problems must deal with uncertainties. A well known example is the greenhouse gas problem. An example with local effects are the continuous sedimentation of toxic materials on the sea floor that can reduce the stock pollution, where the uncertainty often lies in the large differences to the degree this will happen at different locations.

If probability distributions can be derived, we deal with environmental risk, whereas if the assignment of probabilities to all states is not possible, we are dealing with uncertainty in the Knightian sense (Knight 1921). In the absence of probability distributions, uncertainty forces us to make decisions on a somewhat less 'objective' basis. In addition to the suggestions found within decision theory we might apply precautionary principles in our decision making. Whenever we are unable to base our decisions solely on mathematically derived criteria and the consequences of our choices are irreversible or implies large costs, we must err on the side of caution. The degree of cautiousness is something that boils down to the consideration of our responsibility for the stewardship of the earth.

The way uncertainty has been played out in the science and policy arenas is different (see for example Kinzig et al 2003). Science is built on the goal of advancing knowledge and each advance is built on knowledge acquired earlier. Hence, the cost of incorrect knowledge is therefore high since it affects not only the foundation of current knowledge but also that which will follow. The evidentiary standards appear therefore to be relatively high as seen by using significance levels of 1% and 5%. In contrast, the policy process addresses societal ills or challenges and timeliness is consequently an essential factor. Action must sometimes precede knowledge when errors to be avoided are associated with undue social costs, national security or potential environmental catastrophes. Scientists can help illuminate the trade-offs within complex environmental problems with numerous plausible solutions leading to numerous possible futures but there is nothing objective about valuing environmental protection over economic growth. Ultimately we must have a debate in society about values and how we should approach issues related to uncertainty.

The issue related to uncertainty is analyzed in paper 4 by means of a safety rule as part of a precautionary approach to the problem of trading off increased health risk for increased consumption levels. Behind the safety rule is an idea of specifying risk as a function of the pollution level, which is constrained to remain below a given maximum allowable level within a given margin of safety (Lichtenberg and Zilberman 1988). That means that the regulators must decide on the maximum level of health risk that we are willing to accept and how certain we would like to be that this upper limit is not exceeded. The logic is that as long as we are short of information about the true costs associated with some external effect we can never find and implement the optimal level of pollution. The appealing feature of the safety rule approach is that it reflects the challenges within practical politics, namely that regulators must balance social cost against health risk related to some margin of safety. The contribution made from this analysis is that the implementation of a safety rule may lead to an increase as well as a decrease in steady state consumption levels. A more alarming result is that non-linear decay functions combined with uncertainty about health risks can generate a scenario where a steady state or an efficient pollution control path does not exist, which substantiates that some kind of safety measure must be applied in the presence of uncertainty.

Another matter where according to ecological economists the insight from other disciplines is not adequately taken into account is the issue of sustainability and economic growth (Daly 1987). Within orthodox economics sustainability is often defined according to the Hartwick-Solow approach, that is, in terms of maintaining a constant real consumption over an indefinite period of time while recognizing constraints imposed by a given set of resource endowments. The critical assumption is that natural and human-made capital are substitutes, whereas ecological economists view them as complements, which also mean that there are limits to economic growth[4]. In addition it is argued that the Hartwick-Solow conceptualization of sustainability is incomplete since it only refers to the sustainability of an economic system within an anthropocentric perspective. That is, sustainability must include not only the instrumental but also the intrinsic value of other species than *Homo sapiens*. Further, the preservation of sub-human species is a public good, like provision for the distant future, which must be served by collective action (Bentham 1970; Daly 1987). Many ecological economists tend, however, to often lean on an argument based on instrumental value, namely that the sustainability of an economic system is linked to the ecological system, which is not fully acknowledged within neoclassical economics (see Costanza 1991, paper 1). Moreover, since the economic system is a subsystem of the greater ecological system, it is argued that sustainability corresponds to a situation with a nondeclining amount of natural capital and the maintenance of ecological resilience[5].

Among the counterarguments to the view of Daly and others is the belief in the environmental Kuznets curve (EKC), which basically states that environmental degradation is an inverted U-shaped function of income per capita (see for example Grossman and Krueger 1991; Shafik and Bandyopadhyay 1992; Seldon and Song 1994; Cole et al. 1997; Stern 2001). Some theoretical explanations for the existence of an EKC are i) an increasing scale of production implies expanded production for given factor-input ratios, output mix and state of technology, ii) changes in the input mix towards using relatively more environmentally friendly inputs, iii) improvements in technology leads to using less input per unit output and less pollution being emitted per unit of output. Underlying these variables are changes in environmental regulations, awareness and education in the course of economic development.

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[4] Growth is defined here as the quantitative increase in the scale of the physical dimensions of the economy.

[5] Ecological resilience is defined as the ability of the ecosystem to withstand shocks. For more precise definitions see Perman et al. (1999) paper 2.

Besides the methodological criticism that the EKC has been exposed to there are important theoretical arguments that should be mentioned. First of all, a total decoupling between economic growth and environmental degradation is ruled out by the laws of thermodynamics. Despite a decline in the level of many pollutants due to increasingly stringent environmental regulations and technical innovations we have seen increases in other types of pollutants. What lies behind an apparent EKC is often that the mix of effluents has shifted (see e.g. Suri and Chapman 1998). Second, trade plays an important role in the location of industrial production. Assuming free trade, the Heckscher-Ohlin trade theory suggests that developing countries would specialize in the production of goods intensive in labor and natural resources, whereas the developed countries specializes in human capital and manufactured capital intensive production. It is possible that it is this division of work that shows up as an EKC relationship (Lucas et al. 1992; Hettige et al. 1992; Suri and Chapman 1998). The EKC is discussed more in paper 5 where we investigate the relationship between the change in the level of wilderness in Norway with economic growth and economic activity. If the amount of area without human encroachment is used as an indicator for the level of biodiversity we find that the higher the level of economic activity the lower the level of biodiversity, and hence, no support for any EKC relationship. We are aware of the problems related to using the wilderness area as a proxy for biodiversity. Obviously, the level of fragmentation and the type of encroachment are important aspects related to the quality of the wilderness land as habitats.

1.4. The need for an interdisciplinary approach

Economics is a social science that lies at the interface between the natural sciences and the humanities. The emphasis on mathematical formalism in modern neoclassical analyses must be used together with knowledge about the motivations behind the decisions of consumers and entrepreneurs. Within environmental economics the required knowledge of natural science must be greater than in the discipline of economics as such, since the subject matter is the relationship between economic activity and the natural world.

The relationship between human activities and nature is far from simple. On the contrary, ecosystems are interrelated and the interdependencies between nature and economy are complex. Hence, the solution to most environmental problems requires an interdisciplinary approach that can incorporate knowledge about ecological systems, in a wide sense, into analyses of economic systems. A prudent researcher should therefore never 'assume things away' or overly simplify his or her models, but carefully examine the problem under scrutiny and base models on as much of the relevant information that is available. Impacts on the results of the analysis of choices regarding system boundaries must be discussed and simplifications must be justified. Instead of aiming for the 'perfect' model incorporating every aspect of the problem investigated it probably will be more efficient to combine methods. In our opinion this will be the efficient strategy to perform analyses that try to meet some of the criticism articulated against conventional economic analyses by ecological economists (see paper 3). This is the point of departure for the discussion in this section where I focus on the problem of solid waste management.

The two most widely applied methods, Life Cycle Assessment (LCA)[6] and Cost Benefit Analysis (CBA), used in combination with input-output models provide comprehensive environmental profiles for products or product systems. However, the resulting information only gives a static picture of reality and there are two elements that make these approaches insufficient for analyzing recycling systems. First, an encompassing analysis must incorporate changes in the mix of input factors and the effect of different objective functions. Changes in the mix of input factors in the production processes occur as a result of changes in market conditions, technology, and the objective functions. Consequently, the environmental profile of a product system is changed, but the change and its impacts in general cannot be assumed to be linear. Second, mass balance conservation plays an important role in determining efficiency within systems where the aim is to efficiently recycle materials. The first law of thermodynamics should be modeled explicitly in order to capture the 'dynamics' of the recycling system, which works through the marginal costs in each production process. Two other important elements must be considered. First, the optimization of processes change the flow of material and energy in the system, and second, the law of mass conservation together with declining productivity of input factors implies that the cost-curve in each process is dependent of the output in upstream processes. Efficiency hence depends on the objective function of the system planner and on the interdependencies between the processes within the system. This is the background for the use of material balance conditions in papers 2 and 3.

[6] The aim of the method is to specify all environmental impacts of products or services throughout a products life cycle, i.e. from cradle to grave, from raw material acquisition through production, use and recycling, recovery or disposal (Udo de Haes 1996). Normally when carrying out an LCA for a recycling system, the system borders include all flows from the raw material source (upstream-system border) to households to where the material is recovered into new products or energy (downstream-system border) (Finnveden 1999). Moreover, it links changes in the economy to impacts within the environment by studying different options to supply a given function. An example of a functional unit relevant for the analysis in our study is handling and recycling of 1000 kg recyclable used plastic packaging generated in the households.

Over the years several studies have been carried out including a variety of tools for assessing environmental and economic efficiency. A review of the most frequently applied methods is offered in paper 3, which, together with paper 2, is where I derive an approach combining economic analysis with the material balance principle that is applied to the system for recycling household plastic packaging waste in Trondheim.

2. SUMMARY OF THE ESSAYS

Paper 2: Efficiency in a waste treatment system: a material balance approach

Whereas the second law of thermodynamics deals with the quality of energy, the first law of thermodynamics comprises the laws of conservation of mass and energy. Mass and energy balances can be established in which the outputs of mass and energy from a system are accounted for by inputs and changes in storage. From these accounts, the composition of waste streams can be deduced by subtracting the mass of desired outputs and storage from the known inputs into a system. This is the principle that is used in Material Flow Accounting (MFA).

Due to the laws of thermodynamics environmental degradation in some form is a normal and inevitable part of economic activity. The joint production between the wanted product and undesired pollution should therefore be included in analyses of efficient strategies for waste treatment. My analysis is an attempt to combine the MFA approach with the toolbox of an economist. Doing this in a waste treatment system for energy recovery and material recycling I am able to evaluate the system for different levels of waste that has to be processed, which is not the case when using conventional cost-benefit analysis. In addition to identifying efficient combinations of the two waste treatment alternatives my approach enables process interdependencies and the material balance links between production levels and emissions to be identified. From this we can make sure that the suggested solutions are technically feasible.

Besides the aspect of material balance another contribution from my analysis is to point to the important role that the rate of household sorting plays in municipal recycling systems. In 2002, each citizen in Norway generated 354 kg of waste, an increase of nearly 50% from the level in 1992, which shows the magnitude of household waste generation. Not only are households producers of large amounts of waste, they are also located early in the life cycle of the waste and therefore play an important role when it comes to the fate of the waste. Moreover, it is the households that set the upper limit for the recycling rate, since we cannot recycle more than what households have sorted out for recycling. This aspect is left out of most analyses of recycling systems.

In most western countries the authorities have some legislation with requirements about how much waste that should be recycled and how much should be energy recovered. In September 1995 voluntary agreements were signed by the Norwegian Ministry of the Environment (MD) and various industry sectors. The agreements were designed to ensure waste reduction and increase the collection and recovery in the packaging chains. More specifically, the agreement between MD and the plastic-packaging industry states that 80% of the plastic-packaging waste is to be recovered, with a minimum of 30% going to material recovery (Eik 2002). In 2000, 78% of the plastic was recovered, of this, 19% was recycled to new products and 59% was energy recovered. The scientific grounds on which the targets within the agreement were determined are weak. It is very difficult to argue for general recycling targets due to the heterogeneity of the recycling systems found in Norway with respect to both organization of the system but also to the amount of waste that must be processed. A conclusion that can be drawn from my analysis is that there probably will be great variation between municipalities with respect to the efficient recycling rate.

Other results from my analysis are that large amounts of material throughput is neither a necessary condition for securing recycling as a dominant part of an efficient solution, nor is it sufficient to ensure high levels of recycling. A further result is that technical improvements in the recycling process do not necessarily increase the efficient rate of recycling. Consequently, when the authorities instruct recycling plants to implement new technology we might see a reduction in efficient recycling rates since new technology often involves an increase in economic cost which reduces the efficient level of production at the recycling plant.

Paper 3: An empirical assessment of solid waste management: recycling of household waste in Trondheim

Different studies using various methods for assessing the environmental and economic aspects of waste handling systems often arrive at contradictory conclusions when it comes to suggesting efficient waste handling strategies. Much of this discrepancy can be explained by subjectivity in developing, choosing and applying the methodology used to carry out such analyses (Hertwich 2000). As it generally is a very difficult task to develop a method and perform an analysis that is 100% objective, I compiled a brief review of the most frequently applied methods for the assessment of environmental efficiency.

With the review of assessment methods as a backcloth I have estimated the efficient level of recycling of plastic packaging waste generated by households in

Trondheim, Norway's third largest city, using a material balance approach similar to the one developed in paper 2. The alternative way of processing the waste is to produce district heating by incineration of the waste. After the households have separated a fraction of the total amount of plastic waste to recycling it is transported to a central sorting plant. Further processing of the sorted material into recycled granulate is done at three different locations, one in Norway and two in Sweden, depending on the type of plastic that is to be recycled. The incineration and production of district heating takes place in Trondheim.

The novelty of the presented analysis is that production functions and the flow of materials through the system are explicitly modeled and used as a basis for an economic and ecological assessment of the system. The local health effects for two types of emissions are analyzed: NO_X and dioxins. These are modeled both as linear and convex damage functions. Global environmental impacts are represented by the emission of CO_2. Including only three types of emissions obviously limits the validity of the results from my analysis. However, we are able to point to important interdependencies among the processes and the results are consistent with other studies that have been made of the same system.

Our results indicate that producing district heating from the plastic packaging waste is the most efficient solution from an economic point of view. On the other hand, the negative environmental impacts are lower when the waste is recycled into new products. The latter result is turned around if the latest incineration technology is implemented which almost eliminates emissions of dioxins. Which of the waste treatment methods, or mix of them, that turns out to be the most efficient also depends on the total level of waste entering the system. In addition we find that whether recycled materials actually replace products made from virgin material, whether the energy that the incineration of plastic replaces would otherwise be produced by oil, coal or hydroelectric power, and whether we include household costs or not are important when it comes to the determination of the efficient mix of the two treatment alternatives

Paper 4: A safety rule approach to pollution control

It is not possible to assess on a 100 percent objective basis the total cost of CO_2 emissions or the long-term genetic damage of fertilizer run-off from agricultural farm land on fish stocks exposed to these substances. Consequently, we do not know what costs this will entail for present and future generations. Uncertainty in combination with large potential costs means that we are dealing with classes of environmental problems where adverse outcomes may occur, i.e. these problems are not suited for complete reliance on the expected utility approach. Nevertheless, based on available information decision makers must make two decisions. First, the maximum allowable level of risk must be decided and second, the degree of risk exposure we are willing to undertake must be determined.

In order to handle situations with a possibility of adverse outcomes Lichtenberg and Zilberman (1988) propose a regulatory safety rule, which is analyzed within a static model where the goal is to choose the efficient mix of regulatory activities. The idea behind the safety rule is specifying health risk as a function of the pollution level, which is constrained to remain below a given maximum allowable level within a given margin of safety. My contribution is to examine the impact of such a regulatory safety rule in a dynamic model of pollution accumulation and control. Pollution accumulation and control is modeled as one where a benevolent regulator allocates a fixed flow of resources to consumption and pollution control. How the safety rule affects efficient paths and steady states is shown under assumptions of both the linear and quadratic natural assimilative capacity of the environment. I found that the implementation of a safety rule may lead to an increase as well as a decrease in steady-state consumption levels. A more alarming result is that non-linear decay functions combined with uncertainty about health risks can generate a scenario where a steady state or an efficient pollution control path does not exist.

During the analysis it became clear that safeguarding against adverse outcomes carries a cost in terms of foregone consumption. Moreover, trading off consumption against safeguarding against adverse outcomes can be compared to paying an insurance premium. Obviously this trade off depends on the assimilative capacity of nature. I also show that the cost related to subordination under a safety rule is affected by the variables in the model such as the discount rate and technology improvements.

Technologies improve over the years, both with respect to production and abatement efficiency. Nevertheless, we are far from an economy with zero emissions. Moreover, due to stocks of hazardous material already built up, discharges today will have negative impacts on welfare in the distant future. Thus, the main policy implication from paper 4 is that some kind of safety measure must be applied in the presence of uncertainty.

Paper 5: Economic growth and land-use changes: the declining amount of wilderness land in Norway

At the beginning of the 20^{th} century half of Norway's total area consisted of areas more than 5 km from the closest human-made encroachment. Using this definition of wilderness land the proportion had declined to 34% in 1940 and to only 12% in 1994. Wilderness land amounts to only 5% in the southern part of Norway and is absent in three of 18 counties (excluding Oslo). It is important to monitor the amount of untouched land since it has amenity and recreation values in addition to those related to biodiversity and the health of ecosystems. Another important reason for such monitoring is that, compared to other countries in Europe, Norway still has a large amount of wilderness land and untouched nature thereby represents an international public good value. Since the total value of wilderness land cannot be suspected to be included in the market prices of land conversion, it needs to be regulated. In Norway this is done at the local level by "Plan og bygningsloven" ('The Planning and Land-use Act'), and at the national level by "Naturvernloven" ('The Natural Preservation Act').

In 1996, 6.4% of the total land in Norway had some form of protected status. In spite of this, there has been an increase in encroachments which consists basically of road constructions, hydropower projects and electric power lines. We find a number of sectors behind the activities leading to these encroachments. The agricultural and forestry industry and hydropower industry represents the majority of them. The aim is however not to analyze the land-use changes at the sector level, but rather to consider the underlying causes. This is done in an empirical analysis that explains the reduction of wilderness land in Norway by macroeconomic factors.

The regressions are carried out as cross-section models as well as pooled, fixed effects models at the county level (18 counties) for the years 1988 and 1994. The wilderness area in Norway is categorized into three qualities based on distance from larger technical installations; land as more than 5, 3 and 1 km from closest human encroachment, respectively. The explanatory variables comprise GDP per capita, GDP per capita squared, and population density. The main finding from the cross-section analysis is that the wilderness land as a fraction of the total area within each county is lower the higher the level of economic activity, as measured by GDP per capita. However, the fixed effects models suggest that there is a negative relationship between economic growth and the reduction of wilderness land. These effects are tighter for wilderness land defined within a short distance from existing encroachments.

The results of recent empirical studies on land-use and economic growth are mixed. Some studies on deforestation have found evidence supporting an environmental

Kuznets curve, but two qualifications must be made (see for example Cropper and Griffiths 1994). First, the loss of a species is irreversible. A reduction in the deforestation rate may therefore not imply that we are moving closer to a sustainable development since the irreversible loss in biodiversity already has reduced the stability of the ecosystem. Second, an inverted U-shape for the growth in deforestation may not be consistent with sustainability since the level of forest cover still is declining. We have shown that a high level of economic activity and high economic growth per capita is associated with less wilderness land. If we believe that the remaining amounts of wilderness are important for the health of our ecosystems, the study gives no support to a hypothesis that prosperity is a sufficient condition for a sustainable use of our resources.

3. CONCLUDING REMARKS

In this dissertation an attempt has been made to widen the scope of economic analyses. This is done in papers 2 and 3 by incorporating material balance principles into a cost-benefit analysis framework, in paper 4 by analyzing pollution control under a rule of precaution, and in paper 5 by investigating the relationship between economic growth and change in the amount of wilderness land in Norway. The widening of scope is limited to some of the elements pointed to as important for sustainability by advocates of the subparadigm ecological economics. Other elements could have been included such as the inter-industry impacts on a macro level or issues related to the design of waste handling systems.

An intriguing aspect related to the issue of sustainability is the ethicosocial limits to growth. Daly (1987) argues that in addition to the biophysical limits to growth one must also be aware that "the forces propelling economic growth are simultaneously eroding the moral foundations of the very social order which gives purpose and direction to that growth" and that "the pursuit of "infinite wants" leads to a weakening of moral distinctions between luxury and necessity" (Daly 1987, p. 335). Another argument is that the cost imposed on future generations by the running down of the resources that are necessary to finance economic growth will limit the desirability of further growth. This comes in addition to the self-canceling effects on welfare from economic growth caused by the argument that happiness is a function of relative income; if some people's relative income goes up the income of others must go down (Hirsch 1977).

Although the economy approaches its limits to growth, ecological economists argue that we are far from the limits of development, the latter being defined as the qualitative improvement in the structure, design, and composition of physical stocks and flows resulting from an increase in knowledge, both in terms of technology and purpose. The important question may therefore not be how we can secure future economic growth, rather how we can explore the goal of our efforts.

This dissertation includes analyses that have limitations beyond the mere selection of elements from ecological economics that I have made, namely the assumptions underlying our models. Looking into how the analyses can undertake more general problems is one direction for further research. Tackling the issue of the goals and development of society is another, probably more difficult but also potentially more interesting and rewarding one.

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Paper 2

Efficiency in a waste treatment system: a material balance approach

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Abstract

Due to the laws of thermodynamics environmental degradation is in practice a normal and inevitable part of economic activity. The joint production between the wanted product and undesired pollution is therefore a key feature of the analysis, which focuses on efficient strategies for waste handling. By applying a material balance principle to a generic waste treatment system for energy recovery and material recycling, new insight has been developed regarding efficient combinations of these two waste treatment alternatives. The main results suggest that (1) technical improvement in the recycling process does not necessarily increase the efficient rate of recycling, (2) a large amount of material throughput is neither a necessary condition for securing recycling as a dominant part of an efficient solution, nor is it in itself sufficient to ensure a high level of recycling and (3) that when the authorities instruct recycling plants to implement new technology we may see a reduction in efficient recycling rates as new technology involves an increase in economic costs which reduces the efficient level of production at the recycling plant. It is also demonstrated that inclusion of the household sector is crucial for understanding important bottlenecks and interrelations between processes within recycling systems.

Keywords: recycling, energy recovery, material balance, waste handling, household sorting.

1. INTRODUCTION

For several decades we have seen a debate about what the efficient strategy for waste handling should be, how policy measures affect the optimal strategy and ultimately what policy measures would be the optimal to implement. Since there are a variety of different assumptions that have to be made regarding the type of waste that is under scrutiny, the institutional setting, upstream vs. downstream instruments, etc., the literature on this subject has become extensive (for a review, see Choe and Fraser 1998; Goddard, 1995). Economic analyses including and focusing on the role of natural laws, such as the laws of thermodynamics, are not seen to the same extent, however, based on the fact that environmental externalities are normal and, in practice, inevitable parts of economic activity, and that processing cannot destroy waste discharges only alter its form, it is important to trace residual flows in the economy (Ayres and Kneese 1969). It is vital to map and try to forecast the duration and concentration of residuals, not just because intensive economic and population development makes it increasingly important, but also because including mass balance conditions is an important step in ensuring that economic models are consistent with real life situations (Ruth 1999). Identifying flows of substances that potentially could lead to future environmental damage is undoubtedly becoming increasingly important. This is the motivation behind the following analysis which puts more emphasis on the impact of joint production aspects associated with environmental impacts from economic activity than on the choice of specific policy measures.

The analysis of a household waste treatment system presented in this paper is based on a combination of a cost benefit analysis and material flow accounting (MFA). MFA is a method for specifying the flow of materials into, through and out of a nation, a region, a business sector, company or a household for a given period in time (Wriesberg and Udo de Haes 2002). It is a robust tool since one can link various material flows through a variety of different processes and it is therefore also a good basis for dynamic analyses of future scenarios (see for instance Kleijn et al. 2000). Other examples of formalized descriptions of material flows through the economy are Ayres (1978), Gilbert and Feenstra (1992), Van den Bergh and Nijkamp (1994) and Weaver et al. (1997). However, none of these studies considers explicitly the link between material flows and economic behavior and products.

An overview of the limited number of studies that combine economic models with material balance principles at a macro level is found in Ibenholt

(2002). The studies make use of mainly two approaches, either fixed coefficients to indicate the relationship between economic activities and physical outputs, or calculating emissions as the residual between input and output in the economy. At a more micro level, the studies closest to the analysis presented here are Kandelaars and Van den Bergh (1996) and Baumgärtner and Jöst (2000), both of them using an economic model expressed by explicit material balances. Whereas Baumgärtner and Jöst (2000) emphasize the fact that negative environmental impacts and production of the desired product are joint products, Kandelaars and Van den Bergh (1996) focus on substitutability between input factors and price determination. The contribution from the paper presented here is found in the modeling of the environmental externalities as joint products of the production processes, the investigation of the social regulator's possibilities for varying the production at different stages in the materials-product chain, and the impact on the efficient recycling rate of changes in the technological efficiency.

The paper analyzes a waste handling system for varying levels of household waste. Two waste treatment processes are considered, namely energy recovery and material recycling where a regulator, typically within a municipality, maximizes social welfare[1]. A situation where the regulator has no control over the household sorting efforts is also investigated. The first purpose of this analysis is to provide insights about the efficient *mix* of these processes. Second, this will illustrate the role of material balance within a cost-benefit analysis approach, i.e. stress the link between production and emissions which is based on the first law of thermodynamics. At the same time this can assure that the solutions suggested by these analyses are technically feasible.

The material under scrutiny enters the system through waste from households' purchase of goods. Typically, this waste is recyclable and in the form of packaging waste. Within this defined system the waste is later used as an input in the production of district heating or recycled material. The material entering the system is traced through the waste handling system and this material flow is the foundation for the assessment of environmental impacts within an economic model, which is similar to a cost-benefit analysis. The system is first analyzed with general functions before the material flow is formulated explicitly for more clear-cut results. Compared to other methods for the assessment of economic and ecological efficiency such as conventional cost-benefit analysis and life-cycle

[1] Landfill is not included as a viable option for waste treatment since in many countries it is prohibited by law and many studies show that land filling is the least efficient alternative for waste treatment especially in densely populated areas (EU 1999; CIT Ekologik 1999).

assessment, the approach enables analyses of varying levels of waste processing[2]. What is more, such an approach opens the way for a more encompassing analysis, including issues such as the identification of efficient degrees of centralization of waste processing plants. The result is a transparent method that enables the study of issues such as the effect of decisions made in one place on the material flow elsewhere in the system and the joint production between the wanted product and undesired pollution. The method can also consider the impact of increased technical efficiency and increased amounts of waste input to the system.

Although some assumptions for simplification are made, the present analysis produces interesting results that are valid for a variety of waste handling systems. For example, due to mass balance constraints, the system manager must take into consideration the important role of the households. Their level of sorting can be crucial for the implementation of the efficient solution due to their position in the life cycle of the material. Moreover, the maximum level of resources that should be used to motivate increased household sorting of waste is identified. Further, economic growth will, without a significant decoupling of resource use, lead to increased amounts of waste. From this increased pressure on the ecological system, and the following change in dose-response relationships, will lead optimal recycling levels to be based on the environmental efficiency of each of the processes to an increasing extent. Another interesting result is that the implementation of improved technology in the recycling sector can actually lower the efficient recycling rate.

The next section presents a general model for handling household waste where the alternatives for waste processing are material recycling and energy recovery. In order to look beyond the general model production in the material recycling and energy recovery sectors are specified as fixed coefficient production functions in Section 3. However, considering the system as a whole substitution between input factors are possible. Hence, from a system regulator's point of view the production functions can be characterized as homothetic. This way of describing production is relevant for a variety of production processes where substitution among input factors is limited but where the processes are interrelated. Increasing marginal damage is applied to reflect that an increasing level of emissions will lead to overstepping of thresholds in the receiving systems, human and ecological. A summary and conclusions are offered in Section 4.

[2] For a brief introduction to life cycle assessment (LCA) see Wrisberg and Udo de Haes (2002).

2. A GENERAL MODEL

The two most widely applied methods, Life Cycle Assessment (LCA)[3] and Cost-Benefit Analysis (CBA), in combination with input-output models offer encompassing environmental profiles for products or product systems. However, the resulting information works only as a static picture of reality and there are two elements that make these approaches inadequate for analyzing recycling systems. First, an encompassing analysis must incorporate changes in the mix of input factors caused by varying market conditions, technology, and objective functions. The environmental profile of a product system and its ecological and human health impacts in general cannot be assumed to be linearly correlated with the level of production. Second, mass balance conservation plays an important role in determining efficiency within systems since changes in the flow of material and energy in the system, together with declining productivity of input factors, implies that the cost-curve in each process is dependent on the output in upstream processes. Hence, efficiency depends on the objective function of the system planner and the process interdependencies. The first law of thermodynamics should therefore be modeled explicitly in order to capture the interrelations between processes in the recycling system, which works through the marginal costs in each production process (see Eik et al. 2002).

2.1. Model specifications

The model consists of an energy recovery and material recycling system with a benevolent regulator, where households must sort their waste into a recyclable and non-recyclable fraction. The material that the household sector has identified as recyclable is later transported to the material recycling plant, while the unsorted waste goes to incineration in the energy recovery plant. At the material recycling plant the material is processed further, and the fraction of the material input that is not recycled is transported to the energy recovery plant. Externalities are found only in the recycling and recovery processes as any rinsing of the waste is assumed to be done without the use of hot water. Transportation costs are ignored for analytical convenience. The system is illustrated in Figure 1 below.

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[3] The aim of the method is to specify all environmental impacts of products or services throughout a products life cycle, i.e. from cradle to grave, from raw material acquisition through production, use and recycling, recovery or disposal (Udo de Haes 1996). Normally when carrying out an LCA for a recycling system, the system borders include all flows from the raw material source (upstream-system border) to households to where the material is recovered into new products or energy (downstream-system border) (Finnveden 1999). Moreover, it links changes in the economy to impacts within the environment by studying different options to supply a given function. An example of a functional unit relevant for the analysis in our study is handling and recycling of 1000 kg recyclable used plastic packaging generated in the households.

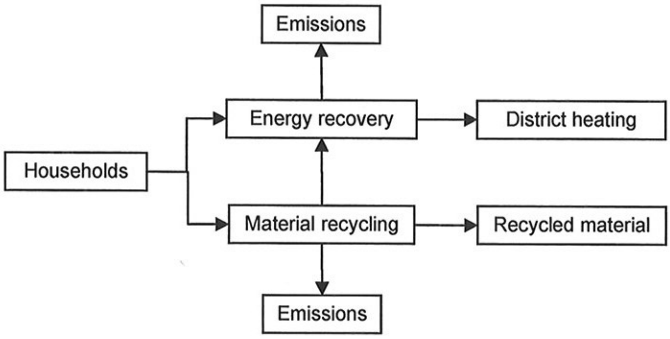

Figure 1 Overview of the material flow through the waste handling system.

Production in the system is based on two input factors; household waste and labor, and that decisions are made by the regulator. Let q_0 be the exogenously determined amount of waste that enters the system and must be processed. L denotes labor input in the production processes and q_i, where $i = H, M, E$, denote the three production sectors within the system, which are the household-, the recycling- and energy recovery sectors, respectively. Parts of the material input in the system ends up as emissions. In the model E_E and E_M are emissions from the energy recovery and material recycling processes, respectively. In two of the three sectors, namely the household and recycling sectors, the use of the two input factors waste and labor must be determined, whereas the energy recovery sector processes the residual amount of waste from the other processes and merely adjust the use of labor. We therefore define the following concave and continuous production functions:

$$q_H = q_H\left(L_H, q_0\right) \tag{1}$$

$$q_M = q_M\left(L_M, q_H\right) \tag{2}$$

$$q_E = q_E\left(L_E, q_0 - q_M - E_M\right) \tag{3}$$

which all exhibit positive and diminishing marginal productivity. The production processes in the system are linked together by the flow of mass through the system since there is a limit for substitutability between the input factors. Hence, production in the household sector, which is amount of material sorted for material recycling, affects the production of recycled material in the whole system, and may therefore limit the maximum output that can be produced from the system. Consequently, the production in the energy recovery sector is also influenced by the decision in the households since the input in this sector is the mass which is not recycled. In the end, total environmental damage is affected by decisions in the households, and next, in the material recycling sector.

The first law of thermodynamics states that the amount of materials and energy is constant within a system, which implies certain conditions for economic analyses (see Ruth 1999). Within the processes in our system the substitution of material for labor is limited to the fraction of waste that the households send to recycling. Moreover, this is the upper limit for the total recycling rate because we cannot recycle more material than the households send to recycling. Moreover, for the system as a whole the following conditions must hold: $q_H / q_0 \leq 1$, $q_M / q_H \leq 1$, and $q_E + E_E = q_0 - q_M - E_M$ [4]. Since production in the material recycling sector is bounded by the production in the households, increasing the input of labor cannot help the fact that beyond some point of labor input the two input factors are complements rather than substitutes. This relationship is illustrated in Figure 2 below. The feasible set of input factor combinations must lie below the 45^{O}-line, i.e. we cannot produce more material than we put into the production process. The distance between the curves indicating production levels and the 45^{O}-line denotes the sum of emissions from the process and the amount of input that will be transported to the energy recovery plant.

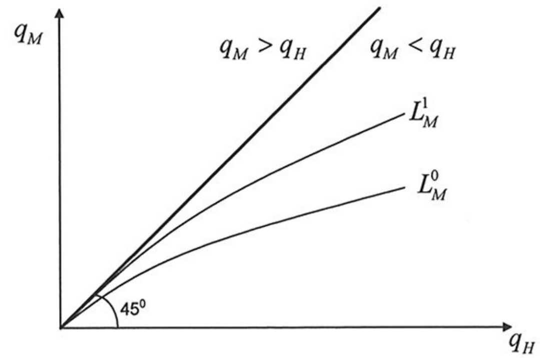

Figure 2 Illustration of possible combinations of production of recycled material, q_M, and input factors labor, L_M, and material, q_H from a material balance perspective. The curves are drawn for $L_M^1 > L_M^0$ assuming diminishing marginal productivity.

The origin of the material in our model is the households' generation of waste. Attempts have been made to identify the relationship between household effort and sorting level but most of the existing studies focus on revealing what motivates or facilitates recycling (see Tasaday 1991; Hornik et al. 1995). Moreover, most studies implicitly assume that there are no costs associated with households' effort in recycling activities, meaning that the net benefit increase

[4] These conditions come in addition to implicit conditions regarding non-negative amounts of materials.

with the household effort. However, in most recycling systems less than 100 % of the potentially recyclable amount of household waste is sorted, which indicates the existence of barriers within the sorting activity (Kinnaman and Fullerton 1999; Eik et al. 2002). Thus, in the absence of legal requirements together with possibilities of sanctions against violators, what lies behind the level of sorting must be some kind of utility maximization.

In order to avoid double counting it is assumed that the environmental gains from material recycling and energy recovery are reflected in their respective product prices and that they are not included in the utility function which later will be defined for the households. Instead, it is assumed that households have preferences for keeping a certain type of self-image which can be derived from contributing to reduction of environmental degradation (Akerlof and Kranton 2000; Brekke et al. 2002), or that individuals obtain "a warm glow of giving" from contributing to public goods (Andreoni 1990).

Whether people obtain "a warm glow" from what is actually recycled or from the level of their own sorting effort put into the recycling activity has not, to our knowledge, been determined by empirical studies. This analysis is therefore based on the assumption that a representative household derive utility from both its own production and the amount of waste which is actually recycled:

$$U = U\left(q_H, q_M\right) \tag{4}$$

It is assumed that the marginal utility is positive and diminishing with the level of sorting efforts and production of recycled material. Any costs related to effort can be justified either as lost leisure or be based on the utility loss from conforming to social norms (Bruvoll and Nyborg 2002). The cost associated with sorting efforts is defined as:

$$C_H = w_H L_H \tag{5}$$

where w_H is the alternative cost per unit time used on sorting. In most recycling systems for household waste, especially when based on curb-side pick-up, there are no significant environmental externalities in the household sector, as is confirmed by empirical studies (Eik et al. 2002). The households simply choose what waste bin to put the waste in and the production in this sector is merely the amount of (correctly) sorted waste[5].

[5] One could of course claim that there are external effects from energy and water consumption due to rinsing of dirty waste or from producing the extra waste bins that may be needed. These effects are assumed negligible in our analysis. Further, if there is a central pick up location it is usually located in or close to grocery stores so the delivery of waste is done together with the purchase of food etc.

Waste handling is usually the responsibility of the municipality. We therefore assume that the municipality is the owner of both the material recycling and the energy recovery plant, and that both the recycler and the energy recovery firm are price takers in their respective markets so that the profit in the system is given as $P_M q_M + P_E q_E - w(L_M + L_E)$, where w is a uniform wage rate. One can argue that the main type of cost in incineration plants is typically capital costs, but at the same time it can be argued that they are sunk costs as the system under investigation is a part of a larger waste treatment system[6].

Emissions from the production processes are generally described as positive relationship between production and emissions for the production of recycled material and district heating, respectively: $E_M = E_M(q_M)$ and $E_E = E_E(q_E)$. The emissions generate environmental damage and health risk according to $D_M = D_M(E_M)$ and $D_E = D_E(E_E)$. Environmental costs are net damage since by assumption this accounts for any avoided damage due to the substitution of products made from virgin material by recycled material and energy produced in oil or coal fired power plants being substituted by district heating made from the incineration of waste. Marginal environmental costs can therefore be both positive and negative depending on the relative size of the environmental cost from the process and the avoided external effects from substitution of virgin material for recycled material.

2.2. The efficient recycling rate and process interdependencies

Households derive utility related to the "warm glow effect" and for keeping a certain self-image as a responsible citizen, and the municipality receives profit from its recycling and energy recovery plants. It is further assumed that the external effects from emissions fall entirely on the inhabitants of the municipality. The municipal social welfare generated by the waste handling system is therefore given as:

$$W = U - w_H L_H + P_M q_M + P_E q_E - w(L_M + L_E) - D_M - D_E \tag{6}$$

The decision problem of the regulator is to determine the optimal levels of production so as to maximize social welfare for the system by controlling the level of material input and labor use in the production processes. From equations (1)-(3) and the restriction on the material balance, it is evident that determination of L_H, L_M and L_E are sufficient for finding the efficient production levels in the

[6] Typically several types of materials are incinerated in energy recovery plants (Eik et al. (2002)).

system. Assuming an interior solution, the first order conditions necessary for the efficient levels of production are given as equations (7)-(9)[7].

$$\frac{\partial W}{\partial L_H}=0$$
$$\Rightarrow \frac{\partial q_H}{\partial L_H}\left(\frac{\partial U}{\partial q_H}+\frac{\partial q_M}{\partial q_H}\left(\frac{\partial U}{\partial q_M}+P_M-\frac{\partial D_M}{\partial q_M}+\frac{\partial q_E}{\partial q_M}\left(P_E-\frac{\partial D_E}{\partial q_E}\right)\right)\right)=w_H \tag{7}$$

$$\frac{\partial W}{\partial L_M}=0\Rightarrow \frac{\partial q_M}{\partial L_M}\left(\frac{\partial U}{\partial q_M}+P_M-\frac{\partial D_M}{\partial q_M}+\frac{\partial q_E}{\partial q_M}\left(P_E-\frac{\partial D_E}{\partial q_E}\right)\right)=w \tag{8}$$

$$\frac{\partial W}{\partial L_E}=0\Rightarrow \frac{\partial q_E}{\partial L_E}\left(P_E-\frac{\partial D_E}{\partial q_E}+P_M+\frac{\partial q_M}{\partial q_E}\left(\frac{\partial U}{\partial q_M}-\frac{\partial D_M}{\partial q_M}\right)\right)=w \tag{9}$$

The efficient level of household sorting, i.e. the share of material input to the system that should be sorted and transported to the material recycling and energy recovery sector, respectively, is determined by Equation (7). As long as the sufficient amount of q_H is available (we will return to the contrary case) the efficient level of production in the material recycling and energy recovery sector are given from equations (8) and (9).

In all three first order conditions marginal social profit is equal to marginal social costs of the labor input in the respective processes. However, a distinct feature of the model is that the production output from the upstream process is an essential material input factor in downstream processes. The interdependency between processes is reflected by that an increase in household effort leads to an increase in production of recycled material, which reduces the production of district heating. This can be identified by investigation of Equations (7)-(9). Combining equations (8) and (9), efficient production of recycled material and district heating is found where the relative marginal productivity of labor in the material recycling and energy recovery processes equals the relative marginal social profit:

$$\frac{\dfrac{\partial q_M}{\partial L_M}}{\dfrac{\partial q_E}{\partial L_E}}=\frac{P_E-\dfrac{\partial D_E}{\partial q_E}+P_M+\dfrac{\partial q_M}{\partial q_E}\left(\dfrac{\partial U}{\partial q_M}-\dfrac{\partial D_M}{\partial q_M}\right)}{\dfrac{\partial U}{\partial q_M}+P_M-\dfrac{\partial D_M}{\partial q_M}+\dfrac{\partial q_E}{\partial q_M}\left(P_E-\dfrac{\partial D_E}{\partial q_E}\right)} \tag{10}$$

Apparently, the household production level accounts for the direct and indirect effect on households' utility, hereby the impact on the social profit from changes

[7] The exposition is simplified by letting $\frac{\partial D_i}{\partial E_i}\frac{\partial E_i}{\partial q_i}=\frac{\partial D_i}{\partial q_i},\ i=M,E$.

in the production in both the energy recovery and material recycling sectors (Equation (7)). Using the information offered by Equation (8) together with Equation (7), the efficient amount of sorting efforts in the household sector can be found as:

$$\frac{\partial q_H}{\partial L_H}\left(\frac{\partial U}{\partial q_H}+\frac{\partial q_M}{\partial q_H}\frac{w}{\partial q_M/\partial L_M}\right)=w_H \tag{11}$$

The first term in the bracket on the left hand side reflects the marginal utility from an increase in q_H, and the second term the total marginal net social profit in the material recycling and energy recovery processes from an increase in q_H. The sum of these elements must be equal to the right hand side which denotes the marginal cost in the households with respect to effort. As we would expect, the amount of sorted waste in the households increases with the marginal net social profit in the downstream processes. Hence, efficiency for the overall system is in general attained when the households take into account the impact of their actions on the rest of the system.

2.3. Economic growth, centralization and waste hierarchy

Unless we are able to decouple the production of waste from consumption, economic growth will lead to increasing flows, and stocks, of waste causing more environmental stress. The increased amount of waste must be taken care of, and how this will change the efficient allocation of mass between recycling and energy recovery depends on the relative change in marginal environmental damage compared to the relative change in household utility and profit within the processing sectors. At this stage it is difficult to state general results beyond this since the various systems for waste treatment are surrounded by different natural environments and demographic structures, and hence, a variety of health risk thresholds. In addition, they might display different economic returns to scale. These factors will jointly determine if the recycling rate will increase or decrease with the amount of waste generated. Additionally, the development of new technology will over time affect which of the available waste treatment alternatives is found to be the most efficient. These issues will be discussed more in Section 3 where a more detailed model is analyzed.

Other aspects of waste treatment policies can also be investigated within our apparatus. By examining how the total welfare in the system is affected by changes in the amount of waste to be processed, it is possible to investigate to what degree it would be wise to centralize municipal waste handling services in larger regions. Generally, welfare will increase as a municipality in which $\partial W / \partial q_0 > 0$ processes waste from municipalities with lower marginal social welfare. Accounting for transportation costs means that increasing distances between where the waste is generated and where it is processed reduces the profit from centralization of waste processing.

With information about the net costs of all policies related to waste problems we could also analyze which of the waste handling strategies would be the most efficient. The results from such an analysis would give valuable input to discussions concerning the much debated waste hierarchy (EU 1999). In Section 3 it is demonstrated that the ranking of waste treatment alternatives may depend on the amount of waste that must be processed, as a linear technology model is specified in order to derive more clear-cut results concerning efficiency in waste handling systems.

2.4. A decentralized decision on sorting efforts

In a more decentralized solution the regulator typically controls only the production levels in the energy recovery and material recycling processes and not the level of sorting done by the households. If the households do not allow for the net social profit from the material recycling and energy recovery processes, the production level in the material recycling sector may be constrained by the access to input material, i.e. we have that $q_H < q_H^*$ and, consequently, $q_M < q_M^*$, where superscript * denotes the efficient solution for the overall system from the regulator's point of view. This implies that from a regulator's point of view, the cost of increasing L_H to a level above what would be efficient from the households' point of view, is warranted by the increase in profit in the recycling sector resulting from an increase in q_H. The regulator should therefore consider allocating resources towards activities that would make households increase their sorting level closer to the efficient level from the regulator's point of view. Examples of such activities are side payments or indirect instruments such as information campaigns to affect their attitude towards waste recycling.

Whenever the households ignore the impact from their efforts on the social profit in the material recycling and energy recovery sectors, their sorting effort is given as the level of effort consistent with equating marginal utility with marginal cost (Equation (12)).

$$\frac{\partial q_H}{\partial L_H}\left(\frac{\partial U}{\partial q_H}+\frac{\partial U}{\partial q_M}\frac{\partial q_M}{\partial q_H}\right)=w_H \tag{12}$$

A subsidy, s^*, must reflect the difference between equations (7) and (12), that is, the difference between marginal social profit in the recycling and energy recovery processes, respectively[8].

$$s^* = \frac{\partial q_M}{\partial q_H}\frac{\partial q_H}{\partial L_H}\left(P_M - \frac{\partial D_M}{\partial q_M} + \frac{\partial q_E}{\partial q_M}\left(P_E - \frac{\partial D_E}{\partial q_E}\right)\right) \tag{13}$$

An interpretation of s^* is that it denotes an upper limit for resources targeted at increasing the household's sorting efforts. Hence, s^* reflects a gain from an increase in household sorting made up of three elements. First, a loss in revenues and avoided environmental costs in the energy recovery sector directly follows a reduction in waste transported to the incineration plant. Second, a larger q_H leads to an increase in the production of q_M, which brings about an increase in revenues and reduction in environmental costs in the recycling sector. Third, a larger production of q_M reduces the environmental costs caused by a reduction in the production of q_E.

It should, however, be noted that the potential increase in household sorting that a side-payment can induce is highly uncertain, and increasingly so, the larger the amount of waste that must be processed. Further, the subsidy would typically be organized as a reduced disposal fee on the residual fraction of the total waste. However, this could lead to an increase in the sorting of other waste fractions than the one targeted by the decision maker. Therefore, the overall effect of the subsidy might be close to zero as it is difficult to design an effective subsidy system. In addition, the cost of controlling sorting in the individual households may be substantial. As demonstrated in Eik et al. (2002), the possible solutions to such problems should generally be implemented as early as possible in the life cycle of the material. An effective measure could for example be to simplify the sorting process by, for instance, standardization of food packaging. Nevertheless, the model points to mechanisms which are important to regulators considering policy measures aimed at increasing the level of household sorting.

3. ANALYZING A SYSTEM WITH FIXED PROPORTIONS TECHNOLOGY

The functions presented in the general model above are now explicitly specified to obtain more insight on recycling systems. This is suitable for the purpose of illustrating the role of the material balance within systems with interrelated processes.

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[8] Note that an alternatively approach could have been to use that $s^* = \dfrac{\partial(W-U)}{\partial L_H}$.

3.1. The efficient recycling rate and process interdependencies

As noted above, the first law of thermodynamics requires that the input to any of the processes must equal the output, which consists of the desired product and emissions. This constraint must be satisfied to ensure that efficient solutions resulting from our analysis are technically feasible (see Figure 2). Further, a purposive way to specify the production functions and their relation to emissions, and hence, external costs, is therefore by using input-output ratios. That is, by combining a cost-benefit analysis with production functions, together with an inventory analysis all material flows are identified. At the same time, a dynamic analysis instead of a conventional static cost-benefit analysis is rendered possible[9].

In the households the judgment that some of the material is not recyclable is based on the degree of fragmentation and traces of food etc. Based on the fact that the level of production of recycled material from household waste is rarely communicated to the households, nor does the majority of households make efforts to gather this kind of information, a simplification is made as the utility associated with the sorting activity is formulated as a function in the amount of sorted material, $U = U\left(q_H\right)$ (see Eik et al. 2002). The cost related to sorting efforts is still given by Equation (5). Maximizing net utility, $U\left(q_H\right) - w_H L_H$, the efficient level of effort, L_H^*, is found where $\left(\partial U / \partial q_H\right)\left(\partial q_H / \partial L_H\right) = w_H$, which implies an efficient level of output: $q_H^* = q_H\left(L_H^*, q_0\right)$. A corollary is an efficient input output ratio: $q_H^* / q_0 \equiv \alpha^*$, where material balance conditions require that $\alpha \in \langle 0,1 \rangle$.

Production in the material recycling and energy recovery sectors are described by linear technology production functions. The choice is made not only because of analytical convenience, but also because it describes processes within waste handling systems well. Moreover, most recycling processes are simple; sorting at the central sorting plants is often done by people standing along a conveyor belt. The sorted material is then grinded or mixed with paper before it is processed into a new product. Hence, we assume that in the material recycling process labor is applied in a fixed proportion to the input of material, where the latter is a fraction β of q_H:

$$\frac{L_M}{\beta q_H} = l_M \tag{14}$$

Hence, l_M is the Leontief input coefficient in the material recovery process. Further, the production of recycled material is described as a function of β and q_H:

$$q_M = k_M \beta q_H \tag{15}$$

---

[9] The analysis is dynamic in the sense that we are able to analyze different levels of material flows.

β is the production decision variable and from the law of mass conservation it follows that $(\beta \in [0,1])$[10]. The parameter k_M indicates the technical efficiency within the recycling process, i.e. the upper limit for input output ratio determined by the technological level of the production process $(k_M \in (0,1))$. The residual fraction, $1-k_M$, is emitted to air and water. The costs in the recycling sector are found from combining equations (15) and (14): $C_M = wL_M \Rightarrow C_M = wl_M q_M / k_M$, where w still is the uniform wage rate[11]. Hence, profit in the recycling sector is given as $\Pi_M = (P_M - wl_M / k_M) q_M$.

The fixed relationship between material input and labor use in the energy recovery sector, l_E, is seen in Equation (16).

$$\frac{L_E}{q_0 - q_M / k_M} = l_E \tag{16}$$

$$q_E = k_E e (q_0 - q_M / k_M) \tag{17}$$

The fixed coefficient relationship between material input and production, which is measured in kWh, is found in Equation (17). Here, k_E denotes the environmental efficiency, and e is the energy content per tonne waste, typically kWh/tonne. Using equations (16) and (17) costs associated with incineration of the specific type of waste can be identified as $C_E = wl_E q_E / k_E e$. Revenue in this sector is generated according to $R_E = P_E q_E$, where P_E is the market price per unit, typically kWh, district heating, q_E. Based on this we formulate profit in the energy recovery sector as $\Pi_E = (P_E - wl_E / k_E e) q_E$.

Given the mass balance conditions we can now describe production of final and intermediate products, and emissions within the system as a function of the material input, q_0, the input output ratios in the household sector (α) and in the material recovery sector (β), which is illustrated in Figure 3. Consequently, to keep track of the material balance conditions it is suitable for the purpose to use α and β as decision variables. Already at this stage we acknowledge that the mass balance conditions imply that the overall material recycling rate is given as $\mathrm{M} \equiv q_M / q_0 = \alpha\beta k_M$, which never exceeds the household sorting rate, the rate of production in the recycling sector or the rate of technical efficiency within the material recycling sector, respectively. As pointed to in the general model this has some specific implications for efficiency considerations and how upstream

[10] We could of course combine equations (14) and (15) and formulate the production of recycled material as a function of labor. This is not done here as the purpose of this analysis is to show the role of material flow as the element connecting the processes together.

[11] Since we assume linear production technology any other input factors besides labor would be applied in the same proportion.

processes can influence downstream efficiency. What is more, the degree to which environmental impacts are joint products of the production processes influences the optimal total material recycling rate, which will be analyzed in subsequent sections.

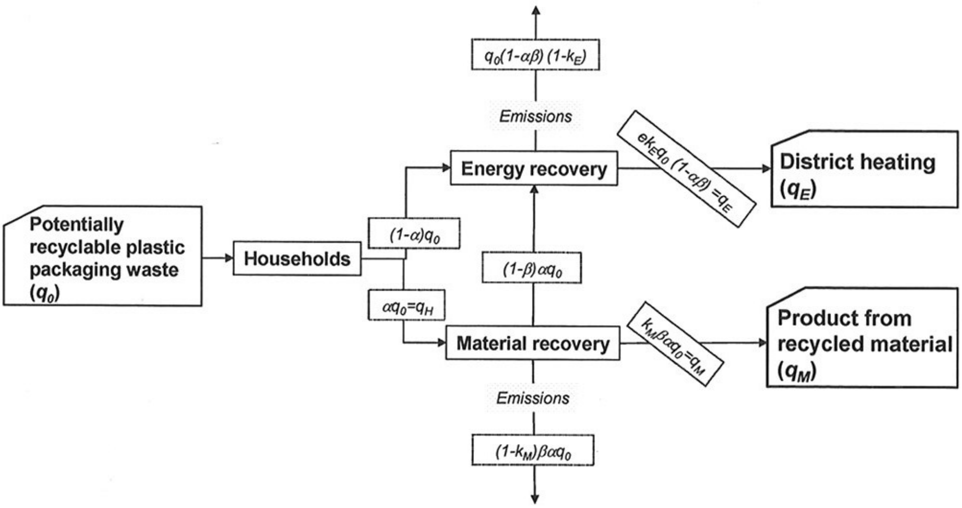

Figure 3 The material flow in a system for recycling and energy recovery of household waste.

Due to the laws of thermodynamics, the production in the energy recovery sector and waste discharges from each of the processing activities are joint products of the production in the material recovery sector and, consequently, from the household sector. Discharges from the processing of the waste generate negative external effects, which are assumed to have local impacts[12]. More specifically, we assume that the external costs can be represented as[13,14]: $D_M = a(1-k_M)\alpha\beta q_0 + \frac{b}{2}((1-k_M)\alpha\beta q_0)^2$ and $D_E = g(1-k_E)(1-\alpha\beta)q_0 + \frac{h}{2}((1-k_E)(1-\alpha\beta)q_0)^2$, where $(1-k_M)\alpha\beta q_0$ and $(1-k_E)(1-\alpha\beta)q_0$ are emissions from the recycling and energy recovery processes, respectively. It is assumed that $a,b,g,h>0$, where a and g is measured

[12] These local impacts can exist in addition to any global environmental impacts.

[13] Notice that the external costs in the energy recovery process are modeled as a function of the input of material, not the amount of produced energy. Further, we have made a simplification here in that it is only the material from the waste input that is included in the damage function. In industrial processes this material must react with substances in the surrounding environment creating some hazardous material, but we assume that the damage can be approximated by measuring the content of waste material.

[14] Convex functions are chosen in order to reflect that increasing levels of emissions increases the likelihood of triggering health effects and hence increases the external cost from the production process.

as \$ per tonne, b and h is measured as \$ per (tonne)^2. The degree of convexity of the external cost functions depends on factors such as the local settlement pattern, emissions from other sources and climatic conditions.

As mentioned earlier, parameters $k_M, k_E \in \langle 0,1 \rangle$ denote technical efficiency within the processes and reflect the joint production of the wanted product and the emissions in the material and energy recovery processes, respectively. The closer the parameter value is to unity, the higher the efficiency in the production processes with respect to emissions. Also note that basing the formulation of the total damage on the material balance enables easy and accurate investigation of the effects of parameters and decision variables on total external costs.

As in Section 2 the regulator's general problem is to maximize the welfare for the overall system for a given amount of material inflow (q_0). We note that zero production of recycled material is obviously among the possible outcomes, whereas 100 % efficient recycling is rejected by the laws of thermodynamics. This aspect is taken care of by the restriction on production technology given by $k_M \in \langle 0,1 \rangle$.

Using all specific functions, defined above, in an expression for social welfare equal to Equation (6), we can formulate social welfare as a function of q_0 and the decision variables α and β:

$$W = U(\alpha q_0) + \alpha\beta q_0\left(\psi - aK_M - \frac{b}{2}K_M^2\alpha\beta q_0 + gK_E + \frac{h}{2}K_E^2 q_0(2-\alpha\beta)\right)$$

$$+q_0\left(P_E e k_E - w l_E - gK_E - \frac{h}{2}K_E^2 q_0\right)$$

where $\psi = P_M k_M - P_E k_E e - w(l_M - l_E)$, $K_M = 1 - k_M$ and $K_E = 1 - k_E$. The first order conditions for an interior solution is given as:

$$\partial U / \partial \alpha = 0 \Rightarrow$$

$$\psi - aK_M + K_E(g + hK_E q_0) - \alpha\beta q_0\left(bK_M^2 + hK_E^2\right) = -\frac{\partial U / \partial \alpha}{\beta q_0} \tag{18}$$

$$\partial W / \partial \beta = 0 \Rightarrow \psi - aK_M + K_E(g + hK_E q_0) - \alpha\beta q_0\left(bK_M^2 + hK_E^2\right) = 0 \tag{19}$$

Combining equations (18) and (19) we find that the efficient level of household sorting is found where $\partial U / \partial \alpha = 0$, which implies that $q_H^* = \alpha^* q_0$. That is, the households base their decision on sorting effort solely on their own direct utility. That this solution is consistent with efficiency for the overall system is related to two aspects. First, the households generate no external effects, and second, they do not derive utility from q_M. The efficient level of β is found from Equation

(19): $\beta^* = \frac{\psi - aK_M + K_E\left(g + hK_E q_0\right)}{\alpha^* q_0\left(bK_M^2 + hK_E^2\right)}$, that is where the marginal social profit for the material recycling and energy recovery processes are equal. Thus, for a given level of k_M the overall material recycling rate is given as:

$$\mu^* \equiv \alpha^* \beta^* = \frac{\psi - aK_M + K_E\left(g + hK_E q_0\right)}{q_0\left(bK_M^2 + hK_E^2\right)} \tag{20}$$

Holding the amount of waste constant an increase in the household sorting rate will therefore lead to a reduction in β^* because of the inverse relationship between α^* and β[15]. This indicates that there is a trade off between recycling efforts in the households and the industrial recycling process. Less output in the households leads to an increase in labor use in the recycling process to retain an efficient production level. In other words, although the individual production processes are characterized by fixed proportions, the regulator can substitute between material input and labor use in the material recycling process to achieve the efficient recycling rate.

In Section 2 we noted that in a decentralized model the production in the household sector could restrict the possibility to implement the efficient solution from a regulator's point of view. Whether the household sorting rate is constraining the overall recycling rate or not is found from investigating how μ^* is affected by the different parameters in the model. Moreover, the likelihood of α being a limiting factor for the overall recycling rate is (see Appendix A for details)[16]:

i) decreasing in a, b (since both parameters affects μ^* negatively),

ii) increasing in g, h, ψ and q_0 (as g, h and ψ affects μ^* positively),

iii) increasing in k_M when the level of q_0 is equivalent to a level where an increase in k_M leads to an increase in μ^*, and vice versa (see Section 3.4),

iv) increasing (decreasing) in w whenever $l_E > l_M$ $\left(l_E < l_M\right)$.

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[15] $\frac{\partial \beta^*}{\partial \alpha^*} = -\frac{\psi - aK_M + K_E\left(g + hK_E q_0\right)}{\left(\alpha^*\right)^2 q_0} < 0$, $\frac{\partial^2 \beta^*}{\partial \alpha^{*2}} = \frac{2\left(\psi - aK_M + K_E\left(g + hK_E q_0\right)\right)}{\left(\alpha^*\right)^3 q_0} > 0$.

[16] Given that $\psi - aK_M + K_E\left(g + hK_E q_0\right) < 0$, the optimal level of recycling is zero, which means that the planner does not have to worry about the household sorting level since α^* automatically will be larger than μ^*. This can alternatively be interpreted in relation to the total amount of waste; the household sorting level is never a limiting factor when $q_0 < \Omega$ (see Figure 4). The results reported are based on a positive overall recycling rate, i.e. that $\psi - aK_M + K_E\left(g + hK_E q_0\right) > 0$. For more details see Appendix B.

Points ii) and iv) both have policy implications and require comments. In Section 3.4 we will see that the relationship between q_0 and μ^* can be both positive and negative, whereas we now see that increased q_0 unambiguously increases the likelihood of α constraining μ. This happens because household sorting is diminishing in q_0, whereas μ^* converges towards a positive constant for increasing levels of q_0[17]. Clearly, this type of information is crucial for policy makers when it comes to explaining bottlenecks in waste treatment systems. Regarding point iv), it is acknowledged that changes in the input factor prices affect the efficient recycling rate. Since it is plausible to assume that the recycling process is more labor intensive than energy recovery, we conclude that an increase in the price of the input factor in the processes besides waste reduces the chance of α being a limiting factor on the efficient recycling rate. This is simply because the cost increase is largest in the recycling sector. The relevancy for policy making is that this result illustrates how taxes on labor influence the efficient recycling rate, for instance that reducing taxes on labor can have a positive effect on the level of material recycling.

3.2. *Total amount of waste input*

An attractive feature of the approach presented in this paper is that we are able to analyze the system for varying levels of waste generated in the households. We can also experience that the efficient rate of recycling can increase as well as decrease with the level of total waste. As a starting point note that as the amount of waste increases the recycling rate ultimately reaches a level given by $\lim_{q_0 \to \infty} \mu^* = \frac{hK_E^2}{hK_E^2 + bK_M^2}$[18]. The rationale behind the main trade-off ultimately being between bK_M^2 and hK_E^2 is related to the formulation of this system which is based on the law of mass conservation and a convex dose-response relationship. In other words, for sufficiently high levels of total waste generated in the household sector the environmental damage dominates the net welfare, which means that efficiency criteria are based on other aspects than market conditions. As we will see, the economic parameters only influence the path towards the convergent recycling rate.

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[17] The negative relationship between α^* and q_0 is seen from the fact that $q_H^* = \alpha^* q_0$.

[18] Remember that the "real" total recycling rate is of course given as $k_M hK_E^2 / \left(bK_M^2 + hK_E^2\right)$ as $M \equiv \alpha\beta k_M$.

Consider a situation characterized by $\psi < aK_M - gK_E$, in other words, a pro energy recovery situation where the profit in the recycling sector is low compared to the profit in the energy recovery sector[19]. For low levels of waste, the relative environmental externalities do not defend a positive recycling rate. However, for an increasing level of waste the efficient rate of recycling will increase as the externalities in the energy recovery sector increases more than the profit. Thus, based on the information offered above, a distinction between two separate situations can be made (illustrated in Figure 4):

a) Whenever market conditions are favorable for material recycling and the level of waste input to the system is sufficiently low $(q_0 < \Gamma)$, the efficient solution is to produce as much recycled materials as technically possible from the total amount of waste. For increasing levels of waste input to the system, the efficient degree of material recycling must be reduced until the overall efficient recycling rate is achieved.

b) In cases where the market conditions are in favor of incineration of the waste, the socially efficient solution is to produce district heating from the waste as long as the level of waste is below a specific level (Ω). Above this level one should increase the recycling rate towards the overall efficient level.

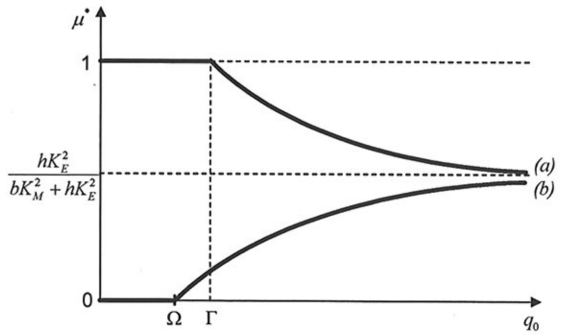

Figure 4 Relationship between total amount of waste input and total efficient recycling rate (indicated by thick lines). Curves (a) and (b) illustrates situations where $\psi < aK_M - gK_E$ and $\psi > aK_M - gK_E$, respectively. $\Omega = \frac{\psi - aK_M + gK_E}{-hK_E^2}$, and $\Gamma = \frac{\psi - aK_M + gK_E}{bK_M^2}$.

3.3. Centralization of waste handling services

An important issue at the regional level is to decide the size of the waste treatment plant. Should each municipality run its own system, or should municipalities

[19] For details about these results, see with Appendix B.

coordinate their waste streams so that a central treatment plant is established? Although we have omitted costs related to the transport of the waste, our model enables important insights to be gained about issues regarding centralization versus decentralization. More specifically, our system may give a positive as well as a negative net welfare for different levels of total waste. For levels of waste input where the net marginal welfare is positive, the owner of the system, typically a municipality, would be interested in receiving waste from nearby municipalities where the marginal social welfare of total waste is lower or negative[20]. However, as environmental costs are likely to increase on the margin and more so than economic returns, the marginal social welfare will eventually become negative as the quantities of waste increases[21]. Thus, there exists an upper limit for the degree of centralization, which is lowered when considering the negative effects from transport.

3.4. Technical improvement

The promise of new and improved production technology is often used as an argument underpinning that certain waste treatment strategies will prove to be efficient some time in the future. In this section we investigate how improved technology affects the efficient recycling rate within a waste treatment system. The answer to this question thus depends on whether it is optimal for the plant owner to implement the new technology, and on the level of waste that is to be processed.

Improved technology in relation to waste streams may be seen as improved labeling or a standardization of packaging. If technological improvement is interpreted as an increase k_M it can be shown to have a positive effect on the overall efficient recycling rate[22]. However, as Khanna and Zilberman (1997) argue, the result of adopting new technology is generally a mix of precision-, productivity- and pollution-effects. This section shows that it is possible within the model to examine what impacts such effects have on recycling systems.

In order to reveal the exact relationship between different production technologies and the efficient level of recycling the model is specified in more detail so that the revenue and costs in the recycling sector reflect different aspects related to the implementation of new technology. Production in the recycling

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[20] Any scale effects are not seen since we assume linear technology, which is relevant for many recycling systems (Eik et al. 2002).

[21] It is shown in Appendix C that $\partial^2 \widehat{W} / \partial q_0^{\,2} < 0$, where $\widehat{W} = W - U$.

[22] For details see Appendix A.

sector is now defined as $q_M = k_{M,i}\alpha\beta q_0$, where subscript $i = 1,2$ represents traditional and precision technology, respectively[23]. We assume that $k_{M,1} < k_{M,2}$. In addition to the precision effect, we also have a productivity and a cost effect, both of which are reflected in the cost function: $C_{M,i} = w_{M,i}l_{M,i}\alpha\beta q_0$. More specifically, we assume that implementing precision technology requires more skilled labor. Thus, we have a cost effect in that $w_{M,1} < w_{M,2}$. Productivity effects due to the use of more skilled labor are seen from the assumption that $l_{M,1} > l_{M,2}$. For simplicity we assume that there is no precision technology available in the energy recovery sector[24].

It is easy to verify that $\Delta k_M > 0 \Rightarrow \Delta\mu^* > 0$, $\Delta l_M < 0 \Rightarrow \Delta\mu^* > 0$ and $\Delta w_M > 0 \Rightarrow \Delta\mu^* < 0$. Hence, we have precision- and productivity effects with positive impacts on the efficient recycling rate, whereas the cost effect works in the opposite direction. Consequently, from the results obtained in Section 3.2 regarding the relative importance of economic and environmental factors it follows that the effect of implementing new technology for moderate quantities of waste might be that the recycling rate is lowered due to the impact from increased costs related to the use of labor dominating modest environmental externalities. For a larger amount of waste the impact of environmental factors increases and the effect on μ^* is unambiguously positive. Figure 5 illustrates a possible scenario for technology implementation when market conditions are in favor of energy recovery.

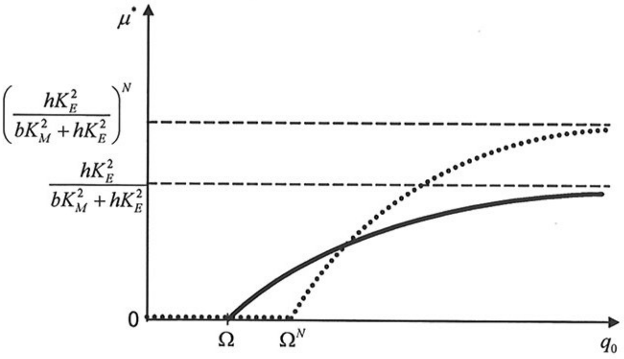

Figure 5 The impact on the efficient recycling rate from applying new and improved technology (illustrated in the figure by the dotted curve). Variables associated with new technology is denoted by superscript *N*.

[23] Following Khanna and Zilberman, precision technology is defined as technology improving the ratio of input to output in the production process.

[24] Note that Khanna and Zilberman also emphasize a pollution effect; precision technology increases the efficiency in the input use and hence reduces the level of residuals, which in our model works through the precision effect.

Note, however, that these effects are seen only when the precision technology is implemented[25]. Conditions under which new technology is implemented can be divided into two scenarios[26]. The first scenario is one where the authorities instruct the recycling plants to use new technology based on environmental savings due to the precision- and pollution effects. For low levels of q_0 the economic factors are the dominant and we may see a reduction in μ^* because the increased costs $(\Delta w_M > 0)$ reduce the efficient level of production at the recycling plant.

A second scenario is one where the use of new technology increases the profit at the recycling plant:

$$P_M \Delta k_M > w\Delta l_M + l_M \Delta w \tag{21}$$

In this situation the new technology is applied but it will lead to a change in the efficient level of μ. Given that wages must be increased to hire more skilled labor, it is easy to show that μ^* decreases when the following additional condition holds:

$$\left(P_M + \left(a + 2bK_M \frac{\psi - aK_M + K_E\left(g + hK_E q_0\right)}{bK_M^2 + hK_E^2} - w_M l_M\right)\right) dk_M < k_M \left(w_M dl_M + l_M dw_M\right)$$

(22)

Conditions (21) and (22) hold at the same time for a sufficiently low level of q_0 since the smaller the amount of waste that must be processed, the larger the influence from economic factors on the efficient recycling rate. We must also have a sufficiently high level of w_M so that the cost effect dominates the positive effect on μ^* of the increase in labor productivity[27].

4. CONCLUDING COMMENTS

The main conclusion is that the likelihood of recycling being part of an efficient strategy for processing waste increases with the amount of total waste generated in the economy. However, recycling is at the same time a limited tool for treating waste as we expect the rate of household sorting to decrease with the level of total

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[25] Note that we have only variable cots related to the precision technology. Generally, we will also have some fixed costs related to required investments linked to the implementation of the new technology.

[26] Whether the productivity effect dominates the cost effect depends also on the heterogeneity in the physical conditions surrounding different production facilities, which means that implementation of precision technology increases profits in various degrees. Consequently, new technology may be implemented in facilities in one region but not in others.

[27] This is seen from investigation of Ω.

waste. In order to deal with this latter problem the planner has to spend money on motivating households to increase their level of sorting. We have shown that the amount of resources that can be allocated towards motivational activities is restricted by the potential gain from increasing the overall recycling rate, where the latter element is determined by the preferences of households together with the rest of the economic and environmental parameters in the model. As a result of possible inelastic household behavior, policy measures to reduce the total amount of waste might be the most efficient way of dealing with costs associated with our production and consumption activities.

Another important finding from the analysis is that large amounts of waste are neither a necessary condition for securing material recycling or energy recovery as a dominant part of an efficient solution, nor are they sufficient *per se* to ensure a "high" level of recycling or recovery to be efficient solutions. This argument is based on the fact that sooner or later the marginal damage from either of the processes will increase as the level of waste to be processed increases. It is therefore important to assess the economic as well as the environmental impacts from each of the alternatives for processing waste for *varying* levels of material flow. Only then can we discuss efficient strategies for waste handling valid for the future, since changes in the flow of material are likely to occur.

The presented analysis has also demonstrated that improvements in technological efficiency in the recycling sector do not necessarily lead to a higher overall recycling rate as it might entail a positive cost component dominating increased input factor productivity. Regulators must therefore be aware that if recycling plants are instructed to implement new technology, the efficient recycling rate will be reduced in some plants.

Some critical assumptions are made which must be commented upon. First, this is a partial analysis on a micro-level. The flow of material and energy at a more aggregated level in society should be considered to ensure that the validity of the external costs and the market prices is satisfactory. The same kind of criticism could be addressed regarding the neglect of general equilibrium aspects. Further, assuming linear production technology limits the validity of the study, but it offers important insights as the processes within many recycling systems is appropriately characterized by this kind of technology. It is also important to remember that preferences and market conditions change over time, which obviously will affect the results from the static analysis that is presented.

The paper has demonstrated that a combination of energy recovery and recycling can serve well as complementary policies, and we have pointed out some of the elements that are important when designing an effective waste treatment system.

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APPENDIX

Appendix A Comparative statics for model in Section 3

Comparative statics for the efficient recycling rate given as:

$$\mu^* = \frac{\psi - aK_M + K_E\left(g + hK_E q_0\right)}{q_0\left(bK_M^2 + hK_E^2\right)}$$

$$\frac{\partial \mu^*}{\partial a} = \frac{k_M - 1}{q_0\left(bk_M^2 + hk_E^2\right)} < 0\,,\ \frac{\partial \mu^*}{\partial b} = \frac{-\left(\psi - aK_M + K_E\left(g + hK_E q_0\right)\right)K_M}{bK_M^2 + hK_E^2} < 0$$

$$\frac{\partial\mu^*}{\partial g}=\frac{1-k_E}{q_0\left(bK_M^2+hK_E^2\right)}>0,$$

$$\frac{\partial\mu^*}{\partial h}=K_E^2\frac{q_0\left(bK_M^2+hK_E^2\right)-\left(\Psi-aK_M+K_E\left(g+hK_Eq_0\right)\right)}{q_0\left(bK_M^2+hK_E^2\right)^2}$$

$$\Rightarrow\frac{\partial\mu^*}{\partial h}\left\{\begin{array}{l}>0\text{ if }q_0>\dfrac{\psi-aK_M+gK_E}{-hK_E^2}\equiv\Gamma\\[2ex]<0\text{ if }q_0<\dfrac{\psi-aK_M+gK_E}{-hK_E^2}\equiv\Gamma\end{array}\right\}\frac{\partial\mu^*}{\partial h}>0$$

The effect from an increase in h is determined by the fact that $\mu<1$.

$$\frac{\partial\mu^*}{\partial q_0}=\frac{hK_E^2q_0-\left(\psi-aK_M+K_E\left(g+hK_Eq_0\right)\right)}{q_0\left(bK_M^2+hK_E^2\right)^2}\begin{cases}>0\text{ if }\Psi-aK_M+gK_E<0\\<0\text{ if }\Psi-aK_M+gK_E>0\end{cases}$$

$$\frac{\partial\mu^*}{\partial w}=\frac{l_E-l_M}{q_0\left(bK_M^2+hK_E^2\right)}\begin{cases}>0\text{ if }l_E>l_M\\<0\text{ if }l_E<l_M\end{cases}$$

$$\frac{\partial\mu^*}{\partial k_M}=\frac{\left(P_M+a\right)\left(bK_M^2+hK_E^2\right)+2bK_M\left(\psi-aK_M+K_E\left(g+hK_Eq_0\right)\right)}{q_0\left(bK_M^2+hK_E^2\right)}>0$$

Appendix B Varying levels of waste

A detailed investigation of μ^* reveals that: $\frac{\partial\mu^*}{\partial q_0}=\frac{-\left(\psi-aK_M+gK_E\right)}{q_0^2\left(bK_M^2+hK_E^2\right)}$ and

$\frac{\partial^2\mu^*}{\partial q_0^2}=\frac{2\left(\psi-aK_M+gK_E\right)}{q_0^3\left(bK_M^2+hK_E^2\right)}$. Our analysis therefore suggests that if:

i) $\psi<aK_M-gK_E$

$$\Rightarrow\frac{\partial\mu^*}{\partial q_0}>0,\ \frac{\partial^2\mu^*}{\partial q_0^2}<0\text{ and }\mu^*<\lim_{q_0\to\infty}\mu^*$$

$$\Rightarrow\mu^*=0\text{ if }0<q_0<\frac{\psi-aK_M+gK_E}{-hK_E^2}$$

ii) $\psi>aK_M-gK_E$

$$\Rightarrow\frac{\partial\mu^*}{\partial q_0}<0,\ \frac{\partial^2\mu^*}{\partial q_0^2}>0\text{ and }\mu^*>\lim_{q_0\to\infty}\mu^*$$

$$\Rightarrow\mu^*=1\text{ if }q_0>\frac{\psi-aK_M+gK_E}{bK_E^2}>0$$

Appendix C Centralization of waste handling

If the marginal social profit differs between municipalities it may be efficient to centralize the waste handling processes. Since the households in 'our' municipality do not have to sort any of the incoming waste from other municipalities we define $\widehat{W} = W - U$. From the following it is evident that there is an upper limit for the degree of waste handling centralization:

$$\frac{\partial \widehat{W}}{\partial q_0} = \alpha\beta\left[P_M k_M + P_E e k_E - w\left(l_M - l_E\right) - aK_M + gK_E - bK_M^2 \alpha\beta q_0\right.$$
$$\left. + hK_E^2\left(2 - \alpha\beta\right) q_0\right] + P_E k_E e - w l_E - gK_E - hK_E^2 q_0$$

and $\partial^2 \widehat{W} / \partial q_0{}^2 = \alpha\beta\left(-bK_M^2 \alpha\beta - hK_E^2\left(\alpha\beta + 1\right)\right) < 0$.

Paper 3

An empirical assessment of solid waste management: recycling of household waste in Trondheim

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Abstract

Various studies that use different methods to assess the environmental and economic aspects of waste handling systems often arrive at contradictory conclusions when it comes to suggesting efficient treatment strategies. Using Material Flow Accounting in combination with a Cost Benefit Analysis it is possible to estimate the efficient level of recycling of plastic packaging waste generated by households in Trondheim, Norway's third largest city. The alternative way of processing the waste is to produce district heating by waste incineration. The novelty of the presented analysis is that production functions and the flow of materials through the system are explicitly modeled and used as a basis for an economic and ecological assessment of the system. The local health effects for two types of emissions are analyzed; NO_X and dioxins, which are modeled both as linear and convex damage functions. Global environmental impacts are represented by the emission of CO_2. Our results indicate that producing district heating from the plastic packaging waste is most efficient from an economic point of view. On the other hand, the negative environmental impacts are lower when the waste is recycled into new products. The latter result is however turned around if one implements the latest incineration technology which almost eliminates emissions of dioxins. Note that the choice of waste treatment method depends on the total level of waste entering the system. The determination of the efficient mix of the two treatment alternatives is also greatly influenced by whether recycled materials actually replace products made from virgin material, whether the energy that the incineration of plastic replaces would otherwise be produced by oil or coal, and whether we include household costs or not.

Keywords: recycling, energy recovery, mass balance, waste treatment.

1. INTRODUCTION

Until a few decades ago, landfilling was the preferred alternative for handling solid waste. However, dumping of solid wastes and the flow of untreated discharges can disturb ecosystems, leading to deterioration of groundwater quality and, consequently, reduced human health. In addition, a number of countries experienced that landfill space became increasingly scarce which led them to implement interventions involving high recycling rates (Goddard 1995). Nevertheless, management of solid waste from households is important since whatever waste disposal alternative is chosen, it carries with it some kind of environmental degradation.

There are an impressive number of studies analyzing solid waste management and efficient rates of recycling and this work has produced extensive recommendations[1]. The primary intention with this paper is to demonstrate a method combining a Material Flow Accounting (MFA) approach with a Cost-Benefit Analysis (CBA). The method is applied to the waste management system for plastic packaging waste in Trondheim, Norway's third largest city, in the year 2000. The presented approach broadens the scope of frequently applied methods, such as conventional cost-benefit analysis and Life-Cycle Assessment. The main contribution from the approach presented in this paper is that production functions and mass balance conditions are explicitly included in the assessment and thereby enables optimization to be made instead of evaluating merely a given project or a functional unit. This approach highlights the interdependency between different processes within the system and, given our various presumptions, enables estimation of the efficient rate of recycling.

Environmental externalities are in practice a normal and inevitable part of economic activity but there has to some extent been a de-coupling between economic and resource throughput on a per capita and per unit gross domestic product basis. Nevertheless, overall resource use and waste flows into the environment are growing in Western countries (Matthews et al. 2000), thus our focus is on how municipalities best can handle the increasing amounts of plastic packaging waste[2]. When addressing this problem we come in contact with the so-called waste hierarchy, which claims that preventing waste is the most environmentally friendly option, followed by reuse, re-manufacturing, mechanical recycling, feedstock recycling, energy recovery, incineration

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[1] See Choe and Fraser (1998) and Kinnaman and Fullerton (1999) for a review of solid waste management.

[2] It should be mentioned that from a broader perspective, a focus on increasing the value added and also on reducing the environmental impacts from production and distribution of the *product* that is packed, is perhaps even more important.

and landfill (EU 1999; Wollrad and Schmied 2000). However, the presented analysis demonstrates that the internal ranking within the waste hierarchy must be determined in each specific case as it depends among other things, on the amount of waste to be processed. Moreover, we advocate a view that the goal in many instances must be to find an efficient *mix* of waste treatment alternatives.

The system in Trondheim can be described as one where households sort their plastic waste into recyclable and non-recyclable fractions. The recyclable fraction is used to produce pallet blocks and granulate, whereas the non-recyclable fraction is incinerated in order to produce district heating. The environmental and economic efficiency within this system has been analyzed earlier by Olaussen (2002), Econ (2002) and Eik et al. (2002). Olaussen (2002) is a cost-benefit analysis of the plastic packaging waste system where a scenario with 100 % energy recovery by incineration is compared to one characterized by 85 % energy recovery and 15 % material recycling. The study shows that the scenario with 15 % material recycling is the most efficient, although it is emphasized that new incineration technology that reduces dioxins emissions, will reverse the conclusion. Econ (2002) also conducts a cost-benefit analysis but for the entire waste system in Trondheim. All waste fractions considered together, they find that 13 % recycling, which was the case in 2001, is more efficient than energy recovery only. However, for the plastic packaging waste fraction they recommend that 100 % of waste should be incinerated. A third study of plastic packaging waste in Trondheim is found in Eik et al. (2002). This is a life cycle assessment of the years 1999-2001, and they find that material recycling is likely to improve the environmental performance of the system but the economic costs point to energy recovery as the efficient solution.

In this paper, the local health effects for two types of emissions are included; NO_X and dioxins, which are modeled both as linear and convex damage functions. Global environmental impacts are represented by the emission of CO_2. Our main results suggest that if one considers only the revenues net of production and transportation costs, producing district heating from the plastic packaging waste is the most efficient solution. However, when accounting for environmental externalities, material recycling is the efficient way to process the waste. On the other hand, the result is reversed if one implements the latest incineration technology which is able to almost eliminate emissions of dioxins. Another result is that what comes out as the efficient waste treatment method depends on the total level of waste entering the system. Additionally, the determination of the efficient mix of the two treatment alternatives is greatly influenced by whether recycled materials actually replace products made from virgin material, whether the energy that the incineration of plastic replaces would otherwise be produced by oil or coal, and whether we include household costs or not.

Several studies of efficiency within waste handling systems have been carried out but there seems to be a somewhat great discrepancy concerning the conclusions that the studies offer. A great deal of the disagreement is based on the different methodological approaches applied in the various studies, and therefore the next section presents a brief review of the tools that are most widely used in assessments of environmental and, to some extent, economic impacts. Section 3 gives an outline of the MFA-CBA approach before the system of waste handling in Trondheim is described in Section 4. The results are reported and discussed in Section 5. Section 6 concludes the paper.

2. EVALUATION METHODS FOR RECYCLING SYSTEMS

From a strictly ecological point of view all materials and products should be reused, remanufactured or recycled, and in this way remain in the economic system as long as possible before being incinerated or placed at landfills. However, several studies have concluded that mainly due to high costs in the collecting and sorting phase, a high degree of recycling is not necessarily a better solution than energy recovery, incineration or landfill (see e.g. Bruvoll 1998; GUA 1999; and APME 2000). On the other hand, other studies have arrived at different conclusions (Raadal et al. 1999, Wollny and Schmied 2000)[3]. Much of the discrepancy is caused by the development, choice and application of the various methodologies used to carry out these analyses, in which some degree of subjectivity is inevitably inherent (Hertwich 2000). We therefore offer a brief review of the most frequently applied methods for evaluating the environmental and economic effects of products, systems or regions before presenting the MFA-CBA approach.

As is well known, Cost-Benefit Analysis is used as a tool for comparing the benefits of investment projects, or a policy measure, with its costs. The method aims at including all positive and negative effects and making a comparison by expressing them in monetary terms. The first step in a CBA is to determine which negative and positive impacts are to be examined, identify them and finally attach some monetary measure to each of them, i.e. the costs and benefits are weighted against each other. The basis for CBA is valuation methods including travel cost, hedonic pricing and contingent valuation (Arnold 1995; Abelson 1996).

Another frequently used method is Life Cycle Assessment (LCA), which has developed rapidly since it was established early in the 1990s and has now reached a

[3] For an extensive survey of analyses of household waste generation see Goddard (1995).

certain level of harmonization and standardization. An ISO standard (the ISO 14040 series, ISO 1998) has been developed along with a number of guidelines. The aim of the method is to specify all environmental impacts of products or services throughout the life cycle of a product, i.e. from raw material acquisition through production, use and recycling, recovery or disposal (Udo de Haes 1996). It also links changes in the economy to impacts within the environment by studying different options to supply a given function. Since a demarcation of system borders in an LCA always implies difficult decisions about which flows and how much of each flow to include within the system borders, the issue of allocation procedures is a hotly debated issue (see, for instance, Finnveden 1999).

The strength of LCA is its comprehensiveness in assessing the environmental impacts that are related to the function under investigation, and its ability to avoid problem shifting from one stage in the life cycle to another and from one location to another. The most important disadvantages of the method are the huge data requirement, the neglect of rebound and other societal effects, and that issues related to scale are outside the scope of LCA because it focuses on function, not volumes. Consequently, analyzing a scenario with an increased recycling ratio is outside the scope of LCA, because we cannot say to what purpose the recycled material is being used and mix of input factors may change as the scale of production changes. According to Brekke and Vennemo (1999), LCA also ignores already implemented policy instruments through the life cycle, and the approach to valuation of environmental impacts is characterized as insufficient and potentially misleading.

In order to remedy some of the weak points of an LCA one can combine it with other assessment methods. The approach found in Weaver et al. (1997) is a combination of linear programming and LCA. The strength of this study is that instead of predicting environmental effects from an exogenously given policy or policy instrument, they use linear programming to find the optimal organization of the sector and hence, offer an input to policy making. It is argued that a holistic approach should include the use of different technologies, lock-in/lock-out of technologies (partly because of command and control policy making), geographical distribution of production, and the individual country's industrial and trade performance. However, some important aspects are missing from this analysis, which should be noted. First, it is a static analysis and it ignores possible effects on other sectors in the economy (which can be assessed by an input-output analysis). The weighting of the various impacts on the environment and human welfare is based on LCA methodology with the problems related to this (see e.g. Finnveden 1999). Further, only environmental impacts are analyzed, other effects such as those arising from increased use of labor are not considered. Lastly, the authors do not allow for policies already implemented that are aimed at internalization of the externalities.

Hendrickson et al. (1998) combines an LCA and an input-output analysis, a method also known as EIO-LCA (see also Matthews and Small 2000). Environmental Input-Output Analysis (EIO) is based on an extension of the well known Leontief model focusing on inter-industry linkages (Miller and Blair 1985). Used for environmental extensions the model includes additional conditions to capture the relations between industrial production, pollution generation and abatement activities. The main problem in this kind of approach is to find the appropriate unit of measurement of environmental quantities, which would be monetary or physical units.

What the Hendrickson et al. analysis investigates is the environmental impacts of the present situation: static, given technological level and demand. The major strengths of the study are that EIO-LCA avoids the problem of drawing arbitrary system boundaries, and that it is a transparent and efficient method, which captures the upstream environmental burdens associated with raw material extraction and manufacturing processes. This approach can therefore be applied to assessments of the extraction and manufacturing-stage of a material which advantageously places its focus on specific processes. This is unlike conventional LCA studies which are often applied only to the product-use and end-of-phase assessments. An important weakness of the approach is that LCA cannot be used in input-output analyses tracking *all* indirect contributions, because cycles among stages would continue indefinitely (Brekke and Vennemo 1999). In addition we still have the problem that Leontief technology implies constant returns to scale and that possible limitations following from a positively sloped supply curve are not reflected. So even though the EIO-LCA approach has achieved some progress, problems still remain related to combining a tool at the micro level (LCA) with a technique developed for a higher aggregated level (input-output) (see Joshi 1999).

An interesting comparison of three different methods has been carried out by Bouman et al. (2000). The methods under investigation are Substance Flow Analysis (SFA), LCA, and Partial Economic Equilibrium Analysis (PEA). By analyzing a case of producing and recycling a conventional battery versus a hypothetical green battery they offer a discussion of the similarities and dissimilarities between the methods. The main conclusion is that the methods are complementary rather than contradictory. Each of them emphasizes different aspects of the problem at hand, and a sequential application of the methods seems to be more fruitful than trying to construct an encompassing model that integrates all models. It should be noted that the study of the different approaches makes some simplifications. There are for instance no stocks of materials, only steady states are considered as it is a static analysis, and there are limited possibilities for substitution among input factors.

The SFA basically identifies the material flows within systems, but it says nothing about costs or emissions of other substances than those directly related to the substance under scrutiny. Hence, as there are no economic mechanisms included in the SFA it leads to all or nothing solutions. On the positive side we note that the SFA is capable of analyzing large systems, it is easy to relate environmental problems to economic origins, and lastly, it is easy to assess impacts of different technical solutions.

The PEA focuses on elements considered to be essential by the researcher. Which elements to include is consequently a critical stage in the PEA, which can influence the result of the analysis to a great extent. Much information is needed for determining parameters and functional forms, but the most severe aspect of the PEA is the, sometimes implausible, assumptions needed to keep the mathematical model tractable. However, it reveals the complexity of economic relations, which determines the effects of different policy measures, which obviously is important in itself.

Generally speaking, we can conclude that the SFA assesses the substance flows, while the LCA is able to illustrate the environmental impacts from different technological solutions (resulting in the same function or product), and the PEA points to the economic relations that influences the possibilities for increasing efficiency (scale, substitution, etc.), and what policy instruments are relevant for achieving the goals.

The analysis presented in this paper is based on Material Flow Accounting (MFA). MFA is a method for specifying the flow of materials into, through and out of a nation, a region, a business sector, company or a household for a given period in time (Wriesberg and Udo de Haes 2002). In contrast to an SFA, the material flow analysis deals with bulk materials, e.g. steel and wood. What this tool basically does is link material flows in the economy to pollution problems as well as resource requirements. Whereas the material flow analyses follow a cradle-to-grave approach, a substance flow analysis considers only the flows and accumulations as far as they can be related to the substance or substance group under scrutiny. Moreover, MFA is an input-related tool with a somewhat uncertain relation to environmental impacts, which makes the normative evaluation uncertain. However, it is a robust tool since one can link various material flows through a variety of different processes and it is therefore also a good basis for dynamic analysis of future scenarios (see for instance Kleijn et al. 2000).

Evidently, many of the methods have a different goal and scope, especially along the ecological-economic dimension[4]. Clearly, the ecological aspects are best assessed by the tools with a clearly environmental approach such as MFA, LCA and EIO. The basic goal of these methods is to grasp the environmental consequences following changes in production and consumption. However, sustainability also consists of economic and social dimensions, which means that methods expressing environmental and economic elements in the same monetized unit (such as CBA and Life Cycle Costing[5]) are valuable as support for decision making since the use of monetary flows as an indicator encompasses a range of social and economic impacts[6].

A conclusion that can be drawn from this brief discussion of assessment methods is that to have the full picture one should combine tools whenever it is appropriate. As it is possible to remedy one tool's weakness by using a complementary tool, successful combinations of tools will have advantages like the avoidance of problem shifting and addressing more of the relevant issues (Wrisberg and Udo de Haes 2002; Bouman et al. 2000). The next section describes an approach, which basically is a combination of MFA and CBA.

3. OPTIMIZING COST-BENEFIT ANALYSIS

In contrast to the two methods most frequently applied to analyses of waste related problems, namely cost-benefit analysis and life cycle assessment, which considers a given project or a functional unit, the method applied in this paper enable us to evaluate what degree of recycling is efficient for *different* levels of waste that have to be processed. This approach is based on the fact that the output in an upstream process is part of the input in a downstream process, and that the output from a process is a share between 0 and 1 of the material input. The residual mass is discharges. Consequently, by combining material balance conditions with production functions we are able to describe and derive efficiency criteria for the system under investigation as a function of the material input to the system. The assessments of environmental costs are based on the discharges from the processes, explicitly expressed as a function of the material input to the system. Hence, we have an overview of the interdependencies between production processes and also of how production in the various sectors affects the emissions from each of the processes. Consequently, a direct link can be established

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4 Other methods that are used in assessment of environmental problems in addition to those discussed above, are Material Input per Unit of Service (MIPS), Environmental Risk Assessment (ERA), and Life Cycle Costing (LCC). A brief review of these is offered in Appendix A.

5 For details on the latter see Appendix A.

6 Note that all tools mentioned here put relatively limited weight on ecosystem effects.

between production and emissions, both within and between processes. In addition, it is possible to ensure that the efficient solutions suggested by this analysis are technically feasible.

Important weaknesses in this approach, as outlined in the the previous section, are that impacts outside the system borders are excluded, that the connection between emissions and environmental costs are heavily influenced by the valuation procedure, that we analyze only one waste fraction, and finally that this is a static analysis.

The logic behind the material balance approach can be illustrated by considering a simplified model with a household sector, a material recycling sector and an energy recovery sector producing district heating, and where transportation costs are excluded. The point of departure for illustrating the flow of materials through the system is the household sector sorting out a fraction α of the total amount of waste, q_0, to be transported to the recycling sector. The residual fraction, 1-α, is transported to the energy recovery sector. Production in the household sector is therefore defined according to Equation (1) below. The incoming material in the recycling sector is next processed so that a fraction $k_M\beta$ of the material input ends up as a new product denoted q_M. β can be thought of as the degree to which the capacity of the recycling plant is utilized and k_M is a technical parameter denoting the upper limit for the input-output ratio in the process. Hence, production in the recycling sector is given as Equation (2) under the assumption that $\beta \in [0,1]$ and $k_M \in \langle 0,1 \rangle$. Following from Equation (2) we have discharges from the recycling process, denoted E_M, as defined by Equation (3). The material input in the energy recovery process consist of the material not sorted for recycling by the households and the material not used in the production of recycled material, which are not emitted as process discharges. Hence, the relationship between production of district heating, q_E, and material inputs follow from the other production functions is given as Equation (4). Here, $k_E \in \langle 0,1 \rangle$ is the environmental efficiency in the process of energy recovery, and e is the energy content per tonne waste, typically kWh/tonne. Consequently, discharges from the energy recovery process, E_E, can be described as in Equation (5).

$$q_H = \alpha q_0 \tag{1}$$

$$q_M = k_M \beta q_H \tag{2}$$

$$E_M = (1 - k_M)\beta q_H \tag{3}$$

$$q_E = ek_E(q_0 - q_M - E_M) \tag{4}$$

$$E_E = (1 - k_E)(q_0 - q_M - E_M) \tag{5}$$

The production processes in the system are linked together by the flow of mass through the system since the degree of substitutability between waste and other input factors is limited. Hence, production in the household sector affects the production of recycled material in the whole system, and can thereby be a restriction on the maximum output that can be produced from the system. Production in the energy recovery sector is also influenced by the decision in the households since the input in this sector is the mass which is not recycled. Consequently, total environmental damage is affected by decisions in the households, and next, in the material recycling sector. The relationships are illustrated in Figure 1 below.

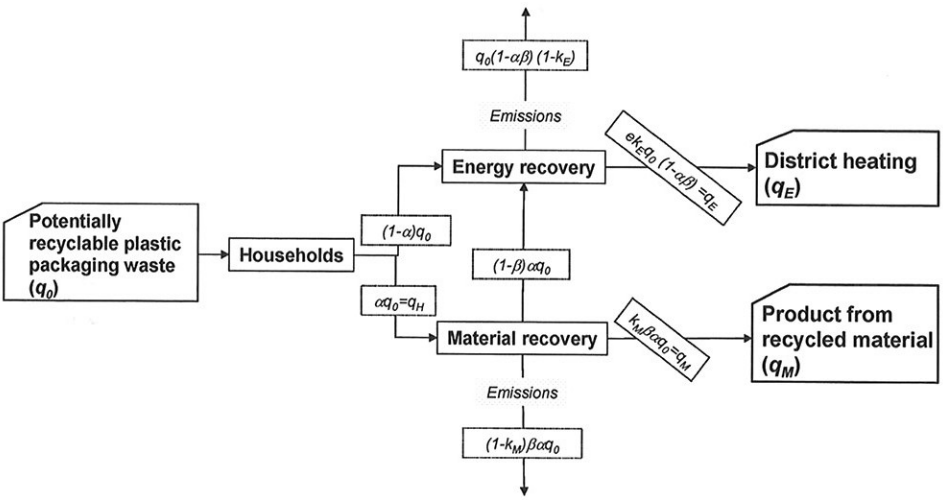

Figure 1 The material flow in a simple household waste handling system.

Combining Equations (1) - (5) with production functions, information about prices and environmental costs, we formulate equations for the profit and external costs within the system which enables us to derive efficient allocations of the material between energy recovery and material recycling based on the amount of waste that has to be processed (see Appendix B for details).

4. RECYCLING OF PLASTIC PACKAGING WASTE IN TRONDHEIM, NORWAY'S THIRD LARGEST CITY

The use of plastic packaging is steadily increasing and the major reason for this is the strengths, transparency, and low weight of plastic packaging. Today 40 % of the production of plastics is used for packaging, and 50 % of all food packaging is made from plastics (APME 2000). In 1998 around 12 million tonnes of plastic packaging was produced in Western Europe. Of this, 33 % was LDPE, 22 % was HDPE, 19 % was PP, and 10 % was PET[7]. To reduce the environmental problems caused by packaging waste voluntary agreements were signed in September 1995 by the Norwegian Ministry of the Environment (MD) and various industry sectors. The agreements were designed to ensure waste reduction and increased collection and recovery in the packaging product chains. The agreement between MD and the plastic packaging industry states that 80 % of the plastic-packaging waste is to be recovered, with a minimum of 30 % going to material recovery (Eik 2002). In 2000, 78 % of the plastic was recovered, of this, 19 % was recycled into new products and 59 % was energy recovered[8].

The plastic packaging waste generated by the households in Trondheim must either be recycled into new products or incinerated in order to produce district heating. The first step is that the households sort their waste into recyclable and non-recyclable fractions, where the latter is incinerated. Next, the recyclable waste is sorted at a central sorting plant where some of it goes back to incineration and the rest is either used as input in the production of pallet blocks (LDPE) or is transported to Sweden for further sorting (HDPE and PP). After the sorting in Sweden the material is transported to plants in Arvika and Töcksfors for production of HDPE and PP granulates, respectively.

In 2000 households in Trondheim generated 822 tonnes of potentially recyclable plastic waste[9]. The households put 40 % of this in their "environmental waste" bins which was transported to the central sorting plant in Heimdal, located in the southern part of Trondheim. The rest was put in their "rest fraction" bins and transported to the incineration plant, which also is located in Heimdal. Of the 328.8 tonnes to be sorted at the central sorting plant, 67.4 tonnes of LDPE (20.5 %) was used as input in the production of pallet blocks in Tydal. 17.9 tonnes of HDPE (5.4 %) and 11.4 tonnes of PP (3.5 %) were transported to Karlstad in Sweden and from there to Arvika and Töcksfors, respectively. Adding it all up we find that nearly 12 % of the potentially recyclable plastic packaging waste generated in Trondheim in 2000 was recycled into pallet blocks or granulate. An overview of the treatment system for plastic packaging waste in Trondheim for the year 2000 is given in Figure 2 below.

[7] A clarification of the plastic types under scrutiny: high density polyethylene (HDPE): bottles, cans and film; low density polyethylene (LDPE): cling film, bags, bin liners; polypropylene (PP): bottles, cans, e.g. yoghurt cups; polyethylene terephtalate (PET): bottles, food packaging.
[8] Note, that this includes other sources than household plastic packaging. Industrial packaging, such as agricultural plastic and reusable beverage crates, is easier to collect and recycle, and has higher recycling rates.
[9] 7.95 kg per person is found to be recyclable out of 12.5 kg per person totally generated (Raadal et al. 1999).

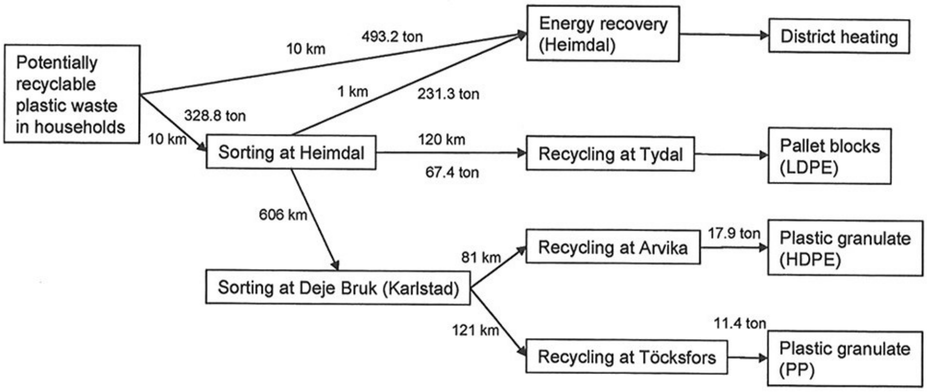

Figure 2 Overview of distances between plants and material flow for the plastic packaging waste generated by households in Trondheim in the year 2000.

Optimization within our model is done only with respect to the level of sorting at the central sorting plant in Heimdal. This is based on two grounds. First, in Tydal the plastic (LDPE) is compressed together with paper to form the pallet blocks, thus 100 % of the incoming material ends up in the new product. Second, since the material loss is only 2 % from the central sorting plant to the production of HDPE and PP granulate, we assume that 100 % of the waste originating from Trondheim is used as input in the production processes in Arvika and Töcksfors. The allocation of the material between Tydal and Karlstad, and between Arvika and Töcksfors is based on the composition of the plastic waste, i.e. the relative contents of LDPE, HDPE and PP.

5. ANALYSIS

5.1 Background and assumptions

The production processes in the system are fairly simple and can be described by fixed coefficient production functions. For instance, the central sorting is carried out manually along a conveyor belt and the production of pallet blocks is a process of compressing paper and recycled LDPE. Further, the environmental costs included in the assessment are based on three types of emissions, namely CO_2, NO_X and dioxins. Incineration of plastic has a direct effect by emitting CO_2, NO_X and dioxins, in addition to emissions due to the use of electricity in the production process. The production processes in the recycling sector are such that emissions are related only to electricity use in production. External costs from transport come in addition to those in the individual production processes.

The material flow model presented in Section 3 is the basis for the analysis, now extended to include three recycling facilities, transportation costs and the avoided environmental damage. The latter is a result of recycled material replacing plastics made from virgin material, and that district heating replaces fossil-fueled energy production. Let $i=M,E$, denote the material recycling and energy recovery sectors, respectively, and D_i and S_i denote environmental damage and avoided environmental damage from production in sector i, respectively. T denotes transportation costs. Total welfare in the system is formulated as: $W = \Pi_M + \Pi_E - D_M - D_E + S_M + S_E - T$, where Π_M and Π_E denotes revenues net of production costs for the recycling and energy recovery sector, respectively. From the first-order condition following a benevolent planner's maximization of total welfare, we are able to find efficient levels of material recycling and energy recovery, or we can evaluate the system for any given material flow through the system[10].

A number of assumptions are made that are worth noting. Given a four-container municipal waste handling system, we do not consider that the organization of the system can be suboptimal[11]. Further, any benefits or costs in the household sector related to the sorting of the waste are not included, i.e. the household sorting rate is treated as an exogenous variable. Assuming that no hot water is being used to clean the packaging, externalities in the households are presumed to be zero. We also exclude transport from households to central sorting since the waste must be transported to Heimdal, the same distance, regardless of whether it is recycled or incinerated. Another assumption is no closed loops, which is reasonable since this is a regional, not a national, level model, and that it is not likely that the material ends up as (food) packaging, let alone in the same municipality. Next, the marginal source of energy is assumed to be power plants using light oil as input. Pallet blocks made from LDPE and paper replaces pallet blocks made from sawdust, which according to Eik et al. (2002), creates no externalities. Further, externalities from transportation are assumed internalized in the fuel price. An overview of the data used in the analysis is found in Appendix D.

The analysis is divided into two parts. The first deals with the system under the assumption that marginal environmental costs are constant, whereas the second part investigates a situation where marginal environmental costs are increasing with the level of emissions[12]. When marginal costs are constant the production levels do not influence the results of the analysis. Hence, the first part of the analysis shares most of its features with a conventional CBA. In other words, as long as social profit from each of the two

[10] See Appendix B for a detailed derivation of the efficient recycling rate.

[11] The different containers are meant for hazardous waste, paper, plastics together with metal, rubber, electric components etc., and a rest fraction.

[12] Marginal costs related to emissions of CO_2 are still assumed to be constant.

treatment alternatives is linear in the production, the implication from our results will be a suggestion of either zero or 100 % material recycling as the efficient solution. In the second part of the analysis nonlinear environmental damage functions are used. Hence, an efficient *mix* of waste treatment alternatives can be estimated. In practice this means determining the efficient level of β, which is interpreted as to what degree the capacity in the recycling process is utilized (cf. Equation (2) and Figure 1). The main difficulty is to determine realistic formulations of the damage functions. In the literature only estimates for constant marginal damage are found (Econ 2000). The chosen approach is therefore an indirect one, i.e. we investigate how the efficient mix changes as the curvature of the damage functions are altered.

5.2 Linear environmental costs

Table 1 summarizes the results of the analysis when marginal environmental costs are constant. Considering only the economic revenues and costs, incineration of the waste is clearly the most profitable. The main reason for this is that recycling is relatively labor intensive compared to energy recovery. However, considering also the environmental costs, recycling of the material into new products is the most efficient solution given our three emission categories and that household costs are assumed negligible. A difference in net marginal damage of NOK 3073 per tonne in favor of recycling is large enough to dominate the difference in marginal profit of NOK 2111 per tonne in favor of energy recovery. The great advantage of recycling is the avoidance of large CO_2 emissions from production using virgin materials, and the environmental disadvantage of energy recovery is the emissions of dioxins, which make up 90 % of total marginal damage related to this particular process. From the material recycling process it is emissions of CO_2 that contribute most to the external costs, namely 63 % of the environmental costs.

Recent technological developments show that the discharge of dioxins in modern incineration plants is reduced substantially relative to the numbers behind the results in Table 1. Moreover, a reduction of around 90 % can be achieved by the implementation of the latest technology (Borgnes and Ringstad 2002; Econ 2000, p.22). Excluding dioxins from our analysis the efficient strategy turns out to be incineration of the plastic packaging waste (see Table 1). Moreover, a reduction in dioxin emissions from the incineration process of 31 % is sufficient to make energy recovery the efficient waste treatment solution. Thus, with the incineration technology available today energy recovery seems to be the efficient choice, and an increase in energy consumption leading to district heating substituting energy made from coal fired plants abroad will add support to this conclusion (see Table 2).

Table 1 Results from the analysis of the system for recycling and energy recovery of household plastic packaging waste in Trondheim in the year 2000. A linear relationship is assumed between environmental costs and emissions. Emissions consist of CO_2, NO_X and dioxins. Household costs are not included. Reported numbers are NOK per tonne of plastic packaging waste. Results are reported in year 2000 prices.

Sector	Social marginal profit	Marginal profit	Marginal damage	Avoided marginal damage
Material recycling	169	-66	114	349
Energy recovery	-792	2045	3173	335
Results excluding dioxins:				
Energy recovery	2060	2045	321	335

Disaggregation of marginal costs (results excl. dioxins in parentheses)

	Percentage of total marginal damage		
	CO_2	*NO_X*	*Dioxins*
Material recycling	63	18	18
Energy recovery	8 (83)	2 (17)	90

Note: i) all numbers may not add up because of rounding, ii) α=0.4, iii) q_0=822

Table 2 Results from the analysis of the system for recycling and energy recovery of household plastic packaging waste in Trondheim in the year 2000, in a "best case" scenario. A linear relationship is assumed between environmental costs and emissions. Emissions consist of CO_2, NO_X and dioxins. Household costs are not included. Reported numbers are NOK per tonne plastic packaging waste. Results are reported in year 2000 prices.

Sector	Social marginal profit	Marginal profit	Marginal damage	Avoided marginal damage
Recycled LDPE replaces 90 % of production from virgin material				
Material recycling	997	-66	114	1177
District heating replaces energy from coal fired power plants (energy efficiency=40%):				
Energy recovery	-180	2045	3173	948
Results excluding dioxins:				
Energy recovery	2672	2045	321	948

Note: i) α=0.4, ii) q_0=822

As already mentioned, recycled LDPE replaces pallet blocks made from sawdust which means that no environmental externalities are avoided by using LDPE in the production of pallet blocks. District heating replaces energy produced by light oil. Let us instead investigate a "best case" scenario where recycled LDPE replaces 90 % of LDPE made from virgin material and district heating replaces energy made from a coal-fired power plant. What we find is a potential increase in the efficiency in the range of a factor of 5 and 4 for material recycling and energy recovery (including dioxin emissions), respectively (see Table 1 and Table 2). Behind these results lies avoidance of large CO_2 emissions since the input of energy in production from virgin material is far greater than in the recycling processes and that coal fired power plants emits more than three times as much CO_2 relative to incineration of plastic waste (NVE 1998). As Norway, during the coldest periods in wintertime, imports energy from Denmark produced by coal, the results reported in Table 2 suggest that during the winter season energy recovery is the most efficient way of treating the household waste, given that dioxin emissions can be reduced by 15 %. From Table 2 we also see that if the emissions of dioxins from the incineration process are reduced substantially, energy recovery is the efficient choice even if recycled LDPE replaces LDPE made from virgin material.

The results reported above do not account for the net utility in the households, which in general would be different for the different choices of treatment solution. Time spent on sorting may have an alternative cost, which can be positive or negative depending on whether the activity is done voluntary or not. If sorting is done to conform to social norms it can be argued that there is a cost rather than a positive net utility associated with sorting activities (Bruvoll and Nyborg 2002). This is not analyzed further, other than noting that the household cost per tonne that would make the two alternatives break even is NOK 1789 (if dioxins are included in environmental costs). What is more, data from Statistics Norway (SSB 2003) on the number of households in Trondheim and time spent on sorting (Bruvoll et al. 2000) combined with our data on amounts sorted in the households (Eik et al. 2002), we find that the alternative cost of time to make energy recovery a more efficient solution than material recycling on average is less than NOK 1 per hour. We would, however, like to stress that estimates on time consumption are associated with great uncertainty. Nevertheless, this shows how sensitive the results are to possible costs in the household sector related to waste processing.

The issue of centralization versus decentralization of waste treatment facilities can be investigated within our model. Total welfare increases if municipalities with relatively low marginal social profit deliver some of its waste to municipalities with high social marginal welfare. Calculations show that if the social marginal profit in a municipality is zero, the municipalities wanting to deliver waste to Trondheim for material recycling can be located as far away as 460 kilometers[13]. If the emission of dioxins is eliminated from the energy recovery process the maximum transport distance for waste to energy recovery exceeds 5000 kilometers[14]. That transport distances is a minor factor in determining the overall efficiency of treatment alternatives is consistent with other studies (STØ 1999; CIT Ekologik 1999; Heyde and Kremer 1999). Note that these figures are likely to overestimate the maximum distance since costs such as road accidents or noise are not accounted for, which can be substantial.

This section has shown that the treatment alternatives that are the most efficient depend heavily on four factors. These are i) the level of dioxin emissions from incineration of plastics and the valuation of dioxins emissions, ii) whether recycled materials actually replace products made from virgin material, iii) whether the energy produced by the incineration of plastic replaces energy produced by coal instead of light oil, and iv) whether we include household costs or not.

5.3 Increasing marginal environmental costs

Low emission levels of hazardous material may have relatively small and proportional effects, but responses may increase sharply and possibly jump discontinuously at higher emission levels, i.e. the marginal cost will increase for higher emission levels. The relevancy of considering convex external cost functions in this analysis is further based on Botterud (2000), which shows that the peak concentration levels of emissions from the energy recovery plant at Heimdal is located about 2 kilometers from the plant. This field consists of commercial as well as large residential areas, and the energy recovery plant is the single main source of dioxins. In addition, the emission of dioxins from the energy recovery plant exceeds limits on the concentration level for new plants set by the Norwegian Pollution Control Authority.

To gain some insights on the effect of non linear external costs, marginal environmental costs which increase with the level of emissions are introduced. Through risk assessment procedures such as the box model and the Gaussian plume model (see Turner 1994), we could identify the exposures at different locations surrounding the incineration plant, and next calculate hazard indices. Combined with data on population densities for different locations and a quantification of the value of a life year (VOLY) or an estimate of the value of a statistical life (VOSL) we would be able to assess costs as a function of emissions. However, given that there are uncertainties in the risk

[13] This is the distance that makes the marginal social welfare from material recycling equal to zero.
[14] This is the distance that makes the marginal social welfare from energy recovery equal to zero.

assessment, and that there is not any unique value on VOLY or VOSL, an alternative approach has been selected.

In order to construct an environmental cost function exhibiting increasing marginal environmental costs estimates of the marginal environmental cost from various studies have been used (see Appendix D for specific references). Given a choice of quadratic environmental cost functions, that emissions are proportional to production, benchmarking the marginal costs to the production levels in 2000 and by assigning a value to the elasticity of the environmental cost function, we end up with two equations with two unknown variables. The procedure can be illustrated by considering environmental costs, D, related to production, Q, given in Equation (6).

$$D(Q) = aQ + \frac{b}{2}Q^2 \qquad a,b > 0 \tag{6}$$

Marginal costs and the elasticity of the environmental cost function with respect to production is given in Equations (7) and (8).

$$\frac{\partial D}{\partial Q} = a + bQ \tag{7}$$

$$\varepsilon_{D,Q} \equiv \frac{\partial D}{\partial Q}\frac{Q}{D} = 2\frac{aQ + bQ^2}{2aQ + bQ^2} \tag{8}$$

Hence, we have enough information to determine the two parameters within our quadratic damage function.

Three scenarios are investigated using elasticities of the environmental costs with respect to production for the level in year 2000 in the range of 1.5, 2.0 and 2.5. Thus, at one end we have a scenario where, for the given level of emissions in 2000, a 1 % increase in production leads to an increase in damage of 1.5 %, and at the other end the same relative increase in production generates a 2.5 % increase in environmental costs[15].

An implication following that the environmental cost functions was estimated based on a benchmark situation where $\beta = 0.3$ is that for $\beta > 0.3$ the net marginal externalities from energy recovery is lower the higher the degree of convexity in the damage function, and vice versa for material recycling. This means that if the marginal social profit had been higher for material recycling relative to that of energy recovery

[15] These scenarios come in addition to the linear marginal environmental cost situation analyzed in the preceding section, which could be interpreted as a scenario with constant marginal environmental costs. It is ignored that the elasticities will generally differ between types of emissions. The modeling of the environmental cost functions is discussed in more detail in Appendix C. Note also that the avoided externalities from substituting products made from virgin material by recycled material and from substituting heating produced by incineration of oil with incineration of plastic waste are assumed to be linearly correlated with the corresponding emissions.

for $\beta = 0.3$ the efficient recycling rate will be higher than 0.3 but negatively correlated with the elasticity of the environmental cost function (for more on this see Appendix C). This should be kept in mind whenever we discuss efficiency for other levels of material throughput than what was the level in the year 2000.

Table 3 Results from the analysis of the system for recycling and energy recovery of household plastic packaging waste in Trondheim in the year 2000. The efficient rates of recycling (μ), capacity utilization in the central sorting plant (β), net marginal environmental costs (NOK) for each of the treatment alternatives, and marginal social profit for the system are all reported. Emissions consist of CO_2, NO_X and dioxins. Household costs are not included. Reported numbers are NOK per tonne plastic packaging waste. Costs and profit are reported in year 2000 prices.

Elasticity of the environmental cost function with respect to production	1.5	2.0	2.5
β	0.94	0.76	0.65
μ	0.37	0.30	0.26
Net marginal environmental costs for energy recovery	1634	1638	1636
Net marginal environmental costs for material recycling	-267	-263	-266
Marginal social profit of increase in q_0	-87	51	145
Results when recycled LDPE replaces 90 % of LDPE made from virgin material:			
β	1	1	1
μ (when α=0.4)	0.4	0.4	0.4
Net marginal environmental costs for material recycling	-1303	-1277	-1306
Results when recycled LDPE replaces 90 % of LDPE made from virgin material and $\alpha=\mu^*$:			
$\mu^*=\alpha$	0.87	0.63	0.54
Marginal social profit of increase in q_0	712	1235	1433

Note: i) α=0.4, ii) q_0=822, iii) μ^* denotes the efficient level of recycling for the overall system

Given our specification of the environmental cost function, the results are reported in Table 3. For a material input to the waste handling system equal to what we observed for 2000 we see that the share of plastic packaging waste that should be recycled is in the range of 26 % to 37 %, depending on the convexity of the damage functions. All these estimates are high compared to the 12 % that was actually recycled in 2000, which indicates that the municipality should consider increasing the production at the central sorting plant. A factor explaining why the recycling rate in Trondheim was much lower than what is suggested by our analysis is that part of the profit generated within the system goes to the producers of recycled material in Sweden and not the city of Trondheim.

From Figure 3 it is evident that the efficient rate of recycling increases with the amount of waste to be processed. The explanation for this relationship is the convex environmental cost functions and linear profit functions implying that the determination of the efficient recycling rate is based mostly on the economic factors for low levels of waste and relatively more on the environmental impacts as the amounts of waste flowing through the system increases. Hence, as profit from producing district heating is larger than that of material recycling and that the marginal environmental costs are lowest for material recycling, the efficient rate of recycling is positively correlated with the amount of waste. A result following from this is that the likelihood of the household sorting rate constraining implementing the efficient recycling rate increases with the amount of waste throughput in the system.

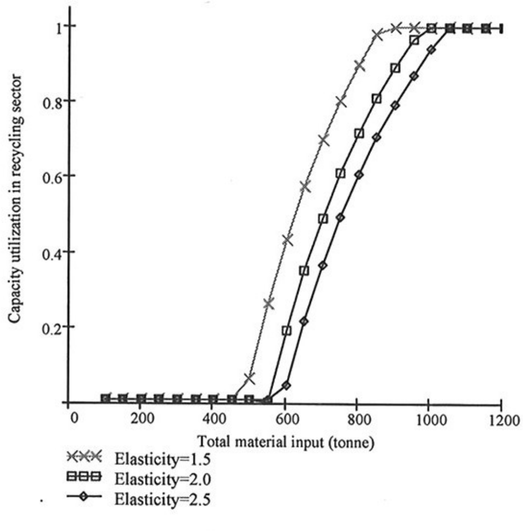

Figure 3 Different levels of capacity utilization in the recycling sector for varying levels of total material input to the waste treatment system. The three different curves illustrate three different levels of elasticity of the environmental cost function at the benchmark q_0=822. External costs from emission of dioxins are included.

As reported in Table 3, the efficient rate of recycling is lower the higher the elasticity of the externalities. In other words, the results of the analysis depend on to what degree marginal environmental costs increases on the margin. Physical surroundings of energy recovery plants differ, which means that the structure of the different plant's environmental cost function also differs. Hence, our analysis suggests that formulating general targets with specific numbers for material recycling and energy recovery is not necessarily efficient as local conditions vary.

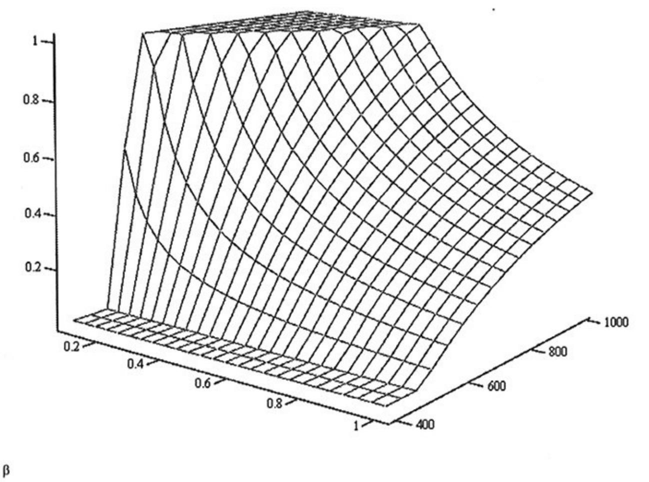

Figure 4 The relationship between efficient levels of capacity utilization in the recycling sector, β, (the vertical axis) and the household sorting rate, α, (the left horizontal axis) for varying levels of material input to the system measured in tonnes (the right horizontal axis). The diagram illustrates the 1.5 elasticity scenario.

The law of mass conservation makes it clear that the ratio of input of material into the recycling process to the output implies that $\beta k_M \leq 1$ (see equations (1) and (2) in Section 3). For our system we have $k_M = 1$, which means that the decision variable in the recycling sector cannot exceed one, i.e. $\beta = 1$ is equivalent to a corner solution with a 100 % utilization of the sector's production capacity. What is more, whenever the efficient level of the decision variable is equal to one, it signals that the household sorting rate is the upper limit for the rate of recycling. Since our estimates on the efficient levels of β are below unity it is clear that the household sorting rate is not a constraining factor. Consequently, given our assumptions, the manager of the waste handling system in the city of Trondheim, should not spend money on motivating households to put more effort into the sorting of their plastic packaging waste. Nevertheless, Figure 3 illustrates that within the 1.5 elasticity scenario a 5 % increase in the amount of waste to be processed in the system would be sufficient for the household sorting rate to constrain the recycling rate. On the other hand, in the 2.5 elasticity scenario the sufficient increase in waste inflow is 27 %. Two conclusions can be drawn from this. First, the valuation of environmental externalities greatly affects the efficient rate of recycling. Second, the positive correlation between recycling rates and generated amounts of waste implies that without dematerialization, economic growth should be followed by an increase in household waste handling efforts.

In addition to identifying the relationship between capacity utilization in the material recycling sector and the amount of waste entering the system, our model also enables identification of the relationship between combinations of household sorting rates, amounts of waste to be processed and the efficient recycling rate (see Figure 4). Since the recycling rate is the product of α and β, efficient rates of recycling for different levels of material input are seen as curves going from north-west to south-east in the diagram, i.e. they are iso-recycling rate curves. The higher the iso-recycling curves are positioned in the diagram the higher the recycling rate. It is further clear that the level of household sorting does not influence the recycling rate other than when it constrains its implementation. The reason for this is that there are no environmental externalities related to the household sorting activity. There is however a trade off between α and β, which means that if the regulator can make the households increase their sorting at a low cost he or she would prefer this strategy rather than increasing the use of labor at the recycling plant, when the goal is to increase the recycling rate. With information on the net utility of household sorting the presented method enables a better understanding of where the effort to increase recycling should be made.

From a regulator's point of view, a problem can arise when households sort a low fraction of a large amount of generated waste. In this situation the household effort is below the minimum level consistent with the efficient rate of recycling. In order to implement the efficient solution the regulator must try to encourage households to increase their level of sorting efforts. However, for households to put more effort into sorting their waste than what is efficient from their isolated point of view, they must be compensated. A planner would know that there is a limit to how much households should be compensated, namely the marginal social profit from a unit increase in recycled material. Using a conventional CBA only the social marginal profit for a given rate of recycling can be calculated since the environmental profile of the system may change as the flow of material changes. However, the approach used in this paper enables us to find information about the upper limit of a subsidy for varying levels of household sorting and material input to the system. We offer no further elaboration on this other than to report that for the situation in 2000 the limit for a subsidy ranged from NOK 1284 to NOK 1306 per tonne for the three scenarios.

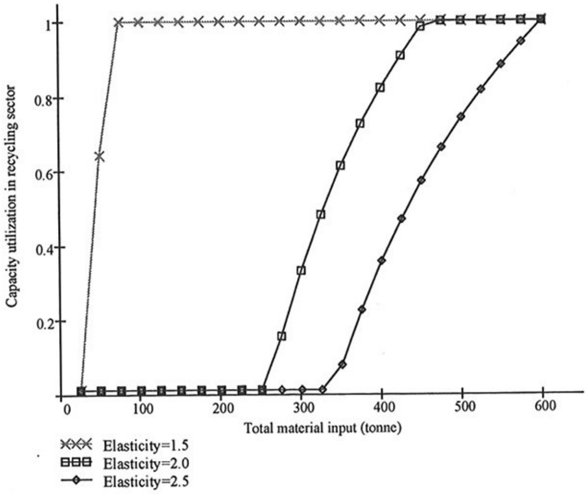

Figure 5 Different levels of capacity utilization in the recycling sector for varying levels of total material input to the waste treatment system when recycled LDPE replaces 90 % of LDPE made from virgin material. The three different curves illustrate three different levels of elasticity of the damage function at the benchmark level q_0=822. External costs from emission of dioxins are included and α=0.4.

As in Section 5.3 we can consider a situation where recycled LDPE replaces 90 % of LDPE made from virgin raw material. From Table 3 and Figure 5 it is clear that for all three scenarios the efficient solution is a corner solution: β=1, with a corresponding recycling rate equal to the household sorting rate 0.4[16]. The potential for material recycling related to recycled LDPE substituting LDPE made from virgin material, can be illustrated by that the efficient household sorting rate, from a regulator's point of view should be set equal to 0.87, 0.63 and 0.54, respectively, for our three scenarios. The three resulting recycling rates in this scenario are more than twice the efficient recycling rate found for the present situation where LDPE replaces pallet blocks made from sawdust. From Table 3 it is also clear that the marginal social profit of material input to the system is exceeding the social profit in the actual situation in year 2000 in a range from around NOK 800 to NOK 1300 depending on the degree of convexity of the environmental damage functions. The conclusion must therefore be that it is very important to find the best possible use of the recycled material if recycling is to be the preferred waste handling alternative in the future.

[16] Recall that in 2000 q_0=822 tonnes.

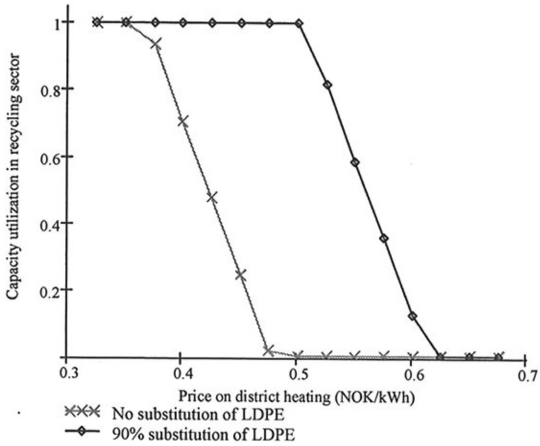

Figure 6 The relationship between the price of district heating (kWh) and the efficient capacity utilization in the recycling sector (β). It is assumed an elasticity of 1.5 in the damage functions. Emission categories are CO_2, NO_X and dioxins. The two curves marked with x's and diamonds illustrate situations where recycled LDPE replaces 0 % and 90 % of LDPE made from virgin material, respectively.

The analysis has shown that whether or not recycled LDPE replaces LDPE made from virgin material has a large influence on the results of the analysis. In other words, the relative prices of the produced goods are important. All else being equal, a price of electricity higher than NOK 0.45 per kWh is sufficient for energy recovery to be the efficient way of handling the plastic packaging waste (see Figure 6), whereas material recycling is the efficient solution for electricity prices lower than NOK 0.34 per kWh. These results therefore indicate that material recycling and energy recovery would be most efficient in the summer and in winter, respectively. The efficient mix of material recycling and energy recovery is quite sensitive to the alternative value of the two products as an "interior" solution, i.e. that it is efficient to have a mix of the two treatment alternatives, is found within a range of only NOK 0.11 per kWh. All our results must therefore be interpreted with this in mind. Another result illustrated in Figure 6 is that whether recycled LDPE replaces LDPE made from virgin material or not greatly affects the relationship between the price of district heating and efficient capacity utilization in the recycling sector.

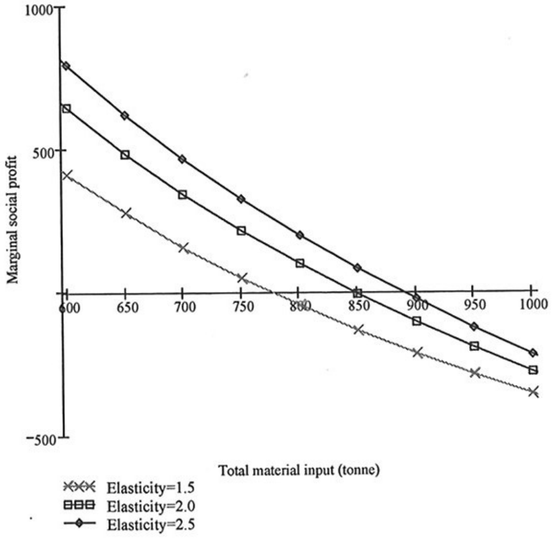

Figure 7 Marginal social profit (NOK) for different levels of material input to the system measured in tonnes. Three scenarios are illustrated with elasticities of 1.5, 2.0 and 2.5 for the benchmark situation q_0=822.

As in the previous section centralization of waste handling services should be carried out if the marginal social profit from an increase in the total amount of waste to be processed differs between municipalities. Due to the convex environmental cost functions the marginal social profit is negatively correlated to the amount of waste that must be processed within the system (see Figure 7). For the amount of waste that was processed in year 2000 the marginal social profit is negative for the elasticity of 1.5 scenario but positive for the two others, which is partly a corollary following our construction of the environmental cost functions (see Table 3 and Appendix C). That the social profit is larger the higher the elasticity of the damage function is a result of the reduction in environmental costs of energy recovery being larger than the increase in environmental cost of material recycling (see Figure A.1 in Appendix C). This holds as the efficient level of β is higher than the benchmark level. If the efficient level of β is lower than the benchmark level, the relatively large increase in environmental costs following an increase in the elasticity of the environmental cost function more than counterbalances the reduction in environmental costs of material recycling. Hence, the marginal social profit is higher in the 2.5 elasticity scenario than in the 1.5 elasticity scenario regardless of β being higher or lower than the benchmark level.

These aspects have implications for the effectiveness of decentralization, which depends critically on how the environmental impact changes for increasing levels of emissions. For instance, it is clear from Figure 7 that for an elasticity of 2.5 in the environmental cost functions, the increase in the amount of material to be processed should not exceed 8 % of the amount processed in the year 2000. This result is based on a zero marginal social profit in the municipality from where the material originates. A higher (lower) percentage increase in the received amounts of waste is efficient if the marginal social profit in the municipality delivering the waste is lower (higher) than zero.

As noted earlier, due to improved abatement technology the emission of dioxins will probably be a smaller problem compared to what it has been, which implies that more plastic packaging waste should be incinerated. Moreover, a clear result from our analysis is that a 40 % reduction in emissions of dioxins from energy recovery is sufficient for material recycling to be the least efficient alternative for waste handling for all relevant levels of plastic packaging waste. But here we cannot rule out that there will be technological improvements that lead to significant reductions in the environmental impacts from the recycling processes. It is issues like these that make it so difficult to determine what the efficient strategy for waste handling should be in the future. There is also the inherent danger of an evaluation made today leading to an undesired technological lock-in in the future.

6. CONCLUDING REMARKS

Applying different methods to the same system for waste handling will often lead to different recommendations concerning the efficient rate of material recycling. The focal point of the methods are different and consequently, the various aspects of the system are emphasized differently by researchers. The idea behind the present analysis shares the view of Wriesberg and Udo de Haes (2002) and Bouman et al. (2000) that in order to the have a balanced view of as many relevant factors possible, a combination of methods is the best practice. Here, this is done by combining material flow accounting with production functions within a cost-benefit framework. Even though the production processes within our system are characterized as linear technology processing, sufficient computational power is available enabling an evaluation to be made of the changes in environmental and economic impacts following changes in the efficient use of input factors within systems with nonlinear production technology.

The main contribution from the presented method, which is based on tracking the flow of materials through the system, is the capability of highlighting interdependencies between the different processes within the system. Developing this approach by including household behavior, a greater number of emissions and all of the waste fractions will result in a model that is better to use as a basis for discussions of the efficient future waste handling system and designing efficient policy measures than conventional cost-benefit and life cycle analysis. That the approach used in this analysis is a fruitful one is obvious as one cannot base the design of future waste handling policies for a world in continuous change on analyses studying a given level of material throughput. Only when models reflect the impacts of varying levels of material input and household sorting rates, changes in product prices and technological improvements can policy measures be designed that are suitable for the purpose.

The main results from the analysis are in accordance with the results of the other studies of the waste handling system in Trondheim. That is, energy recovery is the most economically interesting handling strategy, whereas material recycling may in the future be the prudent choice since the environmental gains from replacing products made from virgin material have considerable potential. Since technologies that reduce emissions from the recycling and energy recovery processes may develop asymmetrically, it is very difficult to predict which of the waste treatment alternatives will be the most efficient.

Another conclusion is that the choice or mixes of handling strategies is heavily influenced by the monetary valuation of emissions, the type of technology applied in the production processes, what type of energy production that energy recovery replaces and whether we include household costs or not. Because of heterogeneity in the physical surroundings and in the type of technology employed in the production processes in the different recycling and incineration plants one should be careful when it comes to stating general prescriptions on waste handling strategies at a national level.

The waste treatment system was partly analyzed under the assumption of increasing marginal environmental costs. How to construct an environmental cost function exhibiting this characteristic is a general problem that should be given more attention. The task of further developing adequate environmental cost functions reflecting increasing marginal costs is left for future research.

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APPENDIX

Appendix A Alternative methods for economic and environmental assessment

In addition to CBA and LCA there are a number of methods frequently used but not necessarily well known to economists. The reason for presenting some of them here are that environmental problems are interdisciplinary and hence, that it is valuable to have some knowledge about other methods.

Material Input per Unit of Service (MIPS) as a method consist of two components, namely material input and service unit (Schmidt-Bleek 1994). The first comprises all man-moved materials that are required in the life-cycle of a product or service, which are aggregated to five categories, i.e. abiotic raw materials, biotic raw materials, soil, water, and air. The material input may be distinguished with respect to different phases of the life cycle: production, usage, repair, recycling or disposal. The second component of the MIPS approach is to assess the utility or function that can be obtained from a product. For non-durable goods such as food and beverages, the service unit is equal to the product unit in kg., liter etc., whereas for durable goods such as furniture and cars, the unit of measurement is usable years or 1 person-km.

Compared to LCA MIPS is easier to perform and communicate since it evaluates only the direct material impacts, hence it is useful for monitoring progress in the process of dematerialization. However, looking only at the input side of the system and using the weight of materials as proxy for environmental impacts are the weaknesses of this tool.

The aim of an Environmental Risk Assessment (ERA) is to examine the risk resulting from technology that threatens ecosystems, animals and people (EEA 1998). ERA is based on hazard identification where the relationship between different levels of exposure and the incidence and severity of effects are studied. Next, an effects assessment is carried out, which means that the predicted no-effect level is determined. Lastly, an exposure assessment is carried out by which predicted exposure concentration or total daily intake are determined in order to describe the nature and size of exposed targets, as well as the magnitude and duration of the exposure.

Like LCA ERA is also very data intensive and hence time and resource demanding. In addition there are significant uncertainties surrounding the risk of adverse health effects for humans. On the other hand, it is probably the best available tool for assessing risks on human health and ecosystems due to emission of hazardous substances.

The tools discussed thus far do not include economic or social costs and benefits. One that does this is the method of Life Cycle Costing (LCC) which aims at describing internal and external costs during the life cycle of a product. This is done by calculating the costs associated with emissions and resource use based on a life cycle inventory and on environmental and human health effects quantified in impact assessments (Evans 1998; Westkaemper 1998). Hence, in an LCC all impacts must be expressed in monetary terms and methods used to assess these are contingent valuation, hedonic pricing, and regulator's revealed preferences (Curran 1996).

Appendix B Deriving efficient rates of recycling

Social welfare is defined as $W = \Pi_M + \Pi_E - D_M - D_E + S_M + S_E - T$. Consider the recycling sector first. Generally we define profit in this sector as $\Pi_M = R_M - C_M$, which consists of revenues defined as unit price times the amount of recycled material produced: $R_M = P_M q_M$, and costs defined as $C_M = q_M \left(wl_M / k_M + P_E I_M \right)$. l_M is the Leontief input coefficient for labor, while l_M / k_M is the fixed input coefficient in the material recovery process controlled for technological level in production process, and w is the uniform wage rate. I_M is the per unit use of energy in the production process, and P_E is the unit price on electricity. Thus, $\Pi_M = P_M q_M - wl_M / k_M$. Let u=0,1,2,3,4 denote the processing plant, where 0=Heimdal, 1=Tydal, 2=Arvika, 3=Töcksfors, 4=Karlstad, and i=1,2,3 denote the type of emission, where 1=CO_2, 2=NO_X and 3=dioxins. We can now formulate the profit function for the recycling sector as:

$$\Pi_M = \alpha\beta q_0 k_0 \left\{ P_1 k_1 j_1 + j_2 \sum_{u=2}^{3} P_u k_u v_u - w_N \left(l_0 + l_1 j_1 k_1 \right) - w_S j_2 k_4 \left(l_4 + \sum_{u=2}^{3} l_u k_u v_u \right) \right.$$
$$\left. - P_E \left[I_0 + I_1 j_1 k_1 + j_2 k_4 \left(I_4 + \sum_{u=2}^{3} I_u k_u v_u \right) \right] \right\}$$

where $(1-k_M)\alpha\beta q_0$ and $(1-k_E)(1-\alpha\beta) q_0$ are emissions from the recycling and energy recovery processes, respectively. $a,b,g,h > 0$, where a and g is measured as NOK per tonne, b and h is measured as NOK per (tonne)^2. In addition there are external costs related to the emissions from production of electricity which is needed in the material recycling process. w_N and w_S are the unit labor cost in Norway and Sweden, respectively.

In the energy recovery sector we have that profit is given as $\Pi_E = R_E - C_E$. Variable costs associated with incineration of plastic waste are identified as $C_E = wl_E q_E / k_E e$, where l_E is the Leontief input coefficient in the process of incineration and k_E is the ratio between energy produced at the plant and energy delivered to the end user. Revenue in this sector is generated according to $R_E = P_E q_E$,

where P_E is the market price per unit, typically kWh, district heating, q_E. Based on this we formulate $\Pi_E = P_E q_E - w l_E q_E / k_E e$. The input factor besides labor (L_E) is the residual material from both household sorting and production of recycled products: $q_E = k_E e(q_0 - q_M / k_M)$ where $q_0 - q_M / k_M$ is the amount of waste that is incinerated in the process of producing district heating, k_E is the environmental efficiency in the process of energy recovery, and e is the energy content per tonne waste. The specific function for the system under investigation is thus given as:
$\Pi_E = (1-\alpha\beta) q_0 \left(P_E (k_E e - I_E) - w_N l_E\right)$.

Emissions from the production processes are generally described as $E_M = E_M(\alpha, \beta, q_0; k_M)$ and $E_E = E_E(\alpha, \beta, q_0; k_E)$, which generates environmental damage and health risk according to $D_M = D_M(\alpha, \beta, q_0; a, k_M)$ and $D_E = D_E(\alpha, \beta, q_0; g, k_E)$. a and g represent the monetary valuation of the damage from emissions. Given linear production technology the external costs from the recycling process are represented as:

$$D_M = a(1-k_M)\alpha\beta q_0 + \frac{b}{2}\left((1-k_M)\alpha\beta q_0\right)^2$$

$$D_E = g(1-k_E)(1-\alpha\beta) q_0 + \frac{h}{2}\left((1-k_E)(1-\alpha\beta) q_0\right)^2$$

Let I_M denote the electricity input coefficient and z denote NOK per tonne and y NOK per (tonne)^2. The specific equations are given as:

$$D_M = a q_M \frac{K_M}{k_M} + \frac{b}{2}\left(q_M \frac{K_M}{k_M}\right)^2 + z q_M I_M + \frac{y}{2}(q_M I_M)^2$$

$$= q_M \left(\frac{K_M}{k_M}\left(a + \frac{b}{2} q_M \frac{K_M}{k_M}\right) + I_M\left(z + \frac{y}{2} q_M I_M\right)\right)$$

The environmental cost related to the recycling process when all three emission categories are accounted for can be calculated as:

$$\begin{aligned} D_M = \alpha\beta q_0 \sum_{i=1}^{3} \Bigg\{ & a_{0,i}(1-k_0) + j_1 a_{1,i} k_0 (1-k_1) + j_2 k_0 \left(a_{4,i}(1-k_4) + \sum_{u=2}^{3} a_{u,i} v_u (1-k_u)\right) \\ & + \frac{\alpha\beta q_0}{2}\left[\left(b_{0,i}(1-k_0)\right)^2 + \left(j_1 b_{1,i} k_0 (1-k_1)\right)^2 + j_2^2 k_0 \left(\left(b_{4,i}(1-k_4)\right)^2 + \sum_{u=2}^{3}\left(v_u b_{u,i}(1-k_u)\right)^2\right)\right] \\ & + k_0\left[z_{0,i} I_0 + j_1 z_{1,i} k_1 I_1 + j_2 k_4 \left[z_{4,i} I_4 + \sum_{u=2}^{3} z_{u,i} v_u k_u I_u\right]\right] + \frac{\alpha\beta q_0}{2} k_0 \Big[y_{0,i} I_0^2 + y_{1,i} j_1^2 k_1^2 I_1^2 \\ & + (j_2 k_4)^2 \left[y_{4,i} I_4^2 + \sum_{u=2}^{3} y_{u,i} v_u I_u^2\right]\Big]\Bigg\} \end{aligned}$$

where j_1 and j_2 denote the share of the material sorted for processing in Tydal and

Karlstad, respectively. v_2 and v_3 denote the share of the material coming to the sorting facility in Karlstad that is processed further in Arvika and Töcksfors, respectively. Let I_E denote the input coefficient of electricity in the incineration process and g and h denote the monetary valuation measured as NOK per tonne and NOK per (tonne)^2, respectively. the subscripts 1 and 2 indicate that the emissions are generated by the material input and electricity use (I_E), respectively. The environmental costs in the energy recovery process are then given as:

$$D_E = g_1(1-\alpha\beta)q_0(1-k_E) + \frac{h_1}{2}((1-\alpha\beta)q_0(1-k_E))^2 + g_2(1-\alpha\beta)q_0 I_E + \frac{h_2}{2}((1-\alpha\beta)q_0 I_2)^2$$

$$\Rightarrow D_E = (1-\alpha\beta)q_0\left[(1-k_E)\left(g_1 + \frac{h_1}{2}(1-\alpha\beta)q_0(1-k_E)\right) + I_E\left(g_2 + \frac{h_2}{2}I_E(1-\alpha\beta)q_0\right)\right]$$

The environmental cost related to the recycling process when all three emission categories are accounted for can be calculated as:

$$D_E = (1-\alpha\beta)q_0\sum_{i=1}^{3}\left[(1-k_E)\left(g_{1,i} + \frac{h_{1,i}}{2}(1-\alpha\beta)q_0(1-k_E)\right) + I_E\left(g_{2,i} + \frac{h_{2,i}}{2}I_E(1-\alpha\beta)q_0\right)\right]$$

1 tonne of recycled material replaces 90 % of plastic material produced from virgin raw material (except for LDPE). The external effects arising from production using virgin material inputs are subtracted from the external effects from the material recycling process in order to calculate the net environmental costs. The avoided external effects in the material recycling sector are calculated according to $S_M = \alpha\beta q_0 k_0 \sum_{i=1}^{3}\left(j_1 k_1 s_{1,i} + j_2 k_4 \sum_{u=2}^{3} v_u k_u s_{u,i}\right)$ where $s_{u,i}$ denote the monetary valuation of the avoided externalities, measured as NOK per tonne. District heating substitutes electricity produced by the use of light oil and the avoided external costs are represented by:

$S_{E,i} = (1-\alpha\beta)q_0 e k_E s_{E,i}$ where $s_{E,i}$ denote the monetary valuation of the avoided externalities, measured as NOK per tonne.

Costs associated with transporting the material between production sites is calculated based on the following:

$$T = t\alpha q_0\left\{\beta k_0\left[d_{0,1}j_1 + j_2\left(d_{0,4} + k_4\sum_{u=2}^{3} d_{4,u}v_u\right)\right] + (1-\beta)d_{0,E}\right\}$$

where t denotes the costs in NOK per kilometer per tonne, and d denotes the distances between the different locations which are represented by the subscripts.

Inserting all specific functions into the social welfare function the first order condition for maximum welfare states that the efficient rate of recycling is found according to:

$$\frac{\partial W}{\partial \beta} = 0 \Rightarrow$$

$$\begin{aligned}
\beta^* = \frac{1}{\eta}\sum_{i=1}^{3}\Bigg[& P_1 k_1 j_1 + j_2 \sum_{u=2}^{3} P_u k_u v_u - w\left(l_0 + l_1 j_1 k_1\right) - w_S j_2 k_0 k_4 \left(l_4 + \sum_{u=2}^{3} l_u k_u v_u \right) \\
& - P_E k_0 \left(I_0 + I_1 j_1 k_1 + j_2 k_4 \left(I_4 + \sum_{u=2}^{3} I_u k_u v_u \right)\right) - P_E\left(k_E e - I_E\right) + w l_E - a_{0,i}\left(1-k_0\right) - j_1 a_{1,i}\left(1-k_1\right) \\
& - j_2 \left(a_{4,i}\left(1-k_4\right) + \sum_{u=2}^{3} a_{4,i} v_u \left(1-k_u\right)\right) - k_0 \left(z_{0,i} I_0 + j_1 z_{1,i} k_1 I_1 + j_2 k_4 \left(z_{4,i} I_4 + \sum_{u=2}^{3} z_{u,i} v_u k_u I_u \right)\right) \\
& + \left(1-k_E\right)\left(g_{1,i} + h_{1,i} q_0 \left(1-k_E\right)\right) + I_E\left(g_{2,i} + h_{2,i} I_E q_0\right) + k_0 \left(j_1 k_1 s_{1,i} + j_2 k_4 \sum_{u=2}^{3} v_u k_u s_{u,i} \right) - e k_E s_{E,i} \\
& - t\left(k_0 \left(d_{0,1} j_1 + j_2 \left(d_{0,4} + k_4 \sum_{u=2}^{3} d_{4,u} v_u \right)\right) - d_{0,E} \right)\Bigg]
\end{aligned}$$

$$\begin{aligned}
\eta = \alpha q_0 \sum_{i=1}^{3}\Bigg[& \left(b_{0,i}\left(1-k_0\right)\right)^2 + \left(j_1 b_{1,i}\left(1-k_1\right)\right)^2 + j_2^2 \left(\left(b_{4,i}\left(1-k_4\right)\right)^2 + \sum_{u=2}^{3}\left(v_u b_{u,i}\left(1-k_u\right)\right)^2 \right) \\
& + k_0^2 \left(y_{0,i} I_0^2 + y_{1,i} j_1^2 k_1^2 I_1^2 + j_2^2 k_4^2 \left(y_{4,i} I_4^2 + \sum_{u=2}^{3} y_{u,i} v_u I_u^2 \right)\right) + h_{1,i}\left(1-k_E\right)^2 + h_{2,i} I_E^2 \Bigg]
\end{aligned}$$

Appendix C Constructing increasing marginal environmental cost functions

The chosen procedure to construct the environmental cost functions with increasing marginal costs is explained in Section 5.3. However, the way the functions are constructed has some implications worth discussing in more detail.

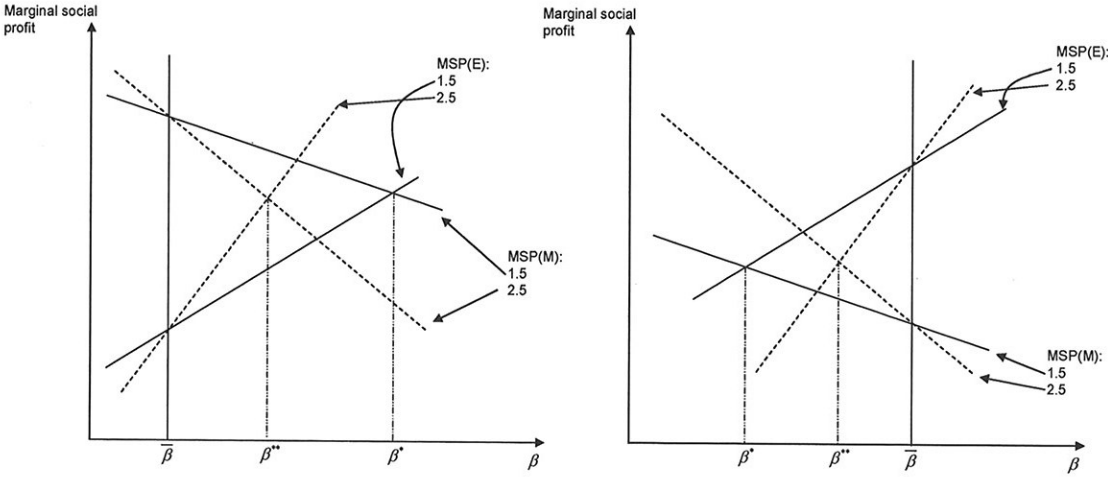

Figure A.1 Illustration of how the environmental cost functions are constructed. MSP(M) and MSP(E) stands for marginal social profit for material recycling and energy recovery, respectively. 1.5 and 2.5 illustrates two of our three scenarios for the degree of convexity in the functions. $\overline{\beta}, \beta^{*}, \beta^{**}$ denote benchmark level, efficient level in the 1.5 and 2.5 elasticity scenarios, respectively. Remember that an increase in the production of district heating is the same as a reduction in β.

To identify the relationship between the chosen elasticity of the environmental cost function and the efficient recycling rate three important factors must be considered:

- The material must either be recycled or incinerated, that is, increasing the production in the material recycling sector generates a reduction in the production of district heating.
- Given linear marginal profit it is evident that the larger the elasticity of the environmental cost function, the larger the change in marginal social profit from one production level to another (see Figure A.1).
- The production level of the processing alternative with the highest (lowest) social marginal profit is increased (decreased) relative to the benchmark production level used in estimating the cost function (see Figure A.1).

From the three points above it is evident that the chosen elasticity underlying the estimation of the environmental cost function influences the efficient rate of recycling.

More specifically, the processing alternative with the lowest (highest) marginal social profit at the benchmark production level, benefits (suffers) from an environmental cost function with a higher (lower) elasticity (see Figure A1). This is evident in our analysis by looking at Table 1 where material recycling has a positive marginal social profit whereas energy recovery has a negative marginal social profit. Comparing this to the result reported in Table 3, we see that the efficient rate of recycling decreases with the elasticity of the environmental cost function, but always stays higher than the benchmark level, which is 12 %.

Another point that also should be mentioned is how the choice of benchmark level and elasticity of the environmental cost function together influences the estimated efficient recycling rate. Moreover, the higher the elasticity the closer the estimated recycling/recovery rates will be to the benchmark rate. The mechanism behind this relationship is that an increase in the elasticity of the environmental cost function changes the slope of the marginal social profit functions in opposite directions. Due to the convex environmental cost functions, we see from the leftmost diagram in Figure A.1 that to the right of the benchmark level the marginal social profit of the processing alternative that was the highest in the benchmark situation is reduced, whereas it increases for the alternative process for higher elasticities of the environmental cost functions. This means that the difference in marginal social profit between the processing alternatives decreases with the elasticity of the environmental cost function. Hence, there is less room for large changes in the efficient mix of the alternative processes.

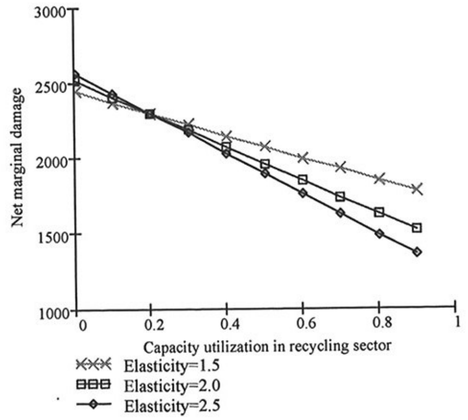

Figure A.2 The vertical axis shows marginal environmental costs from producing district heating (NOK) for elasticities of the damage function with respect to production equal to 1.5 (solid curve), 2.0 (dashed curve) and 2.5 (dotted curve). β is measured along the horizontal axis. $\alpha = 0.4, q_0 = 822$. Environmental costs are related to emissions of CO_2, NO_X and dioxins.

Figure A.2 shows the marginal environmental costs for energy recovery. For β=0.3 the marginal cost is the same for each of the values assigned to the elasticity of the damage function. Increasing production of district heating means a reduction in β, i.e. a movement to the left in Figure A.2, for which the marginal cost is highest in the scenario with the highest elasticity. Since the damage function is specified as a quadratic one, we have that to the right of β=0.3 the marginal cost is the highest for the lowest elasticity.

Appendix D Data

Description	Value	Unit	Source
Potentially recyclable plastic waste generated in households	822	Metric tonne.	Eik et al. (2002)
Labor cost in: Norway Sweden	 192 182	NOK per hour Exchange rate NOK/SEK:0.95	RG-Prosjekt (2000) (Olaussen (2002))
Price per unit recycled material: Pallet block PP-regranulate HDPE regranulate	 944 3800 4560	NOK per tonne Exchange rate NOK/SEK:0.95	Eik et al. (2002)
Price per unit district heating (kWh)	0.35	NOK per kWh	Olaussen (2002)
Input coefficient at: Heimdal, central sorting Tydal Karlstad Arvika Töcksfors	 3.6 2.3 2.3 2.3 3.6	Man hours per tonne production.	Olaussen (2002) Eik (2002)
Heimdal, energy recovery	0.345	Man hours per tonne production	Fossum (2002)
Technical efficiency in the production of district heating	0.675	$\in \langle 0,1 \rangle$	Olaussen (2002) Eik (2002)
Energy intensity in: Heimdal, central sorting Tydal Karlstad Arvika Töcksfors Heimdal, energy recovery	 3 1800 500 500 3 9	Energy use (kWh) per tonne output (input in the energy recovery sector)	Eik et al. (2002), Olaussen (2002
Energy content per unit plastic: At the plant At the consumer	 8.95 6.0	MWh per tonne at the plant	Sandgren et al. (1996) Olaussen (2002)
Emission of CO_2: From incineration of plastic waste From light oil fired power plants From coal fired power plants	 0.332 0.378 0.787	Tonne per MWh	Lindholt (1998) Sandgren et al. (1996) NVE (1998)

Data, continued:

<table>
<tr><th>Description</th><th>Value</th><th>Unit</th><th>Source</th></tr>
<tr><td>Emission of NO_X:
From incineration of plastic waste
From coal fired power plants
From light oil fired power plants</td><td>
0.0004
0.00025
0.00097</td><td>Tonne per MWh</td><td>NVE (1998)
Sandgren et al. (1996)</td></tr>
<tr><td>Emission of dioxins:
From incineration of plastic waste
From light oil fired power plants</td><td>
7.3x10E-6
6.4x10E-10</td><td>
Gram per tonne
Kg per GJ</td><td>
Econ (2000)
Sandgren et al. (1996)</td></tr>
<tr><td>Gross energy required to produce 1 kg of:
LDPE film
HDPE film
PP (injection molding)</td><td>
91.98
79.92
118.94</td><td>MJ</td><td>APME (2000)</td></tr>
<tr><td>Cost associated with transport (incl. tax and labor)</td><td>0.366</td><td>NOK per km per tonne</td><td>Eik et al. (2002)</td></tr>
<tr><td>Distances between plants</td><td><table><tr><th></th><th>Heimdal sorting</th><th>Karlstad sorting</th></tr><tr><td>Energy recovery</td><td>1</td><td></td></tr><tr><td>Tydal</td><td>120</td><td></td></tr><tr><td>Karlstad sorting</td><td>606</td><td></td></tr><tr><td>Arvika</td><td></td><td>81</td></tr><tr><td>Töcksfors</td><td></td><td>121</td></tr></table></td><td>Km.</td><td>Eik et al. (2002)</td></tr>
<tr><td>Share of sorted material from:
Heimdal to Tydal
Heimdal to Karlstad</td><td>
0.712
0.288</td><td>$\in [0,1]$</td><td>Eik et al. (2002)</td></tr>
<tr><td>Share of sorted material from Karlstad to:
Arvika
Töcksfors</td><td>
0.6
0.4</td><td>$\in [0,1]$</td><td>Eik et al. (2002)</td></tr>
<tr><td>External costs:
CO_2
NO_X
Dioxins</td><td>
130
15
2.3x10E9</td><td>NOK per tonne</td><td>ECON (2000)</td></tr>
</table>

Paper 4

A safety rule approach to pollution control*

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Abstract

The aim of this paper is twofold. First, it examines the impact of a regulatory safety rule on a dynamic model of pollution accumulation and control. Second, it investigates the cost of subordinating matters to a safety rule and how the cost is affected by the variables in the model. Pollution accumulation and control are modeled as a social planner's problem in which a risk-neutral planner allocates a fixed flow of resources to consumption or pollution control. The tradeoffs between consumption and pollution abatement are investigated under assumptions of linear and quadratic natural assimilative capacity of the environment. One conclusion is that the cost of safety is dependent on the nature of the assimilative capacity of the environment. Moreover, the implementation of a safety rule may lead to an increase as well as a decrease in steady-state consumption levels. A more alarming result is that nonlinear decay functions combined with uncertainty about health risks can generate a scenario where a steady-state or an efficient pollution control path does not exist. The policy implication is that some kind of safety measure must be applied in the presence of uncertainty.

Keywords: pollution control, consumption, risk, uncertainty, natural decay, safety rule, optimal control

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1. INTRODUCTION

Assessments of the relationship between pollution and human and ecological health are subject to errors and the Neumann-Morgenstern-Savage expected utility theories of decisions under uncertainty argue that we can rely on expected outcomes to efficiently allocate resources. However, available evidence indicates that decision-making under uncertainty systematically departs from the predictions of expected utility theory. A descriptive critique of the expected utility approach called prospect theory argues, *inter alia*, that people underweigh outcomes that are merely probable in comparison with outcomes that are obtained with certainty. The implication is that we will see a tendency towards risk aversion in choices involving sure gains and risk seeking in choices involving sure losses (Kahnemann and Tversky [12]). Instead of gathering information and constructing unbiased estimators people instead tend to use heuristics. Alternative theories explaining some of the same observations are found in Quiggin [20], Loomes and Sugden [17], and Machina [18].

Another, and to some extent, similar approach to the problem of uncertainty is to argue that when errors to be avoided are associated with relatively high costs, for instance in the form of environmental catastrophes, action must precede knowledge. Furthermore, even if we knew the relationship between *present* pollution levels and health risk it would be difficult to achieve an optimal resource allocation since optimality requires that marginal utility equals marginal damage in the *optimal* point (Baumol and Oates [2]). For some problems society must trade abatement efforts off against increased economic activities. Consequently, society must clarify and include its values into decision making, that is, issues related to uncertainty must be determined. More specifically, in the analysis to be presented the regulators must consider the maximum level of health risk that we are willing to accept and how certain we would like to be that this upper limit is not exceeded.

A safety rule as proposed by Lichtenberg and Zilberman [15] is used to analyze the control of stock pollutants over time. The rule is then applied to a dynamic model for pollution control based on earlier literature including Plourde [19], Smith [23], Keeler et al. [13], Cropper [7], Forster [9], [10], and Clarke and Reed [6]. The safety rule is a way of treating environmental problems surrounded not only by risk but also uncertainty where a margin of safety is an indicator of how much weight is put on uncertainty by the decision maker. Note that we define risk as the probability that an adverse health effect is inflicted on an individual randomly selected from a population, where uncertainty is the magnitude of error associated with the health risk estimate.

Given this, the novelty of this analysis is mainly to show how a safety rule will affect the allocation of resources over time for stock pollutants, and how it can both promote and preclude stable equilibriums.

Safeguarding against unwanted and potentially very costly outcomes carries a cost in terms of foregone utility from required reductions in the level of economic activity. This cost can however be seen as paying an insurance premium in an uncertain world. Depending on the degree of risk aversion we pay insurance premiums in order to protect our health, our children and our physical possessions such as our car. It is of course *a priori* difficult to evaluate whether we should or should not insure our car, but generally people pay what they think of as a reasonable insurance premium and are happy that they did not crash their car during that period. So in addition to investigating how a particular variant of the precautionary approach affects efficient levels of pollution control we also explicitly explore how different factors affect the size of the insurance premium. The latter being an interesting issue since there are limits to acceptable insurance premiums.

The main result from this analysis is that decision makers worrying about uncertain effects from economic activity generally must trade off consumption for margins of safety along the optimal path towards steady state. However, in situations with nonlinear natural decomposition of pollution we show that the long-term levels of consumption can be positively correlated with the margin of safety. This happens whenever the natural decay mechanism becomes more effective as pollution is lowered. A rather alarming, and more important, result is that the combination of high discount rates and uncertainty about health risk effects may lead us into a scenario where a steady state, and hence an efficient dynamic path, does not exist. Fortunately, a (stricter) safety rule can generate a stable solution as long as the natural decay mechanism is concave in the stock pollution level. In addition, utilizing the natural ability for self restoration we have the result that a stricter safety rule may generate a higher consumption and lower pollution level in steady state. The most important implication from this study is that it is crucial to have as accurate information as possible about the mechanisms that naturally decompose pollution whenever we are uncertain about the impacts following from economic activity. For important environmental problems such as the greenhouse gas problem we unfortunately cannot claim that this presumption is fulfilled.

The next section takes a closer look at the rational for using the safety rule in intertemporal pollution control problems. Section 3 outlines a baseline linear decay model to which the safety rule is applied. The allocation problem is analyzed with a nonlinear natural decay function in Section 4, and determinants of the size of the insurance premium to avoid unwanted outcomes are analyzed in Section 5. The last section concludes the paper.

2. POLLUTION CONTROL AND THE SAFETY RULE

In the earlier literature on the control of stock pollutants Plourde [19], Smith [23], Keeler et al. [14], and Forster [9], [10], all present deterministic models analyzing effluents emitted smoothly and continuously as by-products of consumption and production. Later Cropper [7] and Clarke and Reed [6] analyzed stock pollutants that could trigger catastrophic losses and both considered cases where the risk of catastrophic outcomes was known with certainty. The study by Clarke and Reed [6] is the one closest to the present analysis, but differs in that they emphasize irreversibility and focus on risk as a nondecreasing function of pollution concentration. However, we generally know that there is significant uncertainty about the effect of toxic materials for a number of reasons. First, only a fraction of the number of the chemicals in our surroundings is tested scientifically. Second, the health risks assessed by risk assessment are more often than not based on the effect of each individual chemical. What we usually see is that chemicals appear in a mixture and that our knowledge about the potential damage from these chemical combinations is insufficient. A specific example of the uncertainty related to risk assessment methods is the identification of weaknesses in extrapolating toxicological data from laboratory to the environment in relation to the question of disease induction in fish (Dethlefsen [8]).

In order to deal with the considerable uncertainty typically surrounding the risks posed by environmental contaminants Lichtenberg and Zilberman [15] proposed an approach that means the application of a safety rule decision criterion to a probabilistic model of risk generation[1]. The idea behind the safety rule is specifying risk as a function of the pollution level, which is constrained to remain below a given maximum allowable level within a given margin of safety. Essentially this can be interpreted as a variant of Kataoka's [13] safety fixed model and also as an extension of the standards and pricing approach of Baumol and Oates [2]. The logic is that as long as we are short of information about the true costs associated with some external effect, we cannot find and implement the optimal level of pollution. The safety rule approach is also appealing because it reflects the challenges in practical politics. The policy process addresses societal ills or challenges in which timeliness often is an essential factor. In instances where errors to be avoided are associated with extensive costs, for instance in the form

[1] For an application of the safety rule see Lichtenberg, Zilberman and Bogen [16].

of environmental catastrophes, action must therefore precede knowledge. Scientists can help illuminate the trade-offs within complex environmental problems with numerous plausible solutions leading to numerous possible futures but there is nothing objective about valuing environmental protection over economic growth. Ultimately, regulators must balance social cost related to some margin of safety against the reduction in health risk. More specifically, a safety rule approach forces regulators to consider the maximum level of health risk that we are willing to accept and how certain we would like to be that this upper limit is not exceeded. The same line of argument is found in Ciriacy-Wantrup [5] and Bishop [3] as the reason for applying a precautionary approach known as the safe minimum standard for conservation (SMS). However, this is believed to be a very conservative rule and a modified rule has been proposed stating that the SMS criterion should be adopted as long as the cost of doing so is not unacceptably large. Supporting this view, Randall and Farmer [21] find that there are no overwhelming arguments in favor of future generations' right to demand societal constructions of the present to be decimated in order to secure a minimum welfare level in the future. Therefore, in situations with uncertainty the SMS and the safety rule are seen reflected in various environmental protocols, where some qualifications have emerged. In the Rio summit the message to all countries is to follow the precautionary principle, but only "... according to their capabilities.", and in the 1990 White Paper protocol the UK Government stated that precaution should be applied "... if the balance of likely costs and benefits justifies it" (Heywood [11]). Further, we have seen that the inability to find the optimal level of pollution has led regulators to prescribe statutes such as the Federal Insecticide, Fungicide and Rodenticide Act and the Safe Drinking Water Act in the USA (see Lichtenberg and Zilberman [15]).

The safety rule is expressed as a condition specifying risk, R, as a positive, continuous and twice differentiable function of stock pollution, P, which is constrained to remain below a given maximum allowable level R_0 within a given margin of safety ρ[2]:

$$\Pr\{R(P) \leq R_0\} \geq \rho \Leftrightarrow \Pr\{R(P) \geq R_0\} \leq 1-\rho \tag{1}$$

Like Lichtenberg and Zilberman, the present approach works with the logarithm of risk, denoted $r(P)$, which is assumed to be a normally distributed random variable with mean $\mu(P)$ and standard deviation $\sigma(P)$. When $f(\rho)$ is the critical value of the standard normal distribution exceeded only with probability $1-\rho$, we have that

[2] Lichtenberg and Zilberman [15] suggest that the level of the safety margin normally would be the counterpart to what is generally used for scientific reliability, for instance a significance level of 5 %. This proposal is not discussed further.

$f(\rho) = (r(P) - \mu)/\sigma$. Specified this way, the safety rule constraint, as in Equation (1) reads:

$$r(P) = \mu(P) + f(\rho)\sigma(P) \leq r_0 \tag{2}$$

where r_o is the log risk standard. The risk associated with certain levels of stock pollution is therefore a combination of mean risk and uncertainty, where uncertainty is the standard deviation weighted by the constant $f(\rho)$. The weight on uncertainty can be interpreted as the decision maker's degree of risk aversion (Lichtenberg and Zilberman [15]), which together with the log risk standard defines an upper limit for the level of stock pollution. From Equation (2) it is clear that complying with the safety rule is consistent with a higher level of stock pollution the lower the uncertainty about health risk.

Applying the safety rule to an intertemporal instead of a static optimization problem such as in Lichtenberg and Zilberman [15] requires commenting. The dynamic aspect differs from a static one with regard to the potential of learning from previous experience. This aspect is therefore included in the model as the most troubling cases for pollution control are those where the lag between emissions and actual impacts on humans and eco-systems are large. It is in this situation that the learning potentials are small or even insignificant[3,4]. For instance, information available today suggest that emissions of greenhouse gases, and especially in the case of CO_2, will lead to great consequences for natural and human systems but where we at the same time do not know the exact linkage between emissions and the magnitude of total costs of the activities causing these emissions. This is the kind of pollution control problem that the safety rule is well suited for; we have some information about unwanted consequences, there are uncertainties related to the assessment of these relationships, the potential damage might be enormous and we may not learn from our experience early enough in order to change our course.

By including the safety rule in models of pollution control we obtain two objectives. First, we are able to analyze a pollution control problem where the cost of stock pollution can be substantial even below some critical point where possibly catastrophic impacts are expected to occur. This means we are in line with the criticism

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[3] As it stands, the model can be interpreted as a framework for efficient allocation in the periods previous to the point in time where information has become available.

[4] This means that the regulator will only consider mean health risk and not the standard deviation in the welfare formulation. In other words, we do not consider any quasi-option values other than through the level of safety margin. A possible modification of the model in situations where information becomes available in the near future would be to let the actual health risk to be included in the welfare formulation, and hence be able to identify the quasi-option value.

of other pollution control models made in the study of Clarke and Reed [6]. Second, the two main factors that have to be considered by regulators are the maximum level of health risk that we are willing to accept and how certain we would like to be that this upper limit is not exceeded. The contribution of our analysis is therefore highlighting the main issues behind the intertemporal balancing of cost against the protection of public health faced by legislation and to signal that uncertainty surrounding impacts from economic activity should be tackled by a precautionary approach[5].

The more technical aspects of irreversible nonlinear decay functions related to consumption pollution trade offs are found in Tahvonen and Salo [28], Tahvonen and Withagen [27], and an extension towards renewable resource harvesting in Tahvonen [26]. The present focus is on emphasizing the trade offs between consumption and pollution considering natural mechanisms and uncertainty especially, in an analysis using straightforward optimal control theory. Therefore we start our analysis by applying a safety rule to a simple pollution control model assuming linear natural decay.

3. APPLYING A SAFETY RULE TO POLLUTION CONTROL

We assume that society's welfare from consumption net of flow externalities[6] can be represented by an increasing, concave function of aggregate consumption, $U(C)$, where $U(0)=0$, and the first and second order derivative are given as $U_C>0$ and $U_{CC}<0$. In order to rule out zero consumption in steady state we assume that $\lim_{C\to 0} U_C = \infty$. Further, consumption increases the stock of pollution at an increasing rate according to the convex function $g(C)$, whereas expenditure on abatement, E, reduces accumulated pollution at a decreasing rate, described by the concave function $h(E)$[7]. A benevolent planner can use the total amount of resources available, Θ, on a mix of consumption and abatement. Since production is kept constant the model focuses solely on the trade off between consumption and pollution control. We can now define a pollution control function, $Z(C)$, as[8]: $Z(C)=g(C)-h(\Theta-C)$ which is further described by $Z_C=g_C+h_E>0$ and $Z_{CC}=g_{CC}-h_{EE}>0$. Thus, the stock of pollution,

[5] Note that no effects are included that might occur from changes in the probability distribution of risk or in the regulator's aversion towards risk.

[6] We assume that any external effects related to the flow of pollution are internalized in the net utility of consumption since externalities from this type of pollution are often characterized by local effects, which presumably are more likely to be internalized than (stock) pollution with global effects.

[7] This specification of *h(E)* permits that the reduction of pollution can happen at the source rather than at the receptor.

[8] This general specification of $Z(C)$ opens the way for situations with net pollution abatement if *C* is small enough.

$P(t)$, changes according to the material balance condition $\dot{P} = Z(C) - F(P)$, where $F(P)$ is the natural decay. We assume that we have information about the initial stock of pollution: $P(0) = P_0$. Further, stock pollution creates a negative (expected) externality in the form of health risk according to $r(P) = \mu(P)$ and $\mu_P, \mu_{PP} > 0$.

The benchmark situation is where the planner's problem is formulated as maximizing expected net utility from consumption:

$$\max_C \int_0^\infty e^{-\delta t} \left(U(C) - \mu(P)\right) dt$$

subject to the mass balance condition for the stock of pollution, and where $\delta > 0$ is society's discount rate. The monetary value of expected health risk is normalized to one. To solve the problem we formulate the current value Hamiltonian:

$$H = U(C) - \mu(P) + \psi\left(Z(C) - F(P)\right) \tag{3}$$

ψ is the co-state variable associated with the pollution stock, and is interpreted as the reduction in future utility caused by an increase in the pollution stock. Assuming an interior solution, the following necessary conditions must be satisfied along an optimal trajectory[9]:

$$U_C + \psi Z_C = 0 \tag{4}$$

$$\dot{\psi} = \psi\left(\delta + F_P\right) + \mu_P \tag{5}$$

$$\dot{P} = Z(C) - F(P) \tag{6}$$

$$\lim_{t \to \infty} e^{-\delta t} \psi(t) P(t) = 0 \tag{7}$$

The maximum-principle condition (4) states that the marginal gain in utility must equal the costs entailed by the decrease in future utility caused by an increasing stock of pollution. Combining this condition with the portfolio balance condition (5) gives the *C*-isocline as[10]:

$$U_C = \frac{Z_C \mu_P}{\delta + F_P} \tag{8}$$

which states that the combinations of consumption and pollution levels consistent with stationary consumption levels are characterized by situations where marginal utility from consumption controlled for marginal net pollution abatement equals the marginal health risk controlled for the discount factor and the marginal natural decay. Together with the *P*-isocline this produces the well-known result that the equilibrium in this unconstrained situation is a saddle-point path. The optimal strategy prescribes a high

[9] These necessary conditions are sufficient for optimality and the transversality condition (7) is fulfilled when steady state is reached.
[10] For details, see Appendix A.

(low) level of consumption for low (high) levels of stock pollution, which decreases (increases) over time until steady state is reached.

Due to the uncertainty which is characteristic for the problems of interest, a safety rule as outlined in Equation (2) is applied to the pollution control model presented above as an upper limit for the level of health risk, which consequently means an upper level of stock pollution. Since expected health risk is already included in the function to be maximized the implementation of a safety rule is based on a regulator concerned about uncertainty associated with stock pollution. Given the restriction on health risk the planner's problem now looks like[11]:

$$\max_{C} \int_0^\infty e^{-\delta t}\left(U(C)-\mu(P)\right)dt$$

subject to $\dot{P}=Z(C)-F(P)$, and

$\dot{P}=Z(C)-F(P)\leq 0$, whenever $r(P)=r_0$

The latter constraint is a state-space constraint and says that the level of stock pollution is not allowed to increase when health risk has reached the log risk standard[12]. r_0 is determined outside the model but later we will investigate the impacts of different levels of safety margins.

We now formulate the Lagrangian $L=H-\Phi\dot{P}$, where H is given from Equation (3), and Φ is the shadow price of a marginal increase in the margin of safety, ρ. The following necessary conditions must be satisfied along an optimal trajectory (Chiang [4]):

$$\partial L/\partial C = U_C+\psi Z_C-\Phi Z_C\leq 0 \tag{9}$$

$$\partial L/\partial \Phi = -\dot{P}=F(P)-Z(C)\geq 0 \quad \Phi\geq 0 \quad \Phi\frac{\partial L}{\partial \Phi}=0 \tag{10}$$

$$r(P)\leq r_0 \quad \Phi\left(r_0-r(P)\right)=0$$

$$\dot{\Phi}\leq 0 \quad \left[=0 \text{ when } r(P)<r_0\right]$$

$$\dot{P}=\partial L/\partial\psi = Z(C)-F(P) \tag{11}$$

$$\dot{\psi}=\delta\psi-\partial L/\partial P=\psi\left(\delta+F_P\right)+\mu_P+\Phi F_P \tag{12}$$

and condition (7).

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[11] The welfare formulation includes only expected health risk since the concern for unwanted impacts is accounted for by the introduction of the safety rule.

[12] The reason for expressing the safety rule as a constraint on $\dot{P}$ is that a constrain like $r(P)>r_0$ does not account for any discontinuities in the costate variable ψ (for a discussion of jump conditions, see chapter 5 in [25]).

Proposition 1 *Given that the marginal natural decay mechanism is positive, and that the log risk standard is a binding constraint on health risk, the optimal levels of consumption and stock pollution along the saddle-point path and in steady state must be reduced relative to a situation where the regulator considers only expected health risk.*

Proof. By combining Equations (9) and (12) we obtain the C-isocline as[13]:

$$U_C = \frac{Z_C\left(\mu_P + \Phi\left(\delta + 2F_P\right) - \dot{\Phi}\right)}{\delta + F_P} \tag{13}$$

From the necessary conditions found above we know that $\Phi F_P + \Phi\left(\delta + F_P\right) - \dot{\Phi} > 0$, and that the larger the shadow price of the safety rule the lower the level of consumption along the saddle-point path and in steady state. ■

Equation (13) states that the combinations of consumption and pollution levels consistent with stationary consumption levels are characterized by situations where utility from a marginal increase in consumption levels are equal to costs consisting of marginal increase in health risk and the potential cost associated with the safety rule, controlled for the discount factor and the marginal natural decay. This is basically a reflection of consumption being traded off for a given level of safety margin. We further know that under the safety rule regime the optimal level of steady-state pollution is defined by a level consistent with $r(P) = r_0$ since there is no net gain from reducing pollution in steady state such that $r(P) < r_0$.

The motions outside stationarity are not affected by the introduction of the safety rule, which means that the new equilibrium is also a saddle-point path[14]. From Equation (13) we also see that a lower discount rate, a lower level of marginal decay, a higher level of pollution added for marginal consumption and a higher level of marginal health risk all have the same effect on the C-isocline as accounting for uncertainty, namely shifting the isocline down in P,C -space[15]. A further investigation of the trade off between consumption and the safety margin shows that the C-isocline is negatively sloped in the P,C space and less steep the larger the expected marginal health risk is, the

[13] For details, see Appendix B.

[14] Note that the P-isocline is not affected by accounting for uncertainty.

[15] This is consistent with the result of Clarke and Reed [6] when risk is policy-independent, i.e. increases with the pollution stock.

tighter the margin of safety (evaluated by the shadow price) and the more weight is put on the uncertainty[16]. In other words, the more a reduction in stock pollution reduces expected health risk and the cost induced by uncertainty, the more the level of consumption can increase along the *C*-isocline. We have two effects stemming from accounting for uncertainty, namely that it does not only shift the *C*-isocline down in *P,C* space, it also alters its slope. The shift is a result of an "income" effect due to the increase in costs associated with consumption. Thus, for a given amount of resources the new equilibrium must reflect the fact that an increase in the level of pollution control means that consumption must be reduced for all levels of pollution, which is illustrated in Figure 1 as the move from point a to point b. Second, we have a "price" effect. Since consumption now is relatively more expensive compared to pollution control the new equilibrium is not at point c, but at point d reflecting the relative change in efficient combinations of consumption and pollution control.

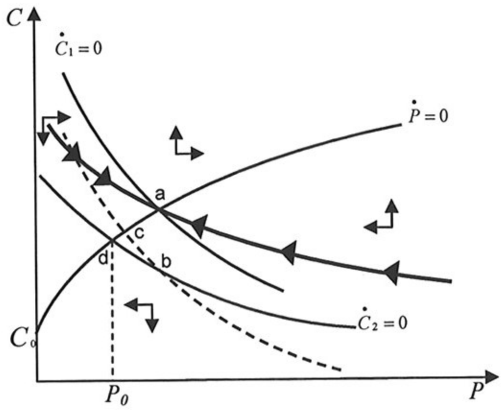

Figure 1. Comparing a situation where a safety rule is applied (indicated by point d) with a situation without a restriction on the level of stock pollution (indicated by a). P_0 indicates the maximum level of pollution equivalent to the constraint following the safety rule.

Note that the magnitude of the change in consumption and stock pollution from points a to d in Figure 1 depends on to what degree a reduced pollution level affects either the expected health risk or the costs associated with uncertainty, or both. Consequently, the more uncertain we are about the relationship between health risk and stock pollution the larger is the reduction in consumption and stock pollution that is required for compliance to the safety rule.

[16] Details are found in Appendix B.

4. SITUATIONS WITH NONLINEAR DECAY

Because of the idiosyncrasy and complexity characterizing the impacts of human activities on eco-systems, it is of course difficult to comprise the many possible relationships between emissions, health risk and the level of pollution over time within a single analysis. However, by introducing a natural decay function characterized as logistic into our model it is possible to highlight a wider range of relations between the economic and ecological aspects within the defined problem[17].

This approach works with a logistic absorption mechanism which is often more realistic than a linear one because it reflects the fact that the marginal removal can be zero and negative as well as positive. Further, it is worth noting that in the case of logistic decay functions there is a 'carrying capacity'. Above this level the natural abatement is zero, and in the case of irreversibility, there is no way of restoring the natural abatement service (Tahvonen and Withagen [27])[18]. Another distinction between linear and nonlinear decay functions is that while the number of steady states with a linear absorption function is limited to one, situations with nonlinear absorption mechanisms can produce numerous equilibriums, which can be both stable and unstable (Tahvonen and Salo [28]).

A natural renewal characterized by a logistic function can be formulated as $F(P) = \eta P(1 - P/K)$, where η is the maximum marginal decay and where the natural decay is zero for levels of stock pollution above *K*. Including this in the model changes the *P*-isocline, which now has a positive slope for $P \in \langle 0, K/2 \rangle$ and negative for $P \in \langle K/2, K \rangle$. In other words, the marginal decay is positive for low pollution stocks, but becomes negative for sufficiently high levels of pollution. We can still have positive consumption levels above $P = K$ that are consistent with stationary levels of stock pollution as long as man-made resources are allocated to abatement activities.

The *C*-isocline is still given by Equation (13) even with the formulation of a nonlinear natural decay mechanism. However, we find that the positioning and slope of the isocline changes since the marginal decay affects the marginal costs of consumption. A second implication is that we now can have multiple steady states, since the *P*-isocline in *C,P* space is concave up to the point where *P=K*. For pollution levels above this point the isocline is horizontal since stationary pollution levels are found where

[17] A further discussion on eco-system services, irreversibility, and further references on the subject, is found in Scheffer et al. [22].

[18] If there also exists uncertainty about the effects of eliminating the natural renewal mechanism, there consequently is an option value present. This aspect of the problem is not analyzed in this analysis.

$Z(C)=0$. In the following we will focus mainly on the equilibriums found for pollution levels where the natural renewal mechanism is still functioning.

It is also important to point out that the *C*-isocline is no longer defined for specific levels of stock pollution. As long as the safety rule is not an effective constraint on the optimal path and in steady state, $\delta + F_P > 0$ must hold in order for a *C*-isocline to exist. This condition constrains the *C*-isocline and hence the maximum steady-state level of stock pollution, defined by the point where $\delta + F_P = 0$ (see Equations (8) and (13)). This becomes clear when we recall that whenever natural decay is linearly related to the level of pollution the planner can increase the level of future decay with a constant rate of return, namely F_P, by increasing the level of pollution. On the other hand, when marginal decay diminishes with the level of pollution it is obvious that the gain from having large levels of pollution is also reduced. For a discount rate equal to zero the upper level for stock pollution defined by the *C*-isocline coincides with *P=K/2*, which reflects that it is not possible to maintain a constant consumption-pollution level above this pollution level due to the negative marginal return from investing in the natural decay. For discount rates larger than zero the negative effect of decreased marginal decay is discounted, and it is possible to increase the steady-state pollution level above *K/2*.

In order for a steady state to exist, and an efficient path leading to it, there has to be present a level of consumption for a given pollution level which can be shown to be constant over time. In other words, a *C*-isocline must be identified. Given our logistic decay function two conditions have to be met for the *C*-isocline to exist, and we also find that the *C*-isocline can be both downward and upward sloping in *C,P* space. Consider the following two scenarios:

Scenario I: $F_P > -\delta$ *and* $F_P > \dfrac{\dot{\Phi} - \mu_P - \delta\Phi}{2\Phi}$

These conditions are necessary for the existence of a *C*-isocline and hold by definition as long as $P \leq K/2$. Given that $P > K/2$ and that the above conditions still hold, together with the reasonable assumption that $\mu_{PP} / F_{PP} > -2\Phi$, we also know that the slope of the *C*-isocline is negative in *C,P* space[19]. Consider next:

Scenario II: $F_P < -\delta$ *and* $F_P < \dfrac{\dot{\Phi} - \mu_P - \delta\Phi}{2\Phi}$.

[19] As long as decay is linearly related to the level of stock pollution the conditions in scenario I hold by definition.

Obviously these conditions are necessary for existence of the C-isocline only for situations where $P > K/2$[20]. Scenarios I and II are illustrated in Figure 2 below. Evidently, there are levels of pollution for which the C-isocline is not defined. Which of the two constraints that is binding with respect to the existence of the C-isocline is determined by the following parameter constellations: If $\delta < \left(\mu_P - \dot{\Phi}\right)/\Phi$ then it is the condition $F_P > -\delta$ that constrains the maximum steady-state pollution level. In the opposite case when $\delta > \left(\mu_P - \dot{\Phi}\right)/\Phi$ it is $F_P > \dfrac{\dot{\Phi} - \mu_P - \delta\Phi}{2\Phi}$ that constrains the steady state pollution level. The implication from this is that the higher the discount rate and the larger the cost associated with uncertainty related to effects from pollution, the more likely is it that we do not have a steady-state equilibrium. The explanation behind this result is that high discount rates work in the direction of high steady-state levels of pollution, whereas large costs due to uncertainty work in the opposite direction[21].

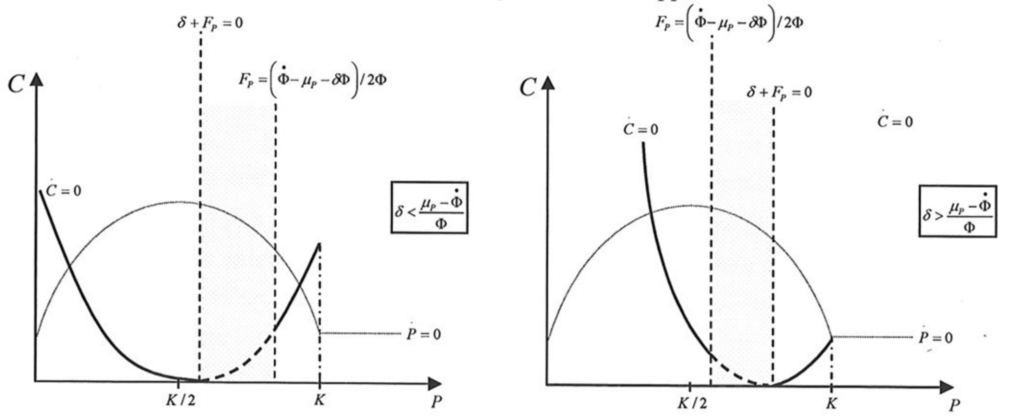

Figure 2 Possible positions and slope of the C-isocline under different assumptions about the marginal decay, the discount rate, expected risk and the shadow price of the safety margin. Figures (a) and (b), illustrate scenario I and II, respectively. The shaded areas denote levels of *P* where the *C*-isocline is not defined. Note that for *P>K* the slope of the *C*-isocline is always negative, for details see Appendix C.

In situations without a binding safety rule there is always a stable steady state. It has been demonstrated here that this is not necessarily the case when the safety rule is an effective constraint. Therefore we will look more into these situations:

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[20] Still assuming that $\mu_{PP}/F_{PP} > -2\Phi$, the slope of the C-isocline is characterized by the conditions in scenario II can be shown to be positive. If $\mu_{PP}/F_{PP} > -2\Phi$ does not hold the slope of the isocline is undetermined. For details see Appendix C.

[21] In addition we have that $\dfrac{d}{d\Phi}\left(\dfrac{\dot{\Phi} - \mu_P - \delta\Phi}{2\Phi}\right) = \dfrac{-\dot{\Phi} + \mu_P}{2} > 0$.

Proposition 2 *In contrast to substances with linear decay, situations where a stable steady state does not exist can occur when the natural decomposition is nonlinear. However, under given circumstances a higher margin of safety can generate the existence of a stable steady state.*

Proof. Assuming that $\mu_{PP} / F_{PP} > -2\Phi$, it has been shown that when $\delta > \left(\mu_P - \dot{\Phi}\right) / \Phi$ we must have that $F_P > \dfrac{\dot{\Phi} - \mu_P - \delta\Phi}{2\Phi}$ in order for a negatively sloped C-isocline to exist. The existence of a stable steady state requires that the C-isocline is located sufficiently low in the C,P diagram compared to the P-isocline. An increase in the margin of safety shifts the C-isocline down in C,P space for levels of P relevant for a downward sloping C-isocline $\left(dU_C / d\Phi = \left(Z_C\left(\delta + 2F_P\right)\right) / \left(\delta + F_P\right) > 0 \text{ when } 2F_P > -\delta\right)$, and are thus able to produce a stable steady state. ■

A situation as characterized above is illustrated in Figure 3 and is most likely to be found when the discount rate is high compared to the level of marginal decay. In other words, if we are overly impatient we are not able to reach a stable steady state level of consumption and stock pollution since the rate of return from the natural resource, which here is the way nature can decompose an additional unit of pollution, is restricted. By imposing a (stricter) safety rule more emphasis is put on the long run cost of consumption and hence, might make a steady state solution possible.

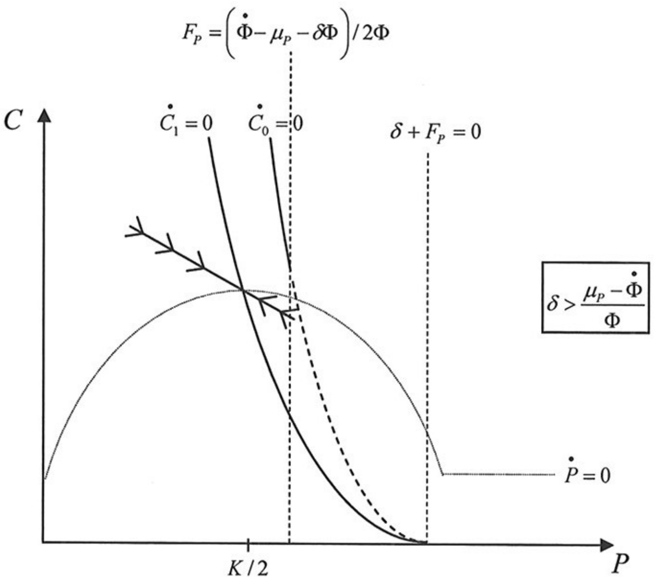

Figure 3 Illustration of situation where the safety rule is an effective constraint and a stable steady state does not exist, but where a higher margin of safety makes the existence of a stable steady state possible. The *C*-isocline denoted 1 illustrates a higher margin of safety than the isocline denoted 0.

From the discussions above it has become clear that the relationship between the safety margin and consumption levels in steady state is highly influenced by the mechanism by which nature decomposes substances. As long as nature decomposes harmful substances linearly the optimal level of consumption and pollution is lower the higher the margin of safety. This does no longer hold for nonlinear decay:

Proposition 3 *Given that the log risk standard is a binding constraint on health risk and that the level of stock pollution in the initial steady state is sufficiently larger than the point where marginal decay starts to diminish with the stock pollution, increases in the margin of safety lead to higher levels of consumption in stable steady state combined with continuously lower levels of stock pollution.*

Proof. As long as $\mu_{PP} > -2\Phi F_{PP}$ and the level of stock pollution is consistent with $F_P > -\delta$ and $F_P > \frac{\dot{\Phi} - \mu_P - \delta\Phi}{2\Phi}$ we have shown that the *C*-isocline is downwardsloping. Given that $2F_P > -\delta$ an increased margin of safety is equivalent to a negative shift in the C-isocline (see Proposition 2). Proposition 3 is thus ensured by the concave P-isocline: $\frac{d^2C}{dP^2}|_{\dot{P}=0} = F_{PP} / Z_C < 0$. ■

Proposition 3 is illustrated graphically in Figure 4. This particular result is an effect of an increase in the natural decay on the margin as the level of pollution is lowered. Because of this the consumption level is allowed to increase. In other words, the trade off between margins of safety and consumption levels is less costly in terms of reduced consumption levels as the level of natural decomposition is increased. Note that this situation is found only when the discount rate is at a level much higher than the marginal decay. It is therefore relevant as a special case characterized by a type of pollution associated with a very low rate of marginal decay or for situations with an extreme rate of discount.

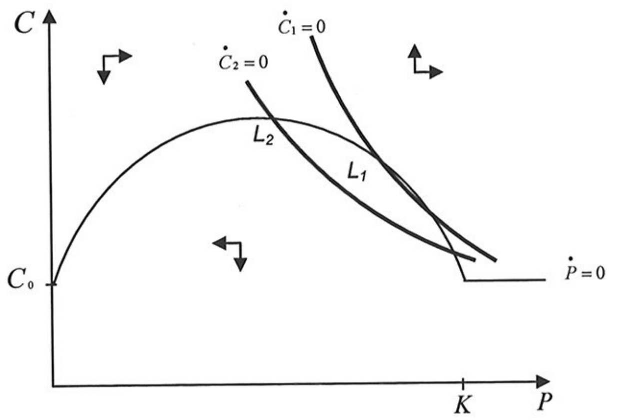

Figure 4 Illustration of a situation where increased margin of safety leads to an increase in consumption and a decrease in stock pollution in steady state. Stable steady states are denoted L_1 and L_2.

A conclusion from this section is that we often can expect a trade off between consumption levels and margins of safety, but this relationship is not as straightforward when natural decay is nonlinear as when it is linear. Possible relationships between consumption in stable steady state and the safety margin are summarized in Figure 5. Regarding the relationship between the margin of safety and steady state levels of pollution we find that given logistic decomposition of substances the pollution levels in stable equilibria are always negatively correlated with the level of the safety margin.

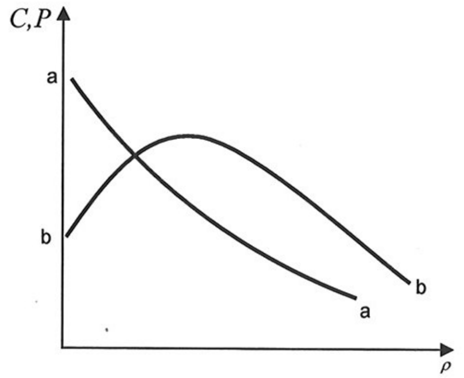

Figure 5. A sketch of the possible relationships between steady state consumption levels for different margins of safety. The lines aa and bb denote stable equilibria when decay is linear, and nonlinear, respectively.

5. THE INSURANCE PREMIUM

Reducing the consumption level according to a safety rule can be interpreted as paying an insurance premium to avoid unacceptable levels of health risk. The justification of this kind of precautionary approach must at some point be done with reference to the cost of confinement to the decision rule. Analogous to precautionary approaches such as the safe minimum standard, the cost of confinement to the decision rule should be undertaken unless the cost of doing so is not intolerably large (Ciriacy-Wantrup [5]; Bishop [3]). For these reasons it is interesting to analyze how the size of the premium is affected by factors like the discount rate, improvement in technology and nature's ability for self restoration.

We note from Equation (6) that the P-isocline is determined from the natural decay and the pollution control function, which is a purely technical relationship. The position and slope of the C-isocline are on the other hand affected by the degree to which uncertainty is accounted for, which allows the effect of neglecting the safety rule to be found from investigation of the C-isocline. Given a level of stock pollution the difference between confinement to the safety rule and neglecting it is reflected by the magnitude of change in consumption level needed to implement the optimal path and the corresponding steady state as developed above. This difference in consumption levels, which we understand as an insurance premium, $I(P)$, can be approximated as[22]:

[22] An alternative interpretation of the insurance premium is a (current value) tax imposed by a regulator wanting to implement an efficient solution. Further, it is reasonable to assume that the insurance premium generally is positive, for details see Appendix D.

$$I(P) = U_C^* - U_C^\# \Rightarrow I(P) = \frac{Z_C\left(\Phi(\delta + 2F_P) - \dot{\Phi}\right)}{\delta + F_P} > 0$$

where the asterisk and the # denote the optimal consumption path with and without a binding safety rule, respectively.

It is easy to show that as long as marginal decay is positive a higher discount rate means a reduction in the insurance premium. This effect is due to consumption being more sensitive to changes in the discount rate when a safety rule is adopted compared to when it is not. The reason for this is that we have an additional effect of the discount rate whenever the safety rule binds; confining to a safety rule means that consumption today brings with it a cost in addition to that related merely to increased expected pollution. Consider a reduction in the discount rate, which means that more emphasis is put on a greater future cost under a safety rule regime. This infers a greater reduction in consumption levels relative to a situation where uncertainty is not accounted for. The same holds for logistic decay functions as long as the marginal decay is larger than $\dot{\Phi}/\Phi Z_C$[23]. Marginal decay below this level counterbalances the effect of the discount rate and the insurance premium increases for a higher discount rate.

Changes in the structure of ecosystems may change nature's ability for self restoration, and the relationship between an exogenous shift in marginal decay and risk exposure is simple:

$$\frac{\partial I}{\partial F_P} = \frac{Z_C\left(\Phi\delta + \dot{\Phi}\right)}{(\delta + F_P)^2}\begin{cases} > 0 \text{ if } \dfrac{-\dot{\Phi}}{\Phi} < \delta \\ < 0 \text{ if } \dfrac{-\dot{\Phi}}{\Phi} > \delta \end{cases}$$

If the relative change in costs associated with the safety rule more than offsets the discount rate, an increase in marginal decay reduces the exposure to uncertainty[24]. Symmetrically, for high discount rates a higher level of natural decay increases the level of risk exposure. Furthermore, over time we would expect technological improvements so that the level of net pollution from one unit consumption is lowered. The impact on the insurance premium is, as we would expect, one where increased technological efficiency reduces the cost of complying with a binding safety rule[25].

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23 This result is possible also when marginal decay is less than $\dot{\Phi}/\Phi Z_C$ but somehow less likely to occur since it is possible that $\partial I / \partial \delta > 0$.

24 See Appendix D for details.

25 Details on the calculations are found in Appendix D.

6. CONCLUDING REMARKS

Even if we have information about probability distributions, that is, we are in a situation dealing with risk, it is an utopian idea to think that we are able to implement the optimal solution in a complex world such as ours. What is more, the possible outcomes may be so adverse that policy makers in any case would want to safeguard against these outcomes. An analogy is the car owner buying insurance even if he or she knows that the insurance company, based on probability distributions, knows that he or she is not likely to cause damage greater than the insurance premium. Despite expected outcomes policy makers, like most people, would like to undertake measures in order to protect their life or investment in physical capital, that is, they buy insurance. This paper has analyzed insurance in the form of a safety rule approach to pollution control.

This analysis has demonstrated how a safety rule affects intertemporal allocations of resources between producing consumption goods and pollution control. Basically it has found that in steady state the cost of erring on the side of caution in terms of reduced consumption levels depends on how nature decomposes harmful substances and hence that the cost can both increase as well as decrease with the margin of safety. Along the optimal path towards steady state we generally have margins of safety that have to be traded off against lower levels of consumption. Further, shortsightedness, represented by a high discount rate, together with a relatively low marginal natural ability for self restoration may lead to situations where steady states do not exist. However, we have shown that a higher margin of safety can generate a stable solution by counterbalancing the discrepancy between impatience and nature's ability to restore itself.

Once the regulator has chosen his or her margin of safety the model does not call for any drastic measures. A considerable change in consumption levels is a product of changes in the degree to which we consider uncertainty. It generally would be efficient to change our course towards new steady state consumption and pollution levels if our preferences for uncertainty change. At the same time, there are limits to the amount of resources that one would be willing to give up to avoid potentially large costs in the future. One interesting result that we have found is that the size of an insurance premium paid to avoid exposure to uncertainty is smaller the higher the discount rate, which is explained by less emphasis on future costs as the rate of discount increases. In other words, caring about uncertainty can be costly, but compared to the alternative solution it may nevertheless be an advisable strategy.

Our analysis has demonstrated how the resilience of nature significantly affects efficient management strategies for pollution control. A hot topic related to this is the problem with greenhouse gases. First, different gases have different decay mechanisms and second, they interact with other particles in the atmosphere making it difficult to have full knowledge about the interrelations among them and the human made emissions. Another example is the continuous sedimentation of toxic materials on the sea floor that can reduce the stock pollution but at the same time we are aware that there are large differences to what degree this will happen at different locations. For various marine locations in Norway it is estimated that the time it will take to sufficiently prevent marine organisms to be exposed to toxic materials in the sediments varies from 50 up to 100 years given that there is no further adding of toxic materials (SFT [24]). There are also stochastic variations in the rate of decay due to bioturbation, whirling caused by currents and waves, the content of oxygen and organic material in the sediment, and the interaction between different environmental toxics.

What we see is that uncertainty pertains not only to health risk but also to natural decay mechanisms. A main policy implication from our analysis is therefore that whenever the relationship between stocks of pollution and the natural decomposition is not known with certainty, precaution should be taken. If not, we have demonstrated that when health risk effects are uncertain we may not be able to find an efficient path for pollution control, and hence, not able to perform the important trade off between consumption and precaution against unforeseen and unwanted events that potentially could reduce society's level of consumption dramatically. Consequently, under uncertainty it is more crucial than ever to have as correct information as possible about how nature handles emissions created by man.

Further expansion of the present analysis with uncertainty regarding the natural decay mechanism is consequently an important topic and interesting candidate for further research.

APPENDIX

Appendix A The baseline model without a safety rule (linear decay)

The equation of motion in C is found by combining the differential of the maximum principle (MP) condition with respect to time with the portfolio (PB) condition. The co-state variable is eliminated and we get the following expression:

$$\dot{C} = \frac{Z_C\left(U_C\left(\delta + F_P\right) - Z_C\mu_P\right)}{Z_C U_{CC} - U_C Z_{CC}} \tag{A.1}$$

From (A.2) we see that the C-isocline is convex in the C-P diagram, approaching the axis asymptotically due to the presumed functional forms. Motion outside stationarity is seen from Equation (A.3).

$$\left.\frac{dC}{dP}\right|_{\dot{C}=0} = \frac{Z_C \mu_{PP}}{U_{CC}\left(\delta + F_P\right) - Z_{CC}\mu_P} < 0, \quad \left.\frac{d^2C}{dP^2}\right|_{\dot{C}=0} = \frac{Z_C \mu_{PP}\left(Z_{CC}\mu_{PP}\right)}{\left(U_{CC}\left(\delta + F_P\right) - Z_{CC}\mu_P\right)^2} > 0 \quad \text{(A.2)}$$

$$\frac{\partial \dot{C}}{\partial P} = \frac{-\mu_{PP} Z_C \left(Z_C U_{CC} - Z_{CC} U_C\right)}{\left(Z_C U_{CC} - Z_{CC} U_C\right)^2} > 0 \quad \text{(A.3)}$$

The P-isocline is found where the net addition of pollution in the receiving media equals the natural removal: $Z(C) = F(P)$. The slope of the P-isocline is seen from

$$\frac{dC}{dP} = \frac{F_P}{Z_C} > 0$$

and motion outside stationarity:

$$\frac{d\dot{P}}{dC}\Big|_{dP=0} = Z_C > 0$$

Increased pollution levels and hence, increased natural removal, allows a higher consumption producing a positively sloped P-isocline (Equation (A.4)). The curve is concave because the net contribution to the pollution stock from marginal consumption is increasing.

$$\left.\frac{dP}{dC}\right|_{\dot{P}=0} = \frac{Z_C}{F_P} > 0, \quad \left.\frac{d^2P}{dC^2}\right|_{\dot{P}=0} = \frac{Z_{CC}}{F_P} > 0 \quad \text{(A.4)}$$

Above the isocline the stock pollution increases, and vice versa.

Appendix B Including a safety rule (linear decay)

Including the safety rule in the pollution control model

$$\frac{\partial L}{\partial C} = 0 \Rightarrow U_C + \psi Z_C - \Phi Z_C = 0$$

$$\dot{\psi} = -\frac{\partial L}{\partial P} + \delta\psi \Rightarrow \dot{\psi} = \psi\left(\delta + F_P\right) + \mu_P + \Phi F_P$$

The equation of motion for consumption levels is given as:

$$\dot{C} = \frac{Z_C\left(\dot{\Phi} - \mu_P - \Phi\left(\delta + 2F_P\right) + U_C\left(\delta + F_P\right)\right)}{U_{CC} + \frac{1}{Z_C}\left(\Phi Z_C - U_C\right) - \Phi Z_{CC}}$$

The C-isocline is found where $\dot{C} = 0$, which is satisfied when

$$U_C = \frac{Z_C\left(\mu_P + \Phi(\delta + 2F_P) - \dot{\Phi}\right)}{\delta + F_P}$$

And the slope is given as:

$$\left.\frac{dC}{dP}\right|_{\dot{C}=0} = \frac{Z_C \mu_{PP}}{(\delta + F_P)U_{CC} - Z_{CC}\left(\mu_P + \Phi(\delta + 2F_P) - \dot{\Phi}\right)}$$

Appendix C Including nonlinear decay

The slope of the C-isocline is given as:

$$\left.\frac{dC}{dP}\right|_{\dot{C}=0} = \frac{Z_C\left[(\delta + F_P)(\mu_{PP} + 2\Phi F_{PP}) - F_{PP}\left(\mu_P + \Phi(\delta + 2F_P) - \dot{\Phi}\right)\right]}{(\delta + F_P)\left[(\delta + F_P)U_{CC} - Z_{CC}\left(\mu_P + \Phi(\delta + 2F_P) - \dot{\Phi}\right)\right]}$$

which is negative if $\mu_{PP} > -2\Phi F_{PP}$.

The C-isocline for levels of pollution above the carrying capacity level, K:

For pollution levels above K we have that $F_P = 0$. It follows that the C-isocline is then given as: $\hat{U}_C = \dfrac{Z_C\left(\mu_P + \delta\Phi - \dot{\Phi}\right)}{\delta}$

Regarding the slope of the C-isocline it is now given as:

$$\left.\frac{dC}{dP}\right|_{\substack{\dot{C}=0 \\ F_P = F_{PP} = 0}} = \frac{Z_C\left(\delta\mu_{PP} - \dot{\Phi}\right)}{\delta\left[\delta U_{CC} - Z_{CC}\left(\mu_P + \delta\Phi - \dot{\Phi}\right)\right]} < 0.$$

Further we have that

$$U_C = \frac{Z_C\left(\mu_P + \Phi F_P + \Phi(\delta + F_P) - \dot{\Phi}\right)}{\delta + F_P} > \hat{U}_C = \frac{Z_C\left(\mu_P + \delta\Phi - \dot{\Phi}\right)}{\delta} \Rightarrow \delta > \frac{\mu_P - \dot{\Phi}}{\Phi}$$

as long as $\delta + F_P > 0$. If $\delta + F_P < 0$ we have that $U_C > \hat{U}_C$ only if $\delta < \dfrac{\mu_P - \dot{\Phi}}{\Phi}$.

Appendix D The insurance premium

As long as $F(P)$ is linear $I(P)$ is positive. When decay is logistic $I(P)$ is positive as long as $\delta + F_P > 0$ and $\dot{\Phi}/\Phi < \delta + 2F_P$. Other constellations are ruled out by the conditions for the existence of a *C*-isocline (see Figure 2). Hence, $I(P)$ is always positive.

Comparative statics:

The impact of an increase in the discount rate on the insurance premium is seen from:

$$\frac{\partial I}{\partial \delta} = \frac{-Z_C\left(\Phi F_P - \dot{\Phi}\right)}{\left(\delta + F_P\right)^2}$$

Moreover, $\partial I / \partial \delta < 0$ if the decay function is linear. In the logistic case we have the following relationship:

$$\frac{\partial I}{\partial \delta}\begin{cases} < 0 \text{ if } F_P > \dot{\Phi}/\Phi \\ > 0 \text{ if } F_P < \dot{\Phi}/\Phi \end{cases}$$

That the *C*-isocline responds more to a change in the discount rate is seen from:

$$\left.\frac{\partial U_C}{\partial \delta}\right|_{\dot{C}=0}^{\#} = \frac{-Z_C \mu_P}{\left(\delta + F_P\right)^2} > \left.\frac{\partial U_C}{\partial \delta}\right|_{\dot{C}=0}^{*} = \frac{-Z_C\left(\mu_P + \Phi F_P - \dot{\Phi}\right)}{\left(\delta + F_P\right)^2}$$

where the asterisk and the # denote the optimal consumption path with and without a binding safety rule, respectively.

The impact of technological improvement on the size of the insurance premium is given as:

$$\frac{\partial I}{\partial Z_C} = \frac{\Phi\left(\delta + 2F_P\right) - \dot{\Phi}}{\left(\delta + F_P\right)}$$

$$\Rightarrow \frac{\partial I}{\partial Z_C}\begin{cases} > 0 \begin{cases} \text{if } \delta + F_P > 0 \cup \dot{\Phi}/\Phi < \delta + 2F_P \\ \text{if } \delta + F_P < 0 \cup \dot{\Phi}/\Phi > \delta + 2F_P \end{cases} \\ < 0 \begin{cases} \text{if } \delta + F_P > 0 \cup \dot{\Phi}/\Phi > \delta + 2F_P \\ \text{if } \delta + F_P < 0 \cup \dot{\Phi}/\Phi < \delta + 2F_P \end{cases} \end{cases}$$

However, together with the constraints following from considering only a positive insurance premium we have that $\partial I / \partial Z_C > 0$ must always hold. Together with the conditions for the existence of a C-isocline (see Figure 2) the impact of technological improvement on the insurance premium can be illustrated in Figure D.1.

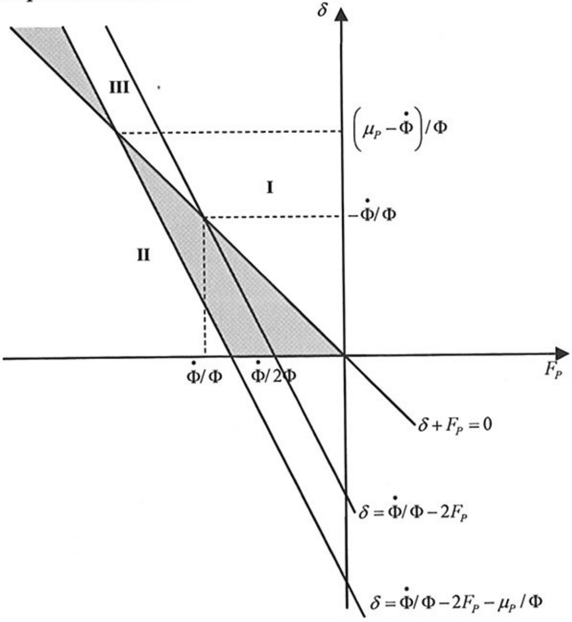

Figure D.1 The impact on the insurance premium, *I(P)*, from an increase in technological efficiency, Z_C. Regions I and II illustrate combinations of δ and F_P that lead to an increase in *I(P)* from an increase in Z_C. Region III illustrates combinations of δ and F_P that leads to a decrease in *I(P)* from an increase in Z_C, but where the insurance premium is negative. The shaded areas indicate combinations of δ and F_P which are not consistent with a steady state solution, confer with Figure 2.

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Paper 5

Economic growth and land-use changes: the declining amount of wilderness land in Norway

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ANALYSIS

Economic growth and land-use changes: the declining amount of wilderness land in Norway

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Abstract

The paper presents the evidence and analyses of macroeconomic factors explaining the reduction of wilderness land in Norway. The analysis is at the county level (18 counties) for the years 1988 and 1994, and the regressions are carried out as cross-section models as well as pooled, fixed effects models. Using a new, and probably unique, database that categorizes the total area in Norway into four zones of distance from larger technical installations, wilderness land has been defined in three ways, reflecting different qualities of the same type of natural resource; land as more than 5, 3 and 1 km from closest man-made encroachment, respectively. The explanatory variables comprise GDP per capita, GDP per capita squared, and population density. The main finding from the cross-sections analysis is that the relative amount of wilderness land (wilderness land as a fraction of the total area within each county) is negatively related to the level of economic activity, as measured by GDP per capita. Secondly, the fixed effects model reveals a negative relationship between economic growth and the reduction of wilderness land. These effects are tighter for wilderness land defined within a short distance from existing encroachments. A high level of economic activity and high economic growth per capita is therefore associated with less wilderness land and, hence, the study gives no support for any Environmental Kuznets Curve (EKC) relationships. © 2001 Elsevier Science B.V. All rights reserved.

Keywords: Wilderness land; Economic growth; Environmental Kuznets curve

1. Introduction

During the last decades the reduction of wilderness land has been substantial in Norway. When defining wilderness land as areas more than 5 km from the closest man-made encroachment, about half of the total area was wilderness land in the beginning of this century. In 1940 this proportion had declined to about 34%, while only 12% of

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the total area of Norway, i.e. 38 200 km^2, was made up of wilderness land according to the above definition in 1994 (DN, 1995). In the southern part of Norway, wilderness land counts for less than 5% of the total area today, and wilderness land is absent in three of the total 18 counties. The largest part of wilderness land is found in mountain areas, but also here this type of land is characterized by a high degree of fragmentation. Wilderness areas stretching from the mountains to the fjords are today very scarce, if not non-existent (DN, 1995).

The presence of wilderness land, and hence, the concern for land conversion of this type, is important for various reasons. First of all, fragmentation and reduction of wilderness land represent a threat to biological diversity and the health of the ecosystem (see, e.g. Perrings et al., 1995 for a general overview, and Alfsen and Sæbø, 1993 for a discussion of a broad set of indicators from Norway). Moreover, unspoiled natural resources, like wilderness land, carry recreation and amenity values (see, e.g. Porter, 1982 who also gives a good analytical exposition). It can also be argued that wilderness land covers an option value because, at least in some instances, consumption of wilderness land is irreversible (Fisher and Krutilla, 1985). When, say, building a dam for a hydropower project, it is very expensive, if not impossible, to bring the land back to its pristine state. This will also be so for land-use changes taking place in a tropical climate, but land restores generally faster in the tropics than in an alpine climate. Finally, because Norway still has a large amount of wilderness land and unspoiled nature compared to other countries in Europe, it can also be argued that Norwegian wilderness land represents an international public good value.

Land-use needs to be regulated because the total economic value of wilderness land, encompassing use values, option values and public good values, clearly is not internalised in the market prices of land-conversion. In Norway, as in most other countries, the regulation is not of the indirect Pigovian tax type, but of the direct type and is implemented basically by passing laws. The central instrument is Plan og Bygningsloven ('The Planning and Land-use Act') together with Naturvernloven ('The Natural Preservation Act') (Backer, 1990). The first is aimed at being a tool for political decision-making at the local level (municipals, counties) in issues concerning preservation and the co-ordination of different economic activities. Typically, the Government and the Parliament set up the various national land-use targets while the counties and municipalities develop comprehensive solutions taking local conditions into consideration. The Natural Preservation Act, on the other hand, is a more specific tool in preservation of wilderness land and species conservation, and regulates the land-use and use of protected areas such as National Parks and Protected Landscape areas.

Through The Natural Preservation Act, about 6.4% of the total land in Norway has obtained various forms of protected status (SFT, 1996). In spite of this, however, most of the wilderness land has been lost during the last decades. In what follows, we will present evidence and analyse factors affecting these changes during the very last years. The various types of man-made encroachments consist basically of road constructions, hydropower projects and electrical power lines. These projects are due to activities of several sectors such as agriculture and the forest industry, national defence, tourism and the hydropower industry. The total reduction of wilderness land is the sum of the small encroachments to which all these sectors contribute. The most important sector behind the land-use changes has been the forest sector because of road constructions for lorries and tractors transporting lumber (DN, 1995). The present study aims, however, not to explain the land-use changes at the sector level, or the direct sources behind land-use changes. Instead, we are interested in the underlying causes, or the driving forces, explaining reduction of wilderness land. The analysis is therefore carried out at the macroeconomic level, meaning that, among others, GDP per capita and GDP per capita growth are included as explanatory variables.[1] So while we are seeking to demonstrate

[1] Angelsen et al. (1999) distinguish between variables explaining deforestation and land-use changes at different levels; direct sources (forest sector, residential land, etc.), immediate causes (timber prices, various input prices, etc.) and the underlying causes (macro-level variables). We follow this taxonomy, and explaining land-use changes with macro economic variables represent accordingly the underlying causes.

that GDP per capita is an underlying cause behind land-use changes, it cannot be proven by our statistical analysis. However, an indication that the causality is not running the opposite direction is that the forest sector, the most important sector behind the land-use changes, contributes only slightly to total production.[2]

The following analysis has some similarities with deforestation and land-use conversion studies for tropical areas, but is as far as we know the only one of this type carried out for an area within an alpine climate. The study is at the county level (Norway consists of 19 counties, two of which are aggregated in the analysis) and covers the years 1988 and 1994; the only two periods with reliable wilderness land-use data as the evidence presented above for the other years represent only crude estimates. The period 1988–1994 is a quite short period, but there are large variations in the land-use changes among the counties. In addition to the above-mentioned type of wilderness land, i.e. areas more than 5 km from man-made encroachment, we will also present data and analyse to what extent economic growth is associated with consumption of wilderness land under less restrictive definitions.

The evidence is first presented in Section 2. In Section 3 we discuss briefly the connection between the underlying forces represented by macroeconomic changes and land-use changes, and the connection between economic growth and environmental changes in general. During the last decade this debate has been related to the presence, or absence, of so-called Environmental Kuznets Curve (EKC) relationships. In Section 4 we present the hypothesis between growth and land-use to be tested, and outline the models for the econometric analysis in Section 5. In the final section the main findings are summarised.

[2] Somewhere between 0 and 3% of GDP is coming from the forest sector in various counties (Statistics Norway: Fylkesfordelt Nasjonalregnskap (Regional National Account)).

2. The data and the evidence

As mentioned, the amount of wilderness land referred to in the introduction for the years 1900 and 1940 are only crude estimates for areas more than 5 km from man-made encroachments. More accurate data exists only for 1988 and 1994 due to a recently finished project carried out by The Directorate for Nature Management (DN). In an international perspective these data from the DN are probably unique, and are as far as we know the only well-documented land-use data of fairly good quality covering man-made encroachments in a systematic way. The Appendix A gives further details on the construction of these data.

By using the database from DN, categorising the total area of Norway into four zones with the distance from larger technical installations as the basic indicator, we have defined wilderness land in three ways. These three types of land cover land more than 5 km from man-made encroachment, WL5, land more than 3 km from man-made encroachment, WL3, and finally, land more than 1 km from man-made encroachment, WL1. WL5 is therefore a subset of WL3, which again is a subset of WL1. Thus, we have wilderness land in three different classes that can be interpreted as three different indicators of environmental quality. In the following analysis, these three types of land are expressed as relative magnitudes; that is, as a percentage of the total amount of land within each county.

For Norway as a whole, the amount of wilderness land of type WL5 declined from about 39 500 km^2, or 12.2% of the total land in 1988, to 38 200 km^2, or 11.8%, in 1994, representing a yearly reduction of 0.5%. For WL3 the fractions were 18.2 and 17.6%, respectively, representing a reduction of 0.6% per year, while WL1 declined from a fraction of 43.1% in 1988 to 42.0% in 1994, a reduction of 0.4% per year. Hence, for the country as a whole, the largest relative reduction took place for WL3, land more than 3 km from man-made encroachment. The reduction of wilderness land observed all through the century has therefore continued during the 1980s and 1990s, but at a lower speed. Based on the crude estimates referred to in the introduction, which

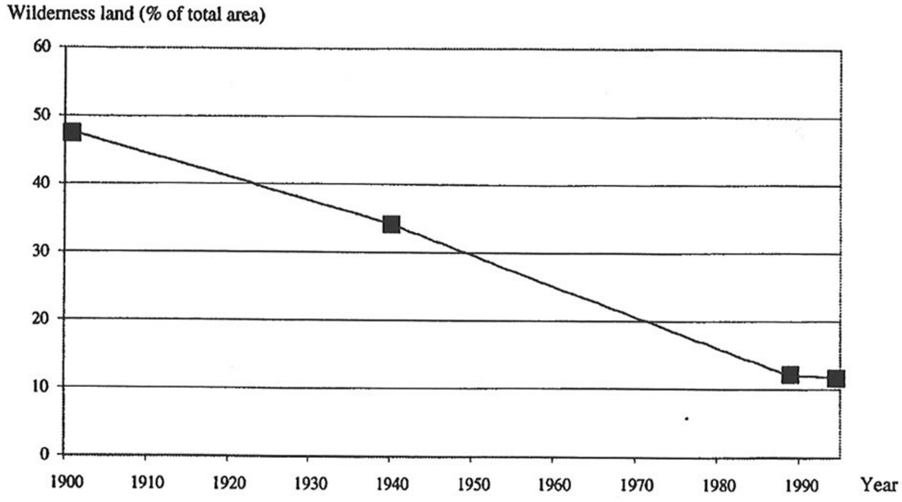

Fig. 1. Wilderness land more than 5 km from man-made encroachments as a percentage of the total area (WL5) 1900, 1940, 1988 and 1994 for Norway as a whole.

corresponds to WL5, the yearly average reduction was 0.9% between 1900 and 1940, and 2.1% between 1940 and 1988. See Fig. 1.

Table 1 gives the summary statistics at county level and comprises the land-use data used in the regressions in Section 5.[3] As seen, the amount of wilderness land varies considerably between the 18 counties to be analysed. Evaluated by the coefficient of variation, the dispersion is largest for WL5 while it is smallest for WL1. Hence, the largest variation is related to wilderness land farthest from man-made encroachment. Altogether three counties had no land of type WL5 in 1994 and in 1988, and hence, no reductions happened there. The largest reduction of wilderness land took place in the north, in the Nordland county. This county together with Troms and Finnmark, also counties in the far north, have the largest fractions and the largest absolute amount of wilderness land. For more details, see the Appendix A.

[3] Notice the discrepancy between the averages of WL5 in Table 1 and Fig. 1. Because Table 1 gives unweighted averages over the countries while Fig. 1 gives WL5 for Norway as a whole, the regional presence of wilderness land is therefore biased in the sense that the largest counties have the largest fraction of wilderness land.

3. Economic growth and the use of wilderness land

As discussed in the Section 1, we want to analyse how macroeconomic variables affect consumption of wilderness land. At least since the middle of the 1960s, there has been an ongoing discussion whether continued economic growth is compatible with a 'sustainable' use of renewable as well as non-renewable resources. The present analysis has therefore strong links to this discussion. On the one hand, environmentalists and others such as the Club of Rome, have argued that the finiteness of the natural resource base must put an end to continued economic growth. On the other hand, particularly neoclassical economists, have argued that there are sufficient substitution possibilities between natural capital and man-made capital to make further growth possible. In addition, some have argued that technological progress will reduce the dependence on natural resources. As is well known, the Brundtland report (WCED, 1987) supported the view that economic growth was the only way out of poverty and environmental degradation. This position is also taken, among others, by Beckerman (1992) and in a more modest way by international

Table 1
Descriptive statistics wilderness land 1988 and 1994, 18 counties

Wilderness land as a fraction (in%) of total county area	Mean	Standard deviation	Coefficient of variation	Minimum	Maximum
WL588 (land more than 5 km from encroachment, 1988)	7.1	9.4	132.4	0.0	37.8
WL388 (land more than 3 km from encroachment, 1988)	14.5	13.9	95.9	0.0	54.4
WL188 (land more than 1 km from encroachment, 1988)	35.7	21.5	60.2	0.7	78.8
WL594 (land more than 5 km from encroachment, 1994)	6.9	9.2	133.3	0.0	37.5
WL394 (land more than 3 km from encroachment, 1994)	14.1	13.7	97.2	0.0	54.1
WL194 (land more than 1 km from encroachment, 1994)	34.7	21.3	61.4	0.6	78.5

agencies such as the World Bank, see e.g. World Bank (1992). On the other side Daly (1987), Daly and Cobb (1988) and Ayres (1996), among others, argue that the substitution possibilities are limited and technological progress cannot change the fundamental laws of thermodynamics.

This controversy is basically of an empirical nature, but only during the last decade or so the relationship between economic activity, environmental performance and the depletion of natural resources have been analysed systematically for various environmental indicators. These studies have to a large extent been interpreted in light of the Environmental Kuznets Curve (EKC) hypothesis. This hypothesis maintains that there exists an 'inverted U-shaped' relationship between environmental degradation and economic activity, as measured by income per capita. That is, it states that environmental degradation will initially increase, reach a peak, but eventually decline as per capita income increases. The EKC-relationship has its reference to Simon Kuznets because of his famous findings of an inverted U-shaped connection between income inequality and GDP per capita for different countries (Kuznets, 1955).

The great interest in analysing possible EKC-relationships is therefore whether economic growth is harmful for the environment or not, and consequently, whether economic growth as a goal for economic policy, at least in the long term, means environmental improvements. In addition, studying a possible EKC-relationship hypothesis is clearly testable. It can be tested for different geographic areas, over time and for various indexes of environmental degradation, flow pollution as well as for depletion of natural resources. The main results of the EKC-studies, however, seem to be that there are no general examples of curvature à la Kuznets. The exception is various flow pollution problems of local character, meaning that the pollution level will be 'low' for a 'low' income per capita level, 'high' for a 'medium' size income level and then again 'low' when income per capita is 'high' (see, e.g. Cole et al., 1997). Hence, as Barbier (1997) concludes in a review of this literature, the EKC-studies offer very little support for the strong policy conclusion that economic growth alone is the solution to environmental problems. Rather, it is clear that specific policies to protect the environment are necessary to reduce environmental degradation problems. This view is also supported by the economy–ecology manifest by Arrow et al. (1995).

Recent empirical studies on land-use and economic growth have focused on deforestation (see, e.g. Cropper and Griffiths, 1994; Antle and Heidebrink, 1995; Panayotou, 1995), and expansion of agricultural land (James, 1998). The results from these studies are mixed, and in the deforestation studies evidence claimed to support EKC-relationships has been found. However, these results are derived under the assumption that the rate of growth, i.e. changes in the degree of deforestation, not the level of forest cover, is regressed on income per capita data. Hence, the study of Cropper and Griffiths concludes that environmental degradation, as measured by the growth rate of deforestation, becomes less when the income per capita becomes high (however, this does not hold for Asia). The main findings of the land-use study of James (1998) are that for a large sample of developing countries, cropland as a fraction of the total amount of arable land first increases and then decreases when income per capita becomes high. Because the fraction of non-agricultural land is taken as a proxy for biodiversity, the study also argues that biodiversity increases as the income per capita becomes high. However, due to the fact that the loss of biodiversity often is irreversible, the study's proxy for biodiversity may be questionable.

The following empirical study has some similarities with Cropper and Griffiths (1994) and James (1998) as it is carried out at the macro economic level. However, while they are analysing deforestation and land-use changes at the country level within the tropical climate zone, we are studying changes taking place at the county level within a country in an alpine climate. Moreover, we are studying the effects on land of different classes, or qualities.

4. Hypotheses and presentation of the econometric relations

The analysis will, as already indicated, include econometric relations for three types of land as dependent variables. In contrast to earlier studies, the present analysis has therefore the advantage of studying different attributes of the same type of natural resource, i.e. wilderness land. The explanatory variables comprise real GDP per capita, real GDP per capita squared, and the population density. As in Cropper and Griffiths (1994), it obviously would have been interesting to include additional explanatory variables, say, the opportunity cost of preserving wilderness land (which, among others, could be approximated by timber prices). Partly because of the lack of reliable data, partly because of the relatively low number of observations in our data set and no variation in timber prices by county, this is not done. However, most important, it can be misleading to introduce variables at different levels because price variables, such as timber prices, can be classified as immediate causes of land-use changes, contrasting the underlying forces as represented by variables like GDP per capita and population density (cf. footnote 1).

The empirical analysis is at county level (18 counties) with the above three mentioned types of wilderness land as dependent variables. The analysis is first carried out as simple cross-section regressions for 1988 and 1994 in order to examine the relationship between the level of economic activity and environmental degradation, i.e. the relative amount of wilderness land. The cross-section models are given as

$$\mathrm{WL}_i = \alpha_0 + \alpha_1\,\mathrm{GDPC}_i + \alpha_2\,(\mathrm{GDPC}_i)^2 + \alpha_3\,\mathrm{DEN}_i + \mathrm{u}_i, \tag{1}$$

where WL_i is wilderness land (three types) in 1988 and 1994 in county i, GDPC_i is real GDP per capita and DEN_i is the population density for the same years. α_0 is an intercept term while u_i is a white noise error term. As mentioned, WL_i is measured as a percentage of the total land (in each county), GDPC_i is measured in 1000 NOK per capita in fixed prices, while DEN_i is given as the number of people per km^2 (see the Appendix A for more details). A negative sign of α_1 means a negative environmental impact, while a positive sign of α_2 counterbalances and eventually indicates a positive relationship between income per capita and the amount of wilderness land. DEN_i is a well-established variable in macro-level land-use studies and is included to control for demographic and geographic differences, but the causal link is far from clear. As in Cropper and Griffiths (1994) and James (1998), the à priori effect is assumed to be negative. Eq. (1) is estimated separately for each type of land.

We are also interested in the connection between income and environmental degradation over time. This relationship is examined in two ways. First, the two cross-sections data sets are combined to form a pooled sample. By using a fixed effects model it is then possible to identify the impact of increased income over time on the changing amount of wilderness land; that is, the effect of economic growth can be estimated. In these relations the intercept term for each county is assumed to be correlated with the explanatory variables. Application of a difference operator in the calculation of the coefficients leads the time in-variant variables to drop out of the estimated relation. In this way, the various intercept terms will include all county specific effects, except of variations in GDP per capita and population density, and, hence, the estimated effect of the explanatory variables will be time-specific. The limitation of such an analysis is that we deal with only two observations in time, meaning that the results hold only for this specific time period and cannot be generalised.[4]

The fixed effects model is then given as

$$\mathrm{WL}_{i,t} = \beta_1\,\mathrm{GDPC}_{i,t} + \beta_2\,(\mathrm{GDPC}_{i,t})^2 + \beta_3\,\mathrm{DEN}_{i,t} + v_{i,t}, \qquad (2)$$

where subscript t refers to time, 1988 and 1994, respectively. $v_{i,t} = \alpha_{0i} + \eta_{i,t}$, where $= \alpha_{0i}$ now is an intercept term specific for county i and $\eta_{i,t}$ is a white noise error term. The à priori effect of GDPC is again negative, while a positive effect of $(\mathrm{GDPC})^2$ indicates that the effect of economic growth yields a smaller negative impact for a higher income county than a lower income county. The effect of DEN_i is here to be interpreted as the impact of demographic changes over time because the effects of time invariant differences (geographical) now are represented by the varying intercept terms. Eq. (2) is also estimated separately for each type of land.

An alternative approach to modelling time variations is to study the relationship between the relative change in wilderness land between 1988 and 1994 and the level of the previous explanatory variables, i.e.

$$\frac{\mathrm{WL94}_i - \mathrm{WL88}_i}{\mathrm{WL88}_i} \times 100 = \delta_0 + \delta_1\,\mathrm{GDPC88} + \delta_2\,(\mathrm{GDPC88}_i)^2 + \delta_3\,\mathrm{DEN88}_i + \varepsilon_i, \qquad (3)$$

$\mathrm{WL88}_i$ and $\mathrm{WL94}_i$ are the fractions of wilderness land in 1988 and 1994 in county i (again, three types of land) and where the changes are measured as percentages. $\mathrm{GDPC88}_i$ and $\mathrm{DEN88}_i$ represent GDP per capita and the population density for the start year, respectively, and ε_i is again a white noise error term. By specifying the explanatory variables for the start year, relation Eq. (3) is formulated just as models frequently used in economic growth studies, see e.g. Barro (1991). This type of formulation is also applied by Cropper and Griffiths (1994). However, because they have more observations over time, they estimate this equation as a fixed effects model. A negative effect of GDPC and a positive effect of $(\mathrm{GDPC})^2$ indicates here that the marginal decline of wilderness land becomes smaller, and eventually stops up and becomes positive, as the income level increases.

5. The regressions

5.1. Cross-section analysis for 1988 and 1994

We start to look at the cross-section analysis of Eq. (1), and the results are reported in Table 2. For all regressions there is a negative effect of GDP per capita, a positive effect of GDP per capita squared, and a negative effect of the population density DEN as à priori expected. Some of the t-statistics reveal a high degree of significance. The (absolute) t-values for all variables are higher in the cross-sections for 1988 than in 1994 and when wilderness land is defined within a short distance of existing encroachments. The coefficient of determination is also somewhat higher in 1988 than in 1994 and when wilderness land is defined more broadly. Thus, the relationship between the level of economic activity, population density and wilderness land is strongest for wilderness land defined more broadly, and statistically stronger in 1988 than in 1994.

[4] The fixed effects model is preferrred to a random effects model because the analysis comprises all the regions, not just a sample of the population (Kennedy, 1992).

Because the small effect of $(GDPC)^2$, the first order term GDPC dominates and it is a downward sloping relationship between income per capita and land-use for income levels relevant for the sample. Counties with a relatively high economic activity per capita, when controlled for population density, are therefore likely to have a small amount of wilderness land, and vice versa. The exception to this main rule is to some extent wilderness land in the broadest sense, WL1, where the effect from the $(GDPC)^2$ is relatively strong, especially in 1988. Fig. 2 depicts this estimated relationship together with the actual data. Also this relationship, however, yields basically a negative relationship within the empirical range of the GDP data (Akershus/Oslo county with an income of 135.1 is omitted from the figure, see Appendix A).

As demonstrated, the effect of income per capita varies for the various definitions of wilderness land, and GDPC has a stronger and more robust effect on wilderness land of the less restrictive type. Hence, particularly wilderness land within a small distance from already existing infrastructure tends to be converted into new development when the economic activity becomes higher. The income elasticity, however, becomes smaller as the definition of wilderness land becomes less restrictive. Evaluated at the mean income value and mean population density, we find that one percent higher income in a region compared to the average region goes hand-in-hand with about 10% lower value of the fraction of wilderness land of type WL5 in 1994. The income elasticity of WL394 is, on the other hand, about −4.7 while it is about −0.2 for WL194. The absolute values of the elasticities are somewhat smaller in 1988, but the differences are more or less the same, and altogether this indicates that higher economic activity consumes relatively more wilderness land as the distance to man-made encroachments increases.

Above it was also observed that the effects of DEN are negative, and these effects are quite substantial. Hence, according to Table 2, one more person per km^2 means a reduction in the fraction of wilderness land ranging from 0.14 to 0.63% points. This indicates that the consumption of wilderness land is greatly influenced by crowding in a broad sense, interpreted as county-specific demographic and geographic factors in this cross-section analysis. The population density effect is particularly strong for wilderness land of the least restrictive character, WL1. This result fits intuition, as the other types of wilderness land are located further away from urban areas, already existing infrastructure and developed land.

Table 2
Cross-section regressions 1988 and 1994[a]

	WL588	WL594	WL388	WL394	WL188	WL194
Intercept	118.04	139.31	185.32	205.33	233.48	206.67
	(2.12)	(2.09)	(2.57)*	(2.44)*	(2.69)*	(2.09)
GDPC	−2.35	−1.99	−3.64	−2.90	−4.32	−2.72
	(−2.06)	(−2.02)	(−2.46)*	(−2.32)*	(−2.37)*	(−1.82)
$(GDPC)^2$	0.01	0.01	0.02	0.01	0.03	0.01
	(2.12)	(2.04)	(2.57)*	(2.36)*	(2.62)*	(2.04)
DEN	−0.14	−0.16	−0.28	−0.31	−0.61	−0.63
	(−2.75)*	(−2.08)	(−3.73)**	(−2.52)*	(−4.80)**	(−3.43)*
R^2	0.37	0.35	0.49	0.46	0.67	0.65
R^2, adjusted	0.23	0.21	0.39	0.34	0.60	0.57
N	18	18	18	18	18	18

[a] Dependent variable wilderness land (in% of total county area). *Note:* For definitions of dependent variables see Table 1.
* Statistically significant at 5% level.
** Statistically significant at 1% level. t-statistics in parentheses.

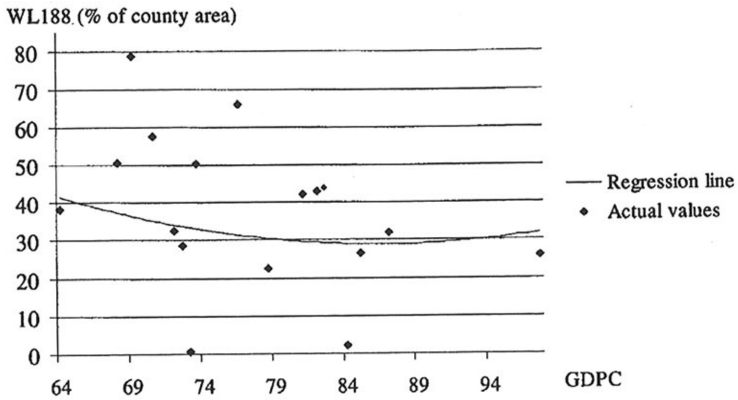

Fig. 2. Estimated cross-section relationship between GDP per capita and WL188. Evaluated at the mean value for the population density. *Note:* For definitions of dependent variables see Table 1

5.2. Pooled regression, fixed effects models

The results from the fixed effects models as given by Eq. (2) are reported in Table 3. The fixed effects models capture in a single equation the changing amount of wilderness land due to increased GDP per capita for all counties over the time period from 1988 to 1994. By assuming the intercept term being correlated with the explanatory variables and letting the intercept vary across the individual counties, the effect of economic growth is separated from the time invariant effects. However, as already mentioned, a weakness of the present pooled cross-section and time series analysis is the low number of observations as it comprises observations for only 2 years.

$(GDPC)^2$ yields no significant effects and when omitted, GDPC always has a negative effect. However, the effect is not significant for WL5. Economic growth, controlled for the population density growth, therefore significantly consumes wilderness land closest to already existing encroachments. On the other hand, and contrary to the cross-sectional analysis, the effects of DEN are quite weak. The reason is that there are small changes in this variable over time, which is confirmed by the fact that the sign changes as $(GDPC)^2$ is included or not.

On average, income per capita increased from 80.6 in 1988 to 114.0 in 1994 (1000 NOK in fixed 1986 prices, see Appendix A). Hence, for a county growing at an average rate, the estimated effect means a reduction of WL1 of about 1.15% points over this 6-year period. This effect is in accordance with the actual change as given in Table 1. For the average county, the income elasticities are found to be about -0.08 for all three types of wilderness land. Although the patterns are the same as in the cross-sectional analysis, the estimates based on the fixed effects model are clearly lower in absolute value. Hence, the effect of higher economic activity on the consumption of wilderness land is greater over cross-section than over time.

5.3. Relative changes over time

Finally, we present the results from the regressions where the relative change in wilderness land is the dependent variable and the explanatory variables represent the situation in the start year 1988, i.e. Eq. (3) above. Evaluated by the t-statistics it is seen from Table 4 that the estimated relationships between income at the starting point and degradation over the subsequent years are not very tight, and none of the coefficients are significant. The conclusion is therefore that the initial level of economic activity, or population density, does not significantly affect the subsequent change in wilderness land over time. This

Table 3
Pooled regressions, fixed effects models[a]

	WL5		WL3		WL1	
GDPC	−0.0102	−0.0943	−0.0183	−0.0828	−0.0344	−0.0092
	(−2.06)	(−1.22)	(−0.34)**	(−0.92)	(−7.27)**	(−0.51)
$(GDPC)^2$	–	0.0005	–	0.0004		−0.0001
		(1.15)		(0.76)		(−1.71)
DEN	0.1733	−0.2366	0.2415	−0.0727	0.1907	0.3016
	(1.54)	(−0.92)	(1.87)	(−0.24)	(5.15)**	(5.95)**
N[b]	30	30	30	30	36	36

[a] Dependent variable wilderness land (in % of total county area). *Note:* For definitions of dependent variables see Table 1.
[b] Three counties missing wilderness land of type WL5 and WL3 in 1988 and 1994 omitted (for details, see Appendix A).
** Statistically significant at 1% level; t-statistics in parentheses.

outcome contrasts with the above-mentioned results of Cropper and Griffiths (1994), which found a statistically significant effect between the degree of deforestation (forests and woodland area) and income in a sample of African and Latin-American countries, and a declining effect of income as income becomes higher.

6. Concluding remarks

Using a new and probably unique database, categorising the total area in Norway into different zones depending on the distance from larger man-made technical installations, wilderness land has been defined in three ways. These types of wilderness land reflect different qualities of the same type of natural resource, and factors affecting the consumption of these resources have been analysed in a macroeconomic context for 18 counties and over the period 1988–1994. The macroeconomic context means that we are dealing with the underlying, and not the direct sources, behind land-use changes, and the analysis is related to the ongoing debate about economic growth, environmental performance and use of natural resources; the presence or not of Environmental Kuznets Curve (EKC) relationships.

The main results from the analysis can be summarised as follows. First, from the cross-sections analysis it is found that the level of economic activity, correcting for variations in population density, explains between 46 and 65% of the variations in the amount of wilderness land among the counties for the two broadest categories of wilderness land. Hence, in these cases, the higher level of GDP per capita in a region, the less wilderness land. Moreover, the higher population density, the less wilderness land when correcting for differences in income per capita. Secondly, the fixed effects models reveal a negative, and linear, connection between GDP per capita growth and conversion of wilderness land; the higher the economic growth, the higher the consumption of wilderness land. This holds again significantly for the two broadest categories of wilderness land, and the effect is most tight for the less restrictive defini-

Table 4
Cross-section regressions[a]

	REWL5	REWL3	REWL1
Intercept	90.849	46.504	26.001
	(1.55)	(1.31)	(1.01)
GDPC	−2.566	−1.296	−0.683
	(−1.73)	(−1.50)	(−1.08)
$(GDPC)^2$	0.018	0.008	0.004
	(1.83)	(1.61)	(1.03)
DEN	−0.170	−0.057	−0.136
	(-1.04)	(−0.83)	(−1.43)
R^2	0.17	0.12	0.64
R^2, adjusted	−0.06	−0.12	0.53
N	15	15	18

[a] Dependent variable relative change in wilderness land 1988–1994 (in%). *Note:* For definitions, see Eq. (3). t-statistics in parentheses.

tion; that is, land more than 1 km from man-made encroachments.

Hence, both the economic activity level and changes in the economic activity tend to have a negative impact on the amount of wilderness land. The study gives therefore no support to any EKC-relationship. This is not at all surprising as conversion of wilderness land often represents an irreversible process, and it takes generally more time to restore land back to its original state in an alpine climate, as in Norway, than in a tropical climate. But even in an alpine climate wilderness land is basically a renewable resource and hence, the possibility of finding positive growth and income effects should be present. If present in the cross-section context, however, it represents no evidence of a 'growth is not harmful for the environment argument' because such a relationship has to do with changes over time. In the literature this fact is often confused. For a clarifying discussion, see de Bruyn et al. (1998). While giving no support to any EKC-relationship, the other main finding of the study is that the impact of economic activity on consumption of wilderness land is generally stronger as wilderness land is defined more broadly. Hence, as income increases in Norway, wilderness land tends to be developed within a small distance, not a large distance, from already existing encroachments.

By using the results from the fixed effects model it is possible, in an ad hoc manner, to look at some possible scenarios of the future consumption of wilderness land in Norway. We strongly emphasize that this forecasting is statistically not valid; it is only meant as a broad illustration (see Stern et al., 1996 for a parallel exercise). Under a low-growth of 1% per annum GDP per capita expansion, taking place in all counties over the 20-year period 1994–2014 and accompanied by no changes in the population so that the population density is fixed, wilderness land will be reduced by an amount of 4% −9% of the existing land in 1994. The largest relative reduction takes place for the least restrictive definition of wilderness land WL1, so growth is less harmful for the most remote type of wilderness land (see Table 3). Under a 'business as usual' scenario of 3% per annum GDP per capita growth, still supposed to be uniform over the regions and with no population changes, the reductions will be between 15 and 32%. Hence, about 1/3 of the existing wilderness land more than 1 km from man-made encroachment will disappear over this 20-year period under this scenario. The results under this scenario can hardly be said to represent a sustainable development path.

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Appendix A

A.1. The wilderness data

Direktoratet for Naturforvaltning (The Directorate for Nature Management, DN), together with Statens Kartverk (The Norwegian Mapping Authority), collected land-use data from all Norwegian municipalities for the years 1988 and 1994. The land was grouped according to the distance from larger technical encroachment, and the following categories were used:

1. Area less than 1 km from larger technical encroachment.
2. Area more than 1, but less than 3 km from larger technical encroachment.
3. Area more than 3, but less than 5 km from larger technical encroachment.
4. Area more than 5 km from larger technical encroachment.

Technical encroachment is defined as public roads, railways (except tunnels), forest roads, roads for lorries and tractors, power-lines, power stations, pipelines and regulations of rivers and lakes. This type of land classification clearly includes various errors. One main source of error is related to transforming handdrawn maps into

Table A1
Wilderness land (% of total county area)

County	WL588	WL388	WL188	WL594	WL394	WL194	Total area (km^2)
Østfold	0.0	0.0	2.1	0.0	0.0	1.7	4171.3
Akershus/Oslo	0.0	0.0	1.3	0.0	0.0	1.1	5368.2
Hedmark	2.2	8.0	28.5	2.2	7.8	27.3	27 401.1
Oppland	8.4	17.6	38.3	8.3	17.6	38.1	25 226.0
Buskerud	1.3	6.4	26.4	1.2	6.2	25.2	14 938.6
Vestfold	0.0	0.1	0.7	0.0	0.1	0.6	2222.4
Telemark	4.9	10.3	31.9	4.8	10.0	30.8	15 336.5
Aust-Agder	4.2	11.6	32.7	4.0	11.0	31.2	9189.5
Vest-Agder	0.7	4.2	22.5	0.6	4.1	21.4	7308.3
Rogaland	0.4	3.8	26.0	0.4	3.7	25.2	9224.3
Hordaland	6.6	14.5	43.1	6.3	13.9	42.1	15648.6
Sogn og Fjordane	4.7	15.7	50.2	4.4	14.9	48.9	18679.5
Møre og Romsdal	4.8	12.6	42.3	4.7	12.3	40.9	15 139.0
Sør-Trøndelag	5.4	15.2	43.9	5.2	14.3	42.4	18 823.1
Nord-Trøndelag	13.9	25.8	50.7	13.3	24.8	49.4	22 375.2
Nordland	13.7	27.0	57.5	11.7	24.8	55.2	38156.0
Troms	18.8	34.4	66.0	18.8	34.1	65.3	26 005.2
Finnmark	37.8	54.4	78.8	37.5	54.1	78.5	48 791.7
Average	7.1	14.5	35.7	6.9	14.1	34.7	
Coefficient of variation	132.4	95.9	60.2	133.3	97.2	61.4	

Table A2
Explanatory variables

County	GDPC 1988	GDPC 1994	DEN 1988	DEN 1994
Østfold	84.0	104.2	61.2	61.5
Akershus/Oslo	135.1	192.4	172.8	182.9
Hedmark	72.4	103.0	7.2	7.1
Oppland	63.9	100.3	7.6	7.6
Buskerud	84.9	114.9	16.2	16.5
Vestfold	73.2	105.1	91.6	95.0
Telemark	87.2	107.2	11.5	11.5
Aust-Agder	71.8	102.1	11.4	11.7
Vest-Agder	78.3	117.1	21.0	21.9
Rogaland	97.4	134.6	39.0	41.4
Hordaland	82.0	120.8	27.3	28.2
Sogn og Fjordane	73.4	118.3	5.9	6.0
Møre og Romsdal	81.2	111.9	16.3	16.5
Sør-Trøndelag	82.3	109.4	14.0	14.4
Nord-Trøndelag	68.0	98.2	6.0	6.1
Nordland	70.5	100.0	6.6	6.7
Troms	76.5	112.0	5.8	6.0
Finnmark	69.0	100.7	1.6	1.7
Average	80.6	114.0	29.1	30.2
Coefficient of variation	19.7	19.0	146.4	148.7

digitised data. Another source of error is due to the fact that data for roads built by the national defence are not available for security reasons. DN (1995) discusses the various measurement problems and sources of errors. The data used in the analysis are reported in Table A1.

A.2. The explanatory variables

GDP per capita data and population data are from Statistics Norway: Fylkesfordelt Nasjonalregneskap (Regional National Account). Because yearly regional national account data are not available, GDP per capita 1986 represents the first period (1988) and GDP per capita 1992 represents the second period (1994). National account data on a regional level is available only in current prices. The current prices are converted to fixed prices using indices from Statistics Norway: Årlig nasjonalregnskap 1978–1997 (Yearly National Accounts 1978–1997) (Table A2). GDPC (GDP per capita) in 1000 NOK fixed 1986 prices, DEN (population density) in persons per km^2.

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