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Economic Efficiency in Waste Management and Recycling

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Abstract

This thesis consists of four self-contained papers that all deal with economic efficiency issues with respect to recycling behavior and waste management policies.

Paper [1] provides an econometric analysis of the most important determinants of inter-country differences in waste paper recovery and utilization rates. The paper concludes that relative waste paper recovery and use depend largely on long-standing economic factors such as population intensity and competitiveness in the world market for paper and board products. We also find evidence that supports the conjecture that rich countries tend to recover relatively more waste paper than low-income countries, which reflects the higher demand for waste management and environmental policies in more developed economies. As recovery and utilization rates are determined by economic and demographic characteristics the degree of policy flexibility in affecting these rates may be limited. In particular, an ambitious utilization rate target may be very costly to enforce as it can conflict with existing trade patterns of paper and board products as well as with other environmental goals.

Paper [2] builds and extends upon paper [1] and provides a critical analysis of Van Beukering and Bouman's article in *World Development* on global paper recycling and trade. We first question their notion that developing countries specialize in waste paper utilization and developed countries in recovery activities primarily because of different patterns of waste paper trade. An increased focus on relative waste paper availability, we argue, provides us with a better understanding of global paper recycling. We also criticize some of the implicit assumptions made in their regression analysis of waste paper utilization rates. In contrast to the approach used by Van Beukering and Bouman our analysis: (a) is consistent with basic microeconomic theory; (b) distinguishes clearly between short- and long-run impacts; and (c) produces results that support our initial conjecture that waste paper availability is the most important determinant of waste paper use.

Paper [3] analyzes the spatial cost efficiency of the Swedish legislation regarding waste disposal handling. We focus on the case of corrugated board and recognize that the different counties in Sweden possess different economic prerequisites in terms of waste paper recovery and utilization potential. We employ data for six corrugated board mills and 20 counties and a non-linear programming model to identify the least cost strategy for reaching the politically specified recycling target of a 65 percent recovery rate for corrugated board. That is, the total costs of recovering a minimum of 65 percent in each county are calculated and compared with the case when the country as a whole recovers 65 percent cost effectively. The conclusion is that from an efficiency point of view the recovery efforts should be concentrated to the highly populated and urbanized counties, and not be uniformly divided throughout the country. In the base case the results suggest that the cost efficient county-specific recovery rates should range from 51 percent to 72 percent.

Paper [4] analyzes households' perceptions of recycling activities in a municipality in northern Sweden, Piteå. The purposes of the paper are to analyze whether moral motives matter for: (a) the assessment of households' waste sorting costs; and (b) for the efficiency of introducing economic incentives for stimulating households' recycling efforts. We employ an economic model of moral motivation with possible motivation crowding-out and econometric techniques. The empirical results support the notion that moral motives significantly lower the costs associated with household recycling efforts. Specifically, the average hourly willingness to pay to let others sort household waste at source was found to be significantly lower than the corresponding income after tax (i.e., the opportunity cost of time). Furthermore, moral motives can in some cases be the cause of inefficient policy outcomes when introducing economic incentives to promote recycling efforts.

To Mom and Dad

List of Papers

This dissertation contains this introduction and the following papers:

Paper [I]: Berglund, C., and P. Söderholm (2002). An Econometric Analysis of Global Waste Paper Recovery and Utilization, re-submitted to *Environmental and Resource Economics.*

Paper [II]: Berglund, C., and P. Söderholm (2003). Complementing Empirical Evidence on Global Recycling and Trade of Waste Paper, forthcoming in *World Development*, Vol. 31, No. 4.

Paper [III]: Berglund, C. (2002). Spatial Cost Efficiency in Waste Paper Handling: The Case of Corrugated Board in Sweden, submitted to *Environmental and Resource Economics.*

Paper [IV]: Berglund, C. (2003). Households' Perceptions of Recycling Efforts: The Role of Personal Motives.

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See you around, buddy boys!

Christer Berglund

Luleå, January 2003

Preface

1. Introduction

The overall purpose of this thesis is to analyze a number of economic efficiency issues with respect to recycling behavior and waste management policy.

We live in an age of escalating environmental consciousness, where recycling of waste materials takes place on an increasing scale and in almost every area of the society. Recycling has traditionally occurred because it has been economical. From the 1970s and onwards, however, the perception in modern rich societies has been that we should recycle even more, something that is expressed by existing or proposed solid waste legislation. In Sweden, since 1994/95, the producer responsibility ordinance has governed the collection and recycling of packaging materials, newsprint, cars and tires. Moreover, environmental legislators in many European countries require that a large part of the waste paper flow should be recycled rather than burnt or deposited in landfills. A result of this is that a "waste hierarchy" has emerged. For instance, the following waste management options are outlined: reduce, reuse, recycle, incinerate and landfill, where the first three are considered "good" options and the last two "bad" ones. This hierarchy was first put forward by environmental organizations such as *Friends of the Earth* and *Greenpeace*. The European Union uses the same order of preference in its directives for packaging waste (Ackerman, 1997).

Thus, far-reaching policy measures have been undertaken in many economically advanced countries, to promote recycling of municipal waste. The dramatic increase in the number of recycling programs in operation is partly a function of non-economic influences such as changes in political concerns, often trigged by voter tastes for the environment (Kinnaman and Fullerton, 2000). However, there exist relatively few economic analyses in support for this policy stance, and the rationale for and the understanding of the consequences of different policy measures are thus often less than complete. Baumol (1977) stresses that from an economic point of view there is nothing inherently benign about recycling. Recycling policies ought, just as any other policies, to pass standard cost-benefit and economic efficiency tests. Baumol concludes (p. 86):

> [R]ecyling is not a free activity. It uses up resources to produce resources and it is therefore not always certain that the marginal net product must be positive.

Recent research efforts have indicated that not all the methods designed to reduce solid waste are paragons of economic virtue (e.g., Palmer et al., 1997). In many cases they are even based on a single-dimensional theory of value, which fails to take proper account of economic scarcities other than the one on which they focus their attention. For example, a number of recycling efforts and policies to promote such efforts tend to focus entirely on their benign environmental impacts, while abstracting from other important costs to society (e.g., Radetzki, 2000). The differences in economic costs across policies can thus often be substantial. The above provides the rationale for a closer study on the efficiency of a set of recycling policies.

We focus in particular (but not solely) on waste paper recycling. The rationale for this focus stems mainly from the following facts. *First*, there is a long tradition of policies aimed at boosting waste paper recycling, but so far we know relatively little about the economic efficiency of these policies. *Second*, waste paper recycling is a global phenomenon (and not just something that is present in developed economies with ambitious waste management policies). In other words, in many countries paper recycling occurs because it is economical, and it is thus not policy-driven. *Third*, waste paper is by far the largest component in the overall municipal solid waste stream. The composition of the municipal solid waste in one advanced economy, the USA, is shown in Table 1. Many European countries have a similar composition of their municipal solid waste.

Table 1: Municipal Solid Waste Composition by Weight in the USA (2000)

Category	Percent of Total Waste
Glass	5.5
Metals	7.8
Plastics	10.5
Food Wastes	10.9
Paper/Paperboard	***38.1***
Rubber, Leather, and Textiles	6.6
Yard Wastes	12.1
Wood	5.3
Other	3.2

Source: EPA (2002).

We also devote considerable attention to households' recycling efforts, this since local and national authorities use both economic and political instruments in the attempt to induce households to contribute to sustainable development. There has hence been an increased burden on the households to sort, clean and transport their waste to recycling centers. However, there is a lack of understanding of how these tools interplay with the motives held by households and the daily constraints they face. We therefore know relatively little about the economic efficiency of the policies in force.

Before proceeding, however, we need to discuss what is meant by efficient recycling policies and behavior. The papers enclosed in this thesis partly employ different approaches to recycling efficiency, and it is useful to outline these in detail.

2. Efficiency Criteria in Recycling Policy

Vilfredo Pareto introduced the most common economic definition of efficiency in 1906. He stated that an efficient situation is one in which it is impossible to make one person better off without making anyone else worse off (Pareto, 1906). Unfortunately, however, the Pareto criterion cannot be used to evaluate most projects, since generally projects make at least one person worse off. In such cases, one may instead apply the compensation principle as suggested by Hicks (1939) and Kaldor (1939). The Hicks-Kaldor criterion states that economic efficiency implies that the sum of the benefits is great enough to offset the costs, whether or not those benefits are used to compensate those who bear the costs.[1] It should be noted that actual compensation is not required, only that compensation could be paid, so the compensation principle is stated in terms of potential compensation. The Hicks-Kaldor criterion is the basis for cost-benefit analysis. Cost-benefit analysis, when properly conducted, will allow the analyst or policy maker to identify potential Pareto improvements and measure the magnitude of the difference between economic gains and losses.

Moreover, efficiency in a broader, economic or non-economic, sense could be defined in other ways. For instance, politicians may use goal fulfillment as a criteria of whether a specific policy is efficient. This goal may in turn be the result of public deliberations resulting in a different efficiency criterion than that provided by Pareto.

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[1] The principal economic criterion for choosing among various allocations occurring at the same point in time is called static efficiency. An allocation of resources that satisfies the static efficiency criterion is one where the net benefits are maximized by that allocation, while dynamic efficiency, on the other hand, do take timing into play and takes into account that society's objective is to balance the current and future uses of the resource by maximizing the present value of the net benefits derived from the use of the resource (Tietenberg, 1996).

For our purposes we make use of three different efficiency criteria, viz., (a) social optimality, i.e., the efficient allocation is determined by comparing social costs and benefits; (b) cost-efficiency (efficiency without optimality), i.e., a policy is cost effective if it achieves a given policy objective (e.g., waste paper recovery rate) at the lowest possible cost; and (c) efficiency in the sense of affecting behavior. For instance, a subsidy on recycling efforts is efficient if it induces a comparatively profound increase in recycling activities.[2]

Social optimality

If social optimality is to be ensured, all costs (both internal and external) need to be monetized and juxtaposed against other input requirements for the alternative waste treatment routes and for the primary output. Figure 1 shows a graphical analysis of such a procedure where the marginal cost (MC) consists of sorting, cleaning and transporting the respective fractions of the waste flow to collection centers, costs for administration of keeping the recycling policies in force, etc.

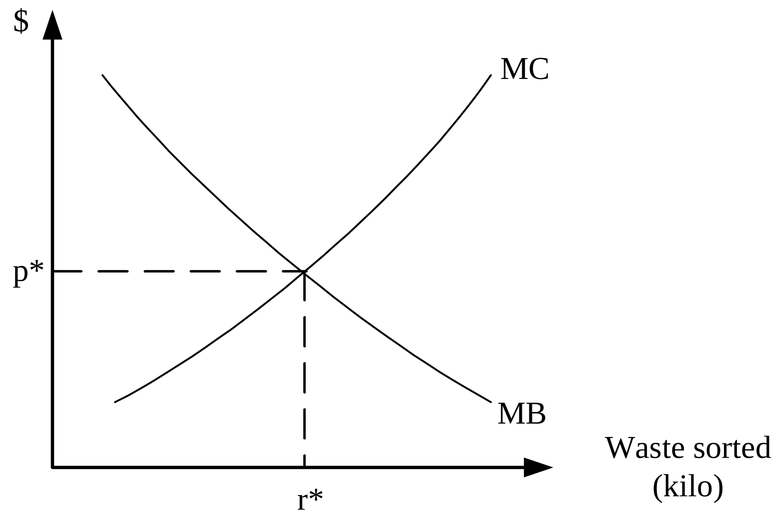

Figure 1: The Socially Optimal Rate of Recycling Activities

[2] We will not consider productive efficiency (or technical efficiency), i.e., when firms using the least amount of resources to produce a given good or service or when output is being produced at the lowest possible unit cost. The center of our attention is instead on allocative efficiency, i.e., a condition achieved when resources are allocated in a way that satisfies any of our employed efficiency criteria.

The marginal benefits (MB) refer to the value of recycled materials in new production, avoided external costs such as reduced environmental impact and reduced depletion of primary resources caused by the use of secondary materials, etc. A recycling policy with a goal of maximizing social welfare must aim at maximizing the net benefits to society, i.e., bring equality between the marginal cost and marginal benefit of recycling. The intersection of the MC and MB curves will also give us the price per kilo waste (p*) disposed that ensures a socially optimal rate of recycling activity (r*). Hence, r* would be a Pareto-optimal provision of waste sorted. Consequently, if the controlling authority knows the marginal costs and marginal benefits they can identify the tax (p*) that will yield an economically efficient level of waste sorted.

Cost-efficiency (efficiency without optimality)

When the requisite for the valuation of costs and benefits is neither available nor sufficiently reliable economic analysis often focuses on finding the least-cost means of achieving an exogenously given environmental target (e.g., recovery rate). This is sometimes referred to as efficiency without optimality (Baumol and Oates, 1988). Cost-efficiency is realized when the marginal costs of all possible means of achievement are equal.

Marginal cost curves can thus help us determine the cost-efficient division of the waste flows between alternative countries, regions or households in the following way. Let us assume that waste recycling in two regions, A and B, involves marginal costs as depicted in Figure 2, and that we wish to divide r* tons (the recycling target) of all waste flows between the two regions so as to minimize the cost to society as a whole. It is important to note that this “goal” of r* tons does not necessarily represent the socially optimal level of recycling activities. Still, if we assume that this is the level of waste to be recycled, cost-effectiveness analysis can be employed. In the analysis region B is assumed to have a low cost of recycling waste, while region A in comparison has a relatively high cost of recycling.

Figure 2 demonstrates that a* tons should be recycled in region A and b* tons should be recycled in region B, this since the marginal costs of recycling between the regions are the same at this level, and the two sum up to the recycling goal (r*). Any other subdivision of the recycling target will carry higher total costs, and will thus be sub-optimal. Hence, a price per kilo waste disposed equivalent to p* ensures cost-efficiency. In addition, by identifying the least-cost means of meeting a particular standard or policy goal and using this cost as a benchmark case, we can estimate how much total costs can be expected to increase from this minimum level if policies that are not cost effective are implemented.

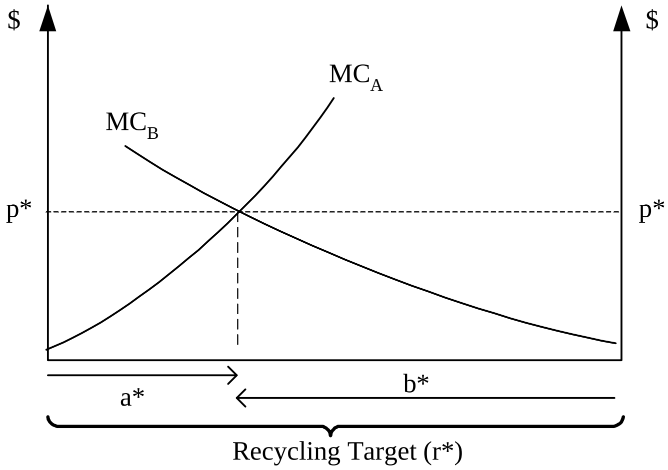

Figure 2: Cost-effectiveness Analysis in a Two Region Waste Recycling Scheme

Efficient in the sense of affecting behavior

A policy could also be considered efficient if it simply changes behavior in an intended way. This definition of efficiency – although it does not follow directly from the economics literature – is often important in practice. Consider two different marginal costs curves for recycling activities as depicted in Figure 3, where p* is the price per kilo waste not sorted the individual has to pay. In scenario A (the left graph) an individual will recycle as long as the financial benefit (i.e., the costs avoided) to recycle is larger than the cost of doing so. By the same token, in scenario B (the right graph), the individual will do the same. However, in scenario B the individual experiences lower costs for recycling due to, e.g., excessive space for sorting waste, shorter distance to recycling center, etc. Economic incentives such as a unit pricing system would accomplish different outcomes depending on whether scenario A or B is present.

Given scenario A, a unit-pricing system would only accomplish a moderate change in the individuals' behavior since the price elasticity of recycled material supply is low. Conversely, under scenario B a unit-pricing system would work well since the price elasticity of recycled material supply is high. Thus, if people do not respond very much to price incentives for waste collection, economic incentives will not be an efficient tool for inducing changes in household behavior. Instead other policy measures (e.g., information campaigns) may be more efficient, i.e., successful in changing recycling behavior.

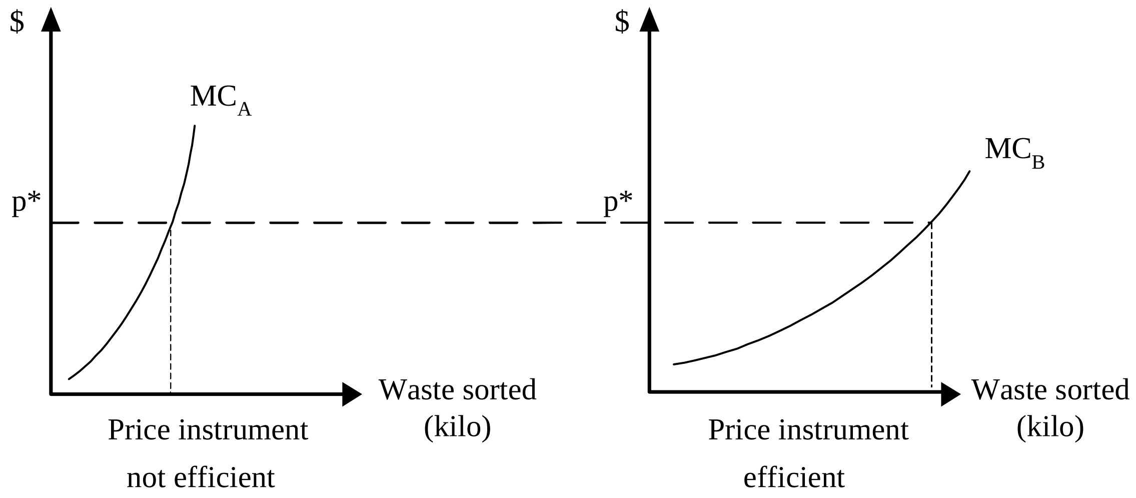

Figure 3: Two Different Elasticities of Supply for Recycling Activities

The analytical models developed in this thesis allow us to think more systematically about how to design efficient policy instruments in the recycling field. The thesis makes use of all the three different efficiency criteria presented above. Before outlining the specific contents of all the attached papers, however, a survey of the relevant literature is provided. We also discuss our overall contributions to this literature.

3. Previous Research on Solid Waste Recycling, and Contributions to the Literature

In this section we present in what way previous research efforts have employed the above criteria of efficiency in recycling policy and behavior, and in what way the papers in this thesis complement and add to this research field.

Social optimality

Social optimality has, naturally, been investigated both theoretically and empirically. For an excellent and broad review on the economics of various solid waste management policies, see Kinnaman and Fullerton (2000). One set of theoretical studies analyzing social optimality derives Pigouvian taxes on waste to ensure Pareto optimum (e.g., Dinan, 1993; Fullerton and Kinnaman, 1995; Palmer and Walls, 1997; Fullerton and Wu, 1998). Another set of studies use dynamic models to analyze waste management options in general, e.g., derives optimal mixes of recycling, incineration and landfilling (e.g., Leach et al., 1997; Huhtala, 1997; Keeler and Renkow, 1994; Powell, 1996; Powell et al., 1996; Leu and Lin, 1998; and

Samakovlis, 2001). For instance, Samakovlis (2001) employs a dynamic general equilibrium growth model and find that before a waste management hierarchy can be established, it is important to focus on total allocative efficiency rather then just on material recycling *per se*.

Previous empirical research focusing on socially optimal levels of recycling policies, e.g., Byström and Lönnstedt (1997); Bruvoll (1998); Radetzki (2000); and Samakovlis (2001), in particular analyzes whether collected waste paper should be incinerated, landfilled or recycled in paper and board mills. Finding socially optimal levels of recycling policies are, of course, not easy. However, according to Pearce and Helm (2003) one study that produces a correct methodology that does acknowledge all social costs of waste treatment is the one by Radetzki (2000) who analyzes the producer responsibility ordinance (PRO) in Sweden. Radetzki criticizes the Swedish producer responsibility legislation related to packaging and paper. The study compares the costs and benefits of the PRO with the net benefits of incineration and landfilling the waste that is recycled under the ordinance. For each waste handling option the internal costs of handling firms are estimated. To these external costs estimates of the value of time undertaken by the waste generators who sort waste at source and transport it to recycling centers. Then other external environmental costs from each option are added, while any environmental taxes are deducted (all items are considered in cost terms) since the taxes internalize some of the externalities. Finally, avoided environmental primary production damages are deducted and environmental taxes on primary production are added back in. Radetzki argues that when only the environmental benefits derived from the legislation are considered, the outcome is benign. In a broader context, however, he concludes that the legislation is exceedingly inefficient. The overall social costs at the margin are shown to be 5-20 times higher than the benefits. He finds that the PRO imposes substantial social costs to society compared to incineration and landfilling. The values placed on the time households spend on sorting, cleaning and transporting waste constitute a substantial share of the total cost of recycling.

Similarly, Bruvoll (1998) examines waste disposal options in Norway. She find that the social costs of recycling are higher than the social costs from both landfill and incineration for several waste fractions. Bruvoll also includes source reduction implemented by a tax on material inputs as a waste treatment method under scrutiny. Her study supports the ranking of “reduce” as the superior alternative in the waste hierarchy. In addition, the largest cost component for recycling was found to be the households’ time for sorting the waste, reducing the social cost-effectiveness of this waste disposal mode. Bruvoll employs the same valuation of time for recycling activities. The debatable item in the above analyses is thus the inclusion of the waste generators' costs of sorting, cleaning and transport, since these studies base the valuation of households' costs on labor costs after tax (Pearce and Helm, 2003). If households undertake recycling activities solely on voluntary basis and/or for moral reasons, the opportunity cost is expected to be lower than the value they attach to time spent sorting and transporting waste. Furthermore, Goddard (1995) reviews earlier published and unpublished research, and finds that the conceptual and empirical basis on which to predicate efficient and effective solid waste management policy is rather incomplete, but that one area where public interventions thus far established in the economic literature is the one for household user fees. However, Goddard adds that there is little focus on the role that such fees can play in motivation for source reduction at the household level.

The research presented in this thesis differs from these earlier studies in the following way. In contrast to earlier studies we analyze in more detail to what extent different recycling motives matter for: (a) the assessment of households' waste sorting costs; and (b) the efficiency of introducing economic incentives for stimulating households' recycling efforts.

Cost-efficiency (efficiency without optimality)

There are several aspects of cost efficient allocations, i.e., different activities that should be allocated in a cost effective manner. One aspect focuses on household recycling efforts. In particular, we want to ensure that households with low costs of time use for these activities recycle more than those with high costs for the same activities. In this case the size of the fee/tax is not the issue under scrutiny but rather the choice between incentive-based instruments and command-and-control instruments. There exist few empirical studies on any of the issues above. However, one important exception is the study by Palmer et al. (1997), in which a partial equilibrium model of waste generation and recycling is used to evaluate the cost-effectiveness of three various policies aimed at reducing solid waste disposal. By using estimates of supply and demand from previous literature and prices and quantities from 1990, the authors conclude that a deposit/refund system is the cost-effective policy under scrutiny (which is supported by the literature such as e.g., Dinan, 1993; Sigman, 1995; Fullerton and Kinnaman, 1996; and Calcott and Walls, 2000).

Another aspect of cost efficient allocation occurs on a regional and global scale, where countries/regions need to minimize recovery costs given a politically set recycling goal. At this junction the focus is on e.g., minimizing costs of collection and input costs for firms, as well as transportation issues. Some attention to this issue has been made by the employment of life cycle analysis (e.g., Finnveden and Ekvall, 1998; Finnveden et al., 2000). In general these studies focus on the total environmental impact of a product through every step of its lifetime, sometimes referred to as measuring the impact of a product "from cradle to grave." Life cycle analysis thus enables all the impacts of a process or product on the environment to be evaluated. However, most life cycle analyses either ignore transport altogether or, when comparing different scenarios, assume that there will be little changes in transport modes or distances traveled. Although this notion might be valid in certain situations, transportation costs may turn out to be very important when analyzing the cost *efficiency* of the waste paper collection process.

Dinan (1992) show that one can accomplish this end of cost efficiency by introducing a tradable permits system. In theory, the cost effective solution could be obtained by setting an overall recovery rate for old newsprint and allow newsprint producers and importers to buy permits from the suppliers of old newsprint. If the recovery rate of 50 percent was set, then the producers of newsprint would have to obtain 0.5 permits per ton produced, and each ton of old newsprint produced would generate one permit. Newsprint producers who (for technical or geographical reasons) face difficulties in consuming large quantities of old newsprint would be inclined to buy this permit. In this way, the production of old newsprint is "subsidized" by the sale of permits, and the production of newsprint is "taxed" by the requirement to purchase permits. Such a policy would, under certain conditions, yield efficiency gains to society. The first and second paper in this thesis deals with differences in waste paper recovery and utilization rates, providing an econometric analysis of the most important determinants of inter-country differences in waste paper recovery and utilization rates. These papers differ from earlier research efforts in a number of ways. Very few studies recognize explicitly that different countries/regions possess varying economic prerequisites in terms of waste paper recovery and utilization potential. Moreover, in contrast to the earlier research, we analyze the long-run factors that drive recovery and utilization rates.[3] This should be of particular interest as these rates in themselves represent important policy targets. Such an understanding is important for policy makers, and it also facilitates aggregate projections of waste paper usage and recovery worldwide. By concentrating on inter-country differences we also broaden the focus of waste paper studies, which in the past have been heavily biased towards the U.S. and Western European markets.

[3] Van Beukering and Bouman (2001) empirically analyze cross-country variations in waste paper recovery and utilization. However, their study focuses mainly on the impact of international trade and it fails to distinguish between short- and long-run impacts.

The research presented in the third paper in this thesis focuses on regions within a country, and differs from these former studies in the subsequent ways. First, it has an explicit consideration of transportation issues. Second, it presents and applies a methodological framework in which cost-effective recovery rates for different spatially distributed regions can be calculated. Third, the model employed accounts explicitly for the fact that different counties/regions possess different economic prerequisites in terms of waste paper recovery and utilization potential.

Efficient in the sense of affecting behavior

Regarding whether (primarily price-based) policies are efficient in affecting behavior, there exists a substantial body of previous research efforts on waste paper economics.[4] Earlier studies have focused a lot of attention on *short-run* supply and demand behavior in waste paper markets (e.g., Anderson, 1977; Blaut and Steiker, 1978; Edwards, 1979; Gill and Lahiri, 1980; Edgren and Moreland, 1989; Nestor, 1992; Rehn, 1995; Samakovlis, 2001). In general these studies conclude that in the short-run both the supply and the demand of waste paper are relatively own-price inelastic. Furthermore, as demand for paper and board products is relatively sensitive to the prevailing level of economic activity, this price insensitivity makes the waste paper market very volatile in terms of price fluctuations. These earlier studies have been motivated largely by the argument that policy makers need a more detailed understanding of waste paper supply and demand in order to be able to implement recycling policies that will affect production and consumption levels.

If people do not significantly respond to price incentives for waste collection, economic incentives will not be an efficient tool for inducing changed household behavior. Solid waste is one of the most obvious areas where economic incentives have entered the arena in reforming environmental policy. A flat fee system, the most common existing strategy, is unrelated to the volume of waste that is discarded. As a result, an observable trend in municipal solid waste (MSW) policy reform is a gradual increase in the use of economic incentives, which is motivated due to its theoretical encouragement of both source reduction as well as waste diversion activity. However, empirical evidence of realized efficiency gains is not unambiguous. For instance, in the absence of illegal dumping opportunities unit-based

[4] Regarding the economics of household recycling behavior, seminal work by two influential economists, Don Fullerton and Thomas Kinnaman, are composed in a recent book (Fullerton and Kinnaman, 2002), pointing out how vigilant use of economic analysis can contribute in illuminating environmental problems.

pricing is the most efficient way of reducing waste (Jenkins, 1993), while Fullerton and Kinnaman (1995) show that in presence of illegal dumping the optimal charge may be zero or even negative.

Callan and Thomas (1999) use data representing 317 communities in Massachusetts, and find that the likelihood that a community will implement a unit-pricing program increases with household income, housing value, age, and if regional landfill is due to close within two years. Kinnaman and Fullerton (1997) employ cross-sectioned data for over 800 U.S. communities, and find that the likelihood increases with local tipping fee, with the use of municipal resources to collect garbage, and with educational level. For discussions of the merits of disposal charges such as unit-based pricing, see e.g., Repetto, et al. (1992), Jenkins (1993), and Miranda et al. (1994). Nestor and Podolsky (1998) use self-reported household data to analyze the effectiveness of a subscription program versus a bag/tag program on garbage and recycling quantities. Neither program is found to encourage source reduction in the presence of curbside recycling scheme, as both programs would subsidize recycling households' overall disposal practices.

Another strand of research points out that motives do matter and that people undertake recycling activities for moral reasons (e.g., Ackerman, 1997; Hornik, et al., 1995). Hornik et al. (1995) employ a meta-analysis of 67 empirical studies (both published as well as unpublished) and find that consumer knowledge and commitment to recycling best predicts propensity to recycle. These reasons may lead to unresponsiveness among households towards price incentives for waste collection, and under these circumstances a unit-pricing system would not work. Ackerman (1997, p. 32) support this notion and concludes: "unit pricing is remarkable not for how much it accomplishes, but for how little," and he goes on to stress the role of non-market motives in household recycling. Thus, if people do not respond very much to price incentives for waste collection, economic incentives, such as e.g. a unit-pricing system, would only accomplish a moderate change in individuals' behavior.

Furthermore, the idea that monetary rewards may crowd out intrinsic motivation has been acknowledged in the economic-psychology literature. Previous studies that take this matter in hand, analyzing the crowding out effect, have, however, mainly been focusing on labor supply performance, common pool resources, services, and constitutions and tax evasions (for a review of these studies, see Frey and Jegen, 2001), and not on environmental issues although this field would benefit from an incorporation of crowding theories (Frey, 1999). Crowding effects are potentially relevant in studying individual behavior in the economy in many different areas. One example is within the area of municipal solid waste management, where the effect of pricing instruments such as pay-by-the-bag for household waste is in question. For these types of schemes it must be considered whether monetary incentives crowd out the notion of commitment for one's own environment. Standard economics largely relies on the skillful application of the price effect. However, people might hold lexicographic preferences, where substitutability of environmental quality with other goods is discarded (Spash, 2000). Notably, if households take a strong positive moral stance to waste sorting, the use of economic incentives may be an ineffective policy tool as it may undermine an individual's sense of civic duty (Frey and Oberholzer-Gee, 1997). In order to implement successful policies we need to know how people judge the input of resources and perceive constraints such as time, money and their motives when recycling household waste. The fourth paper in this thesis analyzes household attitudes to recycling activities in a municipality in northeast Sweden, Piteå. Specifically, we test the crowding out hypothesis in the empirical context of household recycling and price-by-the-bag systems.

4. Summaries of the Papers

Paper [1] An Econometric Analysis of Global Waste Paper Recovery and Utilization (with Patrik Söderholm) (Re-submitted to *Environmental and Resource Economics*)

During the last decades policies aimed at encouraging waste paper recycling have become increasingly popular. Concerns about problems related to forest conservation and waste disposal are generally the prime motivations for these policy efforts. As a part of attempts to measure "recycling success" countries worldwide tend to formulate recycling goals in terms of *recovery* and *utilization* rates for paper and board products (e.g., Buclet and Godard, 2000; Van Beukering and Sharma, 1996). However, if these recycling rates are to serve as major policy variables, policy makers need to understand what determines the size of these rates, and what the consequences are of pursuing different recycling goals.

The main purpose of this paper is to provide an econometric analysis of the most important determinants of inter-country differences in waste paper recovery and utilization rates. By employing pooled time series and cross-section data over 49 countries worldwide and seven years, the paper concludes that relative waste paper recovery and use depend largely on long-standing economic factors such as population intensity and competitiveness in the world market for paper and board products. We also find evidence that supports the conjecture that rich countries tend to recover relatively more waste paper than is the case in low-income countries, reflecting the higher demand for waste management and environmental policies in more developed economies. As recovery and utilization rates are determined by economic and demographic characteristics the degree of policy flexibility in affecting these rates may be limited. In particular, an ambitious utilization rate target may be very costly to enforce as it can conflict with existing trade patterns of paper and board products as well as with other environmental goals. Additional policy targets may therefore be desirable, especially since paper recycling is motivated primarily by environmental concerns and seldom is a benign activity in itself.

Paper [2] Complementing Empirical Evidence on Global Recycling and Trade of Waste Paper (with Patrik Söderholm) (Forthcoming in *World Development)*

This paper builds and extends upon paper [1] and provides a critical analysis of Van Beukering and Bouman's (2001) article in *World Development* on global paper recycling and trade. The main objective of their analysis is to understand what are the main determinants of recovery and utilization rates of lead and waste paper. Their main conclusion is that trade (of recyclable products) can be beneficial to the utilizers of secondary materials (developing countries), and detrimental to the recoverers of recyclable materials (developed countries). They also conclude that a set of geographic variables (e.g., population density, primary commodity endowment) have noteworthy influences on recycling behavior, while the impacts of a number of "market-related" variables (e.g., relative price of waste paper and wood pulp) are referred to as "ambiguous".

We first question their notion that developing countries specialize in waste paper utilization and developed countries in recovery activities primarily because of different patterns of waste paper trade. An increased focus on relative waste paper availability, we argue, provides us with a better understanding of global paper recycling. We also criticize some of the implicit assumptions made in their regression analysis of waste paper utilization rates. An alternative regression model is therefore derived and estimated. In contrast to the approach used by Van Beukering and Bouman our analysis (a) is consistent with basic microeconomic theory; (b) distinguishes clearly between short- and long-run impacts; and (c) produces results that support our initial conjecture that waste paper availability is the most important determinant of waste paper use.

Paper [3] Spatial Cost Efficiency in Waste Paper Handling: The Case of Corrugated Board in Sweden (Submitted to *Environmental and Resource Economics*)

Since 1994/95 the producer responsibility ordinance (PRO) has governed the collection and recycling of waste paper in Sweden. This responsibility is regulated for newspaper and similar paper qualities by the ordinance SFS 1994:1205, and for paper packaging materials by SFS 1997:185. The PRO *de jure* requires separation of paper from other waste products and that collected waste paper must proceed to material recycling, i.e., waste paper should be used in paper and board production and not for energy recovery or landfill (Swedish Environmental Protection Agency, 1999). However, *de facto* Swedish law also requires that waste paper should be collected throughout the country with no special account taken of regional differences such as transport network, distances to paper mills, and other demographic and geographic conditions. In other words, the law does not permit differentiated recovery targets across regions.

This paper analyzes the spatial cost efficiency of the Swedish legislation regarding waste paper handling. The focus is on the case of corrugated board, and we recognize that the different counties in Sweden possess different economic prerequisites in terms of waste paper recovery and utilization potential. We employ data for six corrugated board mills and 20 counties, and a non-linear programming model to identify the least cost strategy for reaching the politically specified recycling target of a 65 percent recovery rate for corrugated board. That is, the total costs of recovering a minimum of 65 percent in each and every county are calculated and compared with the case when 65 percent of all old corrugated board is collected nationwide but there exist no uniform target for the counties.

The conclusion is that from a cost efficiency point of view the recovery efforts should be concentrated to the highly populated and urbanized counties, and not be uniformly divided throughout the country. In the base case the results suggest that the cost efficient county-specific recovery rates should range from 51 percent to 72 percent.

Paper [4] Households' Perceptions of Recycling Efforts: The Role of Personal Motives

Local and national authorities use both economic and political instruments in their attempts to induce households to contribute to sustainable development. However, there is a lack of understanding of how these tools interplay with the motives held by households and the daily constraints they face.

The purposes of the paper are to analyze whether moral motives matter for: (a) the assessment of households' waste sorting costs; and (b) for the efficiency of introducing economic incentives for stimulating households' recycling efforts. We do this by gathering data using a mail-out survey to 850 randomly chosen individuals in the municipality of Piteå, Sweden. We employ an economic model of moral motivation with possible motivation crowding-out and econometric techniques.

Two important conclusions follow from the analysis. First, the results support the notion that moral motives significantly lower the costs associated with household recycling efforts. Specifically, the average hourly willingness to pay to let others sort household waste at source was found to be significantly lower than the corresponding income after tax (i.e., the opportunity cost of time). Second, moral motives are in some cases the cause of inefficient policy outcomes when introducing economic incentives to promote recycling efforts. Those who initially feel intrinsically motivated to sort waste at source are more inclined to be discouraged by the introduction of a pay-by-the-bag waste management scheme. However, those who sort waste because it is a requirement imposed on them by the authorities, have a positive perception of such as system since it permits a larger degree of flexibility in recycling efforts.

The above give rise to an important policy implication; motives do matter for recycling activities in general. The population is heterogeneous in their view of recycling efforts, i.e., there is no single *Recycling Man* out there to be guided. Hence, means or policies aimed at changing behavior are not straightforward. Some individuals have learned to appreciate the reward of economic incentives, while some feel that environmental morale is crowded out by the same means. This insight is vital if the use of such instruments is to be expanded. The goal for policymakers will thus be to find a way of guiding both categories of people in their strive for a sustainable society.

5. Concluding Remarks and Implications

In this thesis we make use of three different efficiency criteria, viz., (a) social optimality; (b) cost-efficiency (efficiency without optimality); and (c) efficiency in the sense of affecting behavior. Regardless of which efficiency criteria is chosen, we need to understand the rationale and behavior of the measures and/or actors involved and the main determinants behind them. The three most important conclusions that can be drawn from this thesis are as follows. *First*, the degree of policy flexibility in affecting recovery and utilization rates appears to be limited. Additional policy targets may therefore be desirable, especially since paper recycling is motivated primarily by environmental concerns and seldom is a benign activity in itself. Policy efforts intended to affect these rates may therefore turn out to be very costly or difficult to enforce as they run counter to other environmental goals. *Second*, since regions are endowed with different sets of geographic and demographic features optimal waste paper recovery rates will differ between regions and thus counties. That is, each region exhibits unique features that need to be considered if efficiency in waste handling is to be attained. Thus, the recovery efforts should be concentrated to the highly populated and urbanized areas, and not, as in Sweden today, uniformly divided throughout the country. *Third*, motives do matter for recycling activities in general. As a consequence, for the assessment of households' recycling costs, moral concerns significantly lowers the cost associated to the time spent on recycling activities compared to the corresponding income after tax. Moreover, the population is heterogeneous in their perception of recycling efforts and waste management strategies, which imply that the implementation of policies aimed at changing recycling behavior is not straightforward. Of course, the papers provided here have not provided any complete answers to the questions raised, and this should open the field for future research in this important field.

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An Econometric Analysis of Global Waste Paper Recovery and Utilization

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Abstract
The main purpose of this paper is to provide an econometric analysis of the most important determinants of inter-country differences in waste paper recovery and utilization rates. By employing pooled time series and cross-section data over 49 countries worldwide and seven years, the paper concludes that relative waste paper recovery and use depend largely on long-standing economic factors such as population intensity and competitiveness in the world market for paper and board products. We also find evidence that supports the conjecture that rich countries tend to recover relatively more waste paper than is the case in low-income countries, reflecting the higher demand for waste management and environmental policies in more developed economies. As recovery and utilization rates are determined by economic and demographic characteristics the degree of policy flexibility in affecting these rates may be limited. In particular, an ambitious utilization rate target may be very costly to enforce as it can conflict with existing trade patterns of paper and board products as well as with other environmental goals. Additional policy targets may therefore be desirable, especially since paper recycling is motivated primarily by environmental concerns and seldom is a benign activity in itself.

Key words: waste paper, recovery rate, utilization rate, inter-country differences.

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1. Introduction

During the last decades policies aimed at encouraging waste paper recycling have become increasingly popular. Concerns about problems related to forest conservation and waste disposal are generally the prime motivations for these policy efforts. As a part of attempts to measure "recycling success" countries worldwide tend to formulate recycling goals in terms of *recovery* and *utilization* rates for paper and board products (e.g., Buclet and Godard, 2000; Van Beukering and Sharma, 1996). However, if these recycling rates are to serve as major policy variables, policy makers need to understand what determines the size of these rates, and what the consequences are of pursuing target values of different recycling goals. The purpose of this paper is to provide an econometric analysis of the determinants of inter-country differences in waste paper recovery and utilization rates.

The main hypothesis of the investigation is that recovery and utilization rates are largely economically driven, i.e., they are determined by long-run supply and demand factors in the waste paper market. Thus, in addition to policy differences, in terms of the number and the quality of waste management schemes, countries are endowed with different sets of demographic and geographical features (e.g., population density, virgin forest supplies etc.), which in turn affect the economics of paper recycling. OECD (1976, p. 29) provides an early support for these arguments and concludes:

> [T]here exists very significant differences between countries both in rates of recovery and utilisation. In part these differences stem from variations between countries in economic characteristics about which little can, or should, be done. In part, however, they reflect attitudes, institutions and policies which are amenable to change. Whilst future policies adopted by various countries for promoting waste paper recovery and utilisation will therefore exhibit certain common features, the precise measures employed will inevitably reflect the particular needs and characteristics of each country.

An analysis of what are the main determinants of recovery and utilization rates is thus important for understanding the degree of policy flexibility in affecting these rates. For example, if the availability of waste paper is an important determinant of relative waste paper utilization, a strict utilization rate goal may be difficult to enforce as it conflicts with prevailing paper and board trade patterns (Huhtala and Samakovlis, 1999). Also, if the recovery rate is largely determined by important cost elements (e.g., population density etc.), it can be costly to pursue very ambitious recovery targets and also to implement harmonized policy targets across countries.

Previous research efforts on waste paper economics have focused a lot of attention on *short-run* supply and demand behavior in waste paper markets (e.g., Anderson, 1977; Blaut and Steiker, 1978; Edwards, 1979; Gill and Lahiri, 1980; Deadman and Turner, 1981; Edgren and Moreland, 1989; Nestor, 1992). In general these studies conclude that in the short-run both the supply and the demand of waste paper are relatively own-price inelastic. Furthermore, as demand for paper and board products is relatively sensitive to the prevailing level of economic activity, this price insensitivity makes the waste paper market very volatile in terms of price fluctuations. These earlier studies have been motivated largely by the argument that policy makers need a more detailed understanding of waste paper supply and demand in order for implemented recycling policies to be able to affect production and consumption levels. The question of whether recycling policies are socially worthwhile is thus not the issue under scrutiny. A second set of studies does, however, take this latter matter in hand and focuses on the efficiency of recycling policies. In particular it analyzes whether collected waste paper should be incinerated, land-filled or recycled in paper and board mills (e.g., Byström and Lönnstedt, 1997; Bruvoll, 1998; Radetzki, 2000; Samakovlis, 2001).

The present paper differs from earlier research efforts in a number of ways. Very few studies recognize explicitly that different countries (and regions) possess varying economic prerequisites in terms of waste paper recovery and utilization potential. In contrast to the earlier market studies, the present paper analyzes the *long-run* factors that drive recovery and utilization rates.[1] This should be of particular interest as these rates in themselves represent important policy targets. Such an understanding is not only important for policy makers; it also facilitates aggregate projections of waste paper usage and recovery worldwide. By concentrating on inter-country differences we also broaden the focus of waste paper studies, which in the past have been heavily biased towards the U.S. and Western European markets. The econometric analysis makes use of a panel data set of 49 countries worldwide over seven years (1990-1996). A number of authors have suggested that the motives for paper recycling

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[1] Grace et al. (1978) first outlined the links between recovery and utilization rates for secondary materials. Van Beukering and Bouman (2001) empirically analyze variations in waste paper recovery and utilization. However, their study focuses mainly on the impact of international trade and it fails to distinguish between short- and long-run impacts. For a critical discussion of their analysis, see Berglund and Söderholm (2003).

differ between developed countries and less-developed countries (see section 2.4). For this reason we divide the data set into one "rich" country sample and one "middle-income" sample, and this allows us to analyze whether there exist important structural differences in waste paper recycling behavior between the two country groups.

The paper proceeds as follows. Section 2 begins with a definition of waste paper recovery and utilization rates, and discusses some potentially important determinants of these rates. Thus, in this section we hypothesize why these rates may (and perhaps should) differ between countries. In section 3 we present two econometric models, which are used to analyze the impact of a set of independent variables on the recovery and utilization rates respectively. Data and model estimation issues are discussed in section 4. The empirical results from the estimations are presented and discussed in section 5, while section 6 provides some concluding remarks and implications for policy.

2. Recovery and Utilization Rates: Definitions and Determinants

2.1 Basic Definitions

In general there is no mystery to the basis for waste paper recycling. *First*, paper is a major component of the overall municipal solid waste stream, and waste paper recycling is seen as one important waste management strategy. It reduces the need for disposal capacity, which in turn leads to lowered emissions from landfills. *Second*, recycled paper is often a relatively inexpensive input factor in the production of new paper and board products, partly because its energy requirements are low. Extended use of waste paper in paper production may also be motivated by the long-term value of forest conservation. Thus, the growing waste paper stock may displace some of the virgin raw material, i.e., wood pulp used in the production process. In other words, paper and board recycling is both supply- and demand-driven, and it is shaped by both economic and political factors. This is not always recognized in the recycling debate, which in recent years often has tended to focus on: (a) the municipal solid waste view of paper recycling; and (b) the pure policy aspects of the issue (Smith, 1997). Other important issues, such as the economic determinants of waste paper use (e.g., Nestor, 1992), are however often neglected.

The two most common measures of recycling progress, the recovery and utilization rates, differ in one very important respect (Grace et al., 1978). The recovery rate (*RR*) is basically grounded in the waste-management (or supply) view of recycling, as it measures the success with which one is able to recover used paper and board from the waste stream. It is normally expressed as:

$$RR = \frac{WP_{CONS} + WP_{EX} - WP_{IM}}{PB_{CONS}} \tag{1}$$

where WP_{CONS} denotes waste paper consumption, WP_{EX} is waste paper export, WP_{IM} waste paper imports, while PB_{CONS} is paper and board consumption (the final output). Thus, the numerator in equation (1), $(WP_{CONS} + WP_{EX} - WP_{IM})$, defines waste paper recovery. In contrast, the utilization rate (*UR*) reveals the extent to which the recovered paper is actually being used to process new useful paper and board products. The most commonly used measure of the utilization rate is defined as waste paper consumption in domestic paper and board production, WP_{CONS}, divided by paper and board production, PB_{PROD}, as shown in equation (2).

$$UR = \frac{WP_{CONS}}{PB_{PROD}} \tag{2}$$

In other words, the *UR* measure relates primarily to the demand side of the waste paper market.[2] It should be clear that the above recycling rates both have policy relevance. The recovery rate provides an indication of to what extent litter and improper disposal has been reduced. If, on the other hand, policy makers are interested in the forest protection or energy conservation advantages of recycling, they also need to know more about the utilization of waste paper.

In 1996 the average global waste paper recovery and utilization rates were both around 40 percent (Pulp & Paper International, 1999). Table 1 summarizes recovery and utilization rates in some selected countries worldwide. Substantial inter-country differences exist, especially with respect to utilization rates. These range from 6 percent in Finland to 100

[2] In defining both *UR* and *RR* we (implicitly) assume that stocks of waste paper remain constant over time. Moreover, we also assume that consumption and waste generation take place in the same time period. Still, for waste paper, given its relatively short lifetime due to material decay, both these assumptions appear fairly realistic (Van Beukering and Bouman, 2001).

percent in Hong Kong. The recovery rates vary less between countries but the differences are still substantial. In 1996 Algeria recovered 21 percent of its used paper and board products, while the corresponding figure for Germany was as high as 71 percent. It is also the case that high recovery rates do not necessarily imply high utilization rates. In Sweden, for example, the recovery rate of 52 percent is fairly high by international standards, but the utilization rate is "only" 17 percent. Since waste paper recovery and utilization rates have become important policy targets, it is important to understand the main determinants of these rates. A number of plausible determinants are discussed below, beginning with those that are likely to affect recovery rates. We then end this section by considering potential differences in paper recycling patterns between developed and less-developed countries.

Table 1: Waste Paper Utilization and Recovery Rates in Selected Countries (1996)

Country	*UR (%)*	*RR (%)*
Germany	60	71
Hong Kong	100	61
Sweden	17	52
USA	39	45
Canada	24	43
United Kingdom	69	40
Finland	6	34
Argentina	44	31
Algeria	71	21
Israel	78	24

Source: Pulp & Paper International (1998).

2.2 Differences in Recovery Rates

Figure 1 shows the historical patterns of waste paper recovery rates in Sweden and the USA. Together with Table 1, this figure provides an appropriate starting point for discussing the factors that drive these rates. Both Sweden and the USA have experienced a more or less consistent increase in recovery rates over the years.

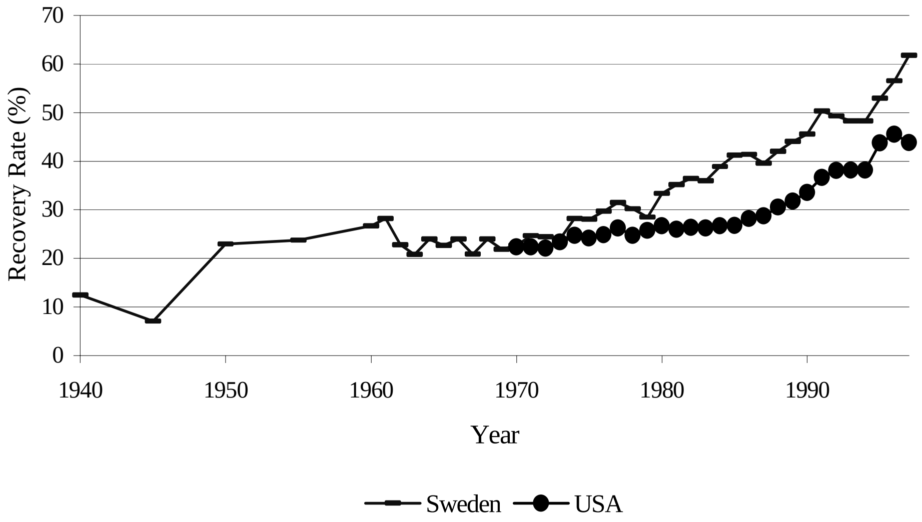

Figure 1: Waste Paper Recovery Rates in Sweden and the USA, 1940-1997

Sources: Swedish Forest Industries Federation (Skogsindustrierna), American Forest & Paper Association (2000), and Smith (1997).

In this paper we hypothesize that differential growth in per capita income is likely to explain parts of this pattern. It is often suggested that as incomes increase so does the value that people assign to the environment and hence to waste management activities. In economic terms, the income elasticities of environmental goods will tend to be greater than one, indicating that the environment is a so-called luxury good. This is consistent with the environmental Kuznets curve hypothesis, which predicts that for richer countries there will be a strong positive correlation between per capita income and the extent to which environmental protection measures, including waste management policies, are adopted.[3]

The increase in awareness about waste management is largely reflected in governmental laws and regulations. For example, recycling obligations of packaging material and municipal collection schemes clearly enhance the recovery of used materials. Moreover, many governments try to inform and educate consumers with the purpose of encouraging them to undertake recycling activities. Previous research results confirm that people tend to engage in recycling activities largely because they believe that recycling is necessary for achieving environmental goals, and they feel committed to these goals (e.g., Hornik et al., 1995).

[3] See for example World Bank (1992), Grossman and Krueger (1995), Kahn (1998) and Selden and Song (1994).

The above policies and commitments, we assume, are especially prevalent in relatively rich countries, and for this reason we would expect waste paper recovery rates to be positively correlated with per capita income. Table 1 also provides some sparse empirical support for this hypothesis. For example, Argentina and Algeria are both countries with relatively low per capita income levels, and their waste paper recovery rates are low as well. Similarly, environmental preferences in favor of increased recycling, whether expressed by policies or by volunteerism, are likely to be important determinants of the increase in recovery rates in Sweden and the USA since the early 1970s.

However, it should be noted that there also exist empirical studies which challenges the view that the environment is a luxury good, and that instead present results that are consistent with fairly low income elasticities of environmental demand (e.g., Arrow et al., 1995; Kriström and Riera, 1996). Still, we believe that our maintained hypothesis of a strong correlation between income and the demand for waste management is a plausible one. *First*, the studies that question this notion on empirical grounds primarily rely on inferences drawn from variations between income groups within a given country, and our study employs data across different countries. In other words, while the former studies put forward (and sometimes reject) the hypothesis that as households in a given country move from lower-income brackets to higher ones the impact on the demand for environmental improvements will increase significantly, we instead argue that as economies *as a whole* grow richer the same increase in environmental demand (and hence in waste management practices) will occur. We believe that it is easier to find empirical support for this latter hypothesis than for the former. *Second*, regardless of the level of aggregation employed there exist other empirical evidence that supports the existence of a close positive correlation between waste paper recovery activities and income.[4]

If one accepts our notion that waste paper recovery is affected largely by economic factors, we also have to consider the costs of collection and recovery. One important cost component in waste paper collection, especially in the past, has been labor. "Although a substantial investment in physical capital is required to produce recycled paper, the sorting, collecting, packing, and even transportation involved in the recycling process tend to be less capital- and more labor-intensive than production methods utilizing non-recycled materials

[4] See, for example, Van Beukering and Bouman (2001) who rely on a panel of country data, and Jenkins et al. (1999) who investigate newspaper recovery by household in the USA.

inputs," (Wiseman, 1990, p. 42).[5] Furthermore, as per capita income grows so does the real wage, and hence the opportunity cost of labor-intensive activities. In other words, with growing per capita income levels, time-consuming recovery and recycling activities become less attractive. Hence, when considering both the demand for waste reduction and the cost of waste paper recovery, it becomes clear that the net effect of per capita income changes on the recovery rate is ambiguous. There exist both an income effect and a cost effect and which of these dominates remains an empirical question.

Since waste paper recovery ultimately stems from recent paper and board consumption, the cost of waste paper recovery will also be affected by demographic circumstances. Other things equal, actions to separate and collect waste paper will be more viable in regions that are densely populated and/or in which people live clustered in highly urbanized areas. Moreover, high population densities are likely to drive up land prices, and in this way increase the costs for landfilling and hence disposal. Thus, since the marginal cost of recovery will depend on the size of the waste stream, recovery rates are expected to be particularly high in small but densely populated regions. Hong Kong, with a recovery rate of 61 percent, probably provides an illustrating example of the latter (Table 1).

It should be noted, of course, that these collection costs are not purely market-driven; they are also affected by government policies. In the USA, for instance, the alternative cost of landfill disposal ("tipping fees") rose dramatically during the end of the 1980s, in part as a way of encouraging recycling activities and in part because of stricter regulation of landfill systems to control leachate etc. So, in effect, the cost of waste paper collection in the USA, and indeed throughout most developed countries, were being driven at least in part by these contemporaneous policy developments.

Finally, waste paper prices will also influence recovery rates since they are an important component of the revenues stemming from waste paper recovery activities. Specifically, we would expect that the higher are national waste paper prices, the larger is the proportion of the waste stream that tends to be recovered.

2.3 Differences in Utilization Rates

Figure 2 shows waste paper utilization rates for Sweden and the USA since 1940. Both countries have experienced significant increases in utilization rates since the 1970s. However,

[5] See also Di Vita (1997).

before this increase we have witnessed an almost consistent decline, at least in the USA. The above suggests that long-term forces have been at work.

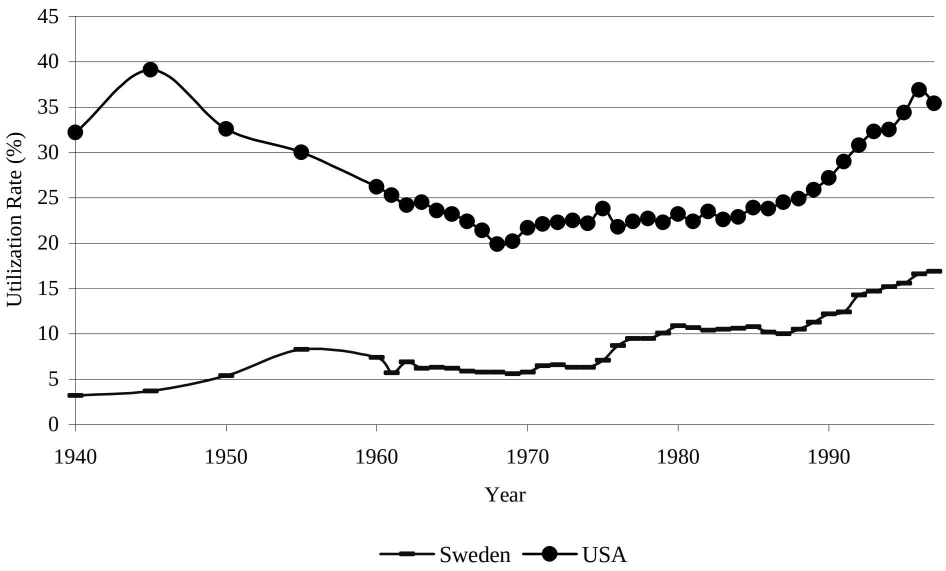

Figure 2: Waste Paper Utilization Rates in Sweden and the USA, 1940-1997
Sources: Swedish Forest Industries Federation (Skogsindustrierna), American Forest & Paper Association (2000), and Smith (1997).

Since the utilization rate measures to what extent waste paper is actually used in producing new paper and board products, we have to consider both the input and the output side of the paper and board industry in order to understand what drives this rate.[6]

The decision to use waste paper as input for the production of paper is primarily determined by the rate of return of different production methods. Thus, under traditional profit maximizing conditions waste paper consumption will depend on the price of waste paper and the price of its substitutes, which in most cases is wood pulp. Hence, the availability of waste paper and wood pulp (or virgin fiber) are likely to be important determinants of waste paper utilization rates. This implies that countries with relatively large forestry resources, utilized by

[6] There exist technical limitations with respect to how much waste paper can be used in the production process. A common assumption is that a fiber can circulate 5-6 times before it is exhausted, i.e., when the quality of the fiber no longer is able to meet the demands of the final product due to lack of strength and brightness (Antti, 2000). However, these technical constraints are unlikely to be important determinants of inter-country differences in utilization rates, and will therefore not be dealt with in more detail in this paper.

modern pulp mills, will have access to inexpensive virgin fiber relative to secondary fiber. Under such circumstances waste paper is obviously less economically attractive as a raw material, and the utilization rate will be comparatively low (e.g., Ince, 1995).

The potential availability of waste paper supplies, on the other hand, is largely reflected in the extent of a country's consumption of paper and board products. For example, in countries where consumption is significantly greater than production (implying net imports of paper and board) waste paper will be relatively abundant in relation to the derived demand for waste paper in paper and board production, and we would expect the utilization rate to be high. This is sometimes known as the *structural effect* or the *trade effect*, as it ultimately indicates the way in which comparative advantage of the domestic paper and board industry affects waste paper use (OECD, 1976). Specifically, while the net trade of paper and board products expresses a country's comparative advantage in the international market, it is also in fact a proxy for the relative availability (and hence the long-run price) of waste paper.

Of course, in a functioning market the above impacts would be reflected in the relative price of waste paper over wood pulp. However, price data on a large sample of countries (and, in particular developing countries) are very hard to obtain. In addition, waste paper prices tend to fluctuate much from one year to another due to temporary shortages and surpluses (e.g., Edgren and Moreland, 1989), making it hard to draw correct inferences based on cross-country variations (Berglund and Söderholm, 2003). In other words, short-run market-determined prices, given their high volatility, are not the major decision variable for paper and board companies when deciding to invest in waste paper using capacity. For these reasons, proxies of the long-run relative availability of waste paper and wood pulp are needed.

Furthermore, the intensity with which this waste paper potential is exploited is also an important determinant of utilization rates. Thus, the higher the actual recovery rate, the greater the likelihood for significant use of secondary fiber. By hypothesizing that changes in the recovery rates are important determinants of utilization rates we implicitly assume that environmental preferences mainly have an *indirect* influence on utilization through their impact on income and ultimately on recovery and collection activities. Of course, there may also exist a *direct* income effect on utilization rates. Based on environmental motives consumers may "convince" producers to increase their share of waste paper in the fiber mix, and producers are keen to pursue this as it opens up new markets and/or increases competitiveness. However, in our preliminary econometric analyses we found no support for this hypothesis, and in this paper the income variable is not included in the utilization rate analysis.[7]

The relative abundance of waste paper is likely to explain some of the differences in utilization rates displayed in Figure 2 and Table 1. Sweden possesses large inventories of forests and is a major exporter of paper and board products. As a consequence its utilization rate is still comparatively low, even though it has increased substantially since 1940. Finland, also a country blessed with large volumes of virgin forests, has a very low utilization rate compared to the world average (Table 1). Major net importers of paper and board, such as Hong Kong and Israel, instead display much higher utilization rates. Figure 2 also indicates some interesting developments over time. We note in particular that both in Sweden and the USA, the 1960s witnessed stagnant and even decreasing utilization rates. Parts of these declines are probably explained by the changing relative economics of virgin versus secondary fiber during this period. Specifically, advances in pulp and paper technology and the introduction of semi-chemical pulp led to a lower relative cost of primary fiber (Wiseman, 1990; Smith, 1997). Nevertheless, in order to gain a more complete picture of the determinants of waste paper utilization, we need to consider the output side of the paper and board industry as well.

Differences in waste paper utilization rates may reflect variations in the paper and board product mix. Due to differences in quality demands, some products simply have higher waste paper coefficients than others. The highest utilization rates are generally found for newsprint, tissue and some paperboard qualities, while the lowest rates normally are in printing and writing papers. For example, in Sweden roughly 85 percent of paper industry demand for secondary fiber comes from newsprint and liner and fluting production (Swedish Forest Industries Federation, 2000). Thus, countries in which overall paper and board production is dominated by waste paper intensive products, will, ceteris paribus, experience a higher utilization rate than otherwise. This factor can have contributed to the depressed utilization rates in the USA during the 1960-1990 period. During this period the printing and writing paper share of paper supply increased from 46 percent in 1960 to almost 60 percent in 1990 (American Paper Institute, 1983, 1991).

[7] See also Berglund et al. (2002) for additional empirical evidence in support of this notion.

2.4 The Existence of Structural Differences between Poorer and Richer Countries

A number of authors suggest that there exist fundamental structural differences between waste paper recycling patterns between richer and poorer countries, especially in the case of recovery activities (e.g., Van Beukering and Bouman, 2001). In the developed world, it is argued, increases in waste paper recovery have been driven largely by higher (often policy-induced) disposal costs and concerns about waste management and environmental and health issues. Recovery activities in the developing world, however, have been determined mainly by purely economic motives, i.e., they occur because recovery is cheap and since there is a demand for the collected waste paper (Cointreau, 1987). Thus, in addition to our hypothesis that long-standing economic factors are important determinants of global waste paper recovery, we also test the notion that these determinants have a stronger effect on recovery behavior in countries with low per capita income levels compared to richer countries.

For example, we have pointed out that the impact of changes in per capita income levels involves both an income effect and a cost effect. The income effect, we assume, will reflect the existence of environmental preferences in favor of ambitious waste management policies, while the cost effect expresses the opportunity cost of labor. If our hypothesis about the differences between richer and poorer countries is valid, we would expect the income effect to be relatively high in rich countries, while the cost effect would be more important in poorer countries. In addition, if recovery behavior were more market-driven in less-rich economies, the recovery sectors in these countries would also be more sensitive to variations in population densities and urbanization rates. In a similar fashion, it may be the case that the impacts of increases in waste paper prices also differ across country groups. The more policy-driven is the waste paper recovery rate – as is plausibly the case in relatively rich countries – the less own-price sensitive one could expect it to be. On the other hand, poorer countries may have a less-developed infrastructure (i.e., roads, railways etc.), and therefore experience problems in supplying more recovered paper in response to higher prices.

In contrast to the case of recovery, we believe that there are fewer reasons to assert that there exist fundamental structural differences in waste paper utilization between richer and poorer countries. In both cases the utilization of waste paper is ultimately determined by the input choices made by (more or less) cost-minimizing paper and board mills. Production technologies are fairly standardized and available worldwide. This suggests that the impact of economic factors, such as waste paper and virgin fiber availabilities, on waste paper use should be virtually the same in rich and less rich countries. However, at least two factors support the idea that the degree of input flexibility may differ between richer and less-richer countries. First, in countries with lower income levels the paper quality requirements are generally less strict, suggesting that the willingness to invest in waste paper consuming facilities as (long-run) relative prices change may be higher here than is the case in higher-income countries. Second, however, many developing countries are primarily endowed with tropical forests, which often have a lower productivity in supplying wood fiber and also shorter fiber lengths that make them less appropriate for pulp and paper production (Van Beukering and Bouman, 2003). This may suggest that the impact of virgin fiber availability is less important in these – often-poorer – countries than in others.

3. Econometric Models of Global Waste Paper Recycling

This section builds on the discussion above and presents two econometric models, which attempt to capture inter-country differences in waste paper recovery and utilization rates respectively. The first model, the waste paper recovery rate equation, is written in log-linear form and can be formulated as;

$$\ln RR_{it} = \alpha_0 + \alpha_1 \ln GDP_{it} + \alpha_2 \ln URB_{it} + \alpha_3 \ln POP_{it} + \alpha_4 \ln PWP_{it}$$
$$i = 1,\ldots,N;\ t = 1,\ldots,T \qquad (3)$$

where RR_{it} is the recovery rate in country i at time period t and is defined as in equation (1). GDP_{it} measures the gross domestic product per capita (in US$) for the same country and time period. URB_{it} is the urbanization rate expressed as the percentage share of the total population living in urban areas, while POP_{it} denotes the population density, i.e., the total population divided by country i's total land area in square kilometers. Finally, PWP_{it} is the domestic waste paper price in US$ per metric ton. The log-linear form allows us to interpret the α coefficients as elasticities.

Following the discussion in section 2, it should be clear that it is hard to form any *a priori* expectations about the sign of the α_1 coefficient. GDP per capita, we have suggested, reflects two opposing determinants of recovery rates. As income is high so is the labor cost of waste paper collection activities, but the same may hold true for the value that people place on environmental amenities and hence on waste management policies. The former should have a negative influence on recovery rates, while the impact of the latter factor ought to be positive. The size of the net effect therefore remains an empirical question. Since both the urbanization rate and the population density measure the "cheapness" of separating and collecting waste paper, we would expect the coefficients α_2 and α_3 to be positive. The α_4 coefficient should also be positive since higher waste paper prices imply a higher revenue from waste paper collection activities.

The second regression model, the waste paper utilization rate equation, is also in log-linear form and it is expressed as:

$$\ln UR_{it} = \beta_0 + \beta_1 \ln FR_{it} + \beta_2 \ln PM_{it} + \beta_3 \ln SE_{it} + \beta_4 \ln RR_{it}$$
$$i = 1,\ldots,N;\ t = 1,\ldots,T \tag{4}$$

where UR_{it} is the utilization rate in country i and time period t defined as in equation (2). FR_{it} denotes the growing stock of forests divided by the total number of inhabitants in country i for a given time period t, a variable intended to measure the relative availability of virgin fibers. Since virgin fibers are substitutes to waste paper in paper and board production we would expect the coefficient β_1 to be negative.

Inter-country differences in the composition of paper and board production are measured by the share of total paper and board production (in tons) that constitutes newsprint, tissue and liner and fluting board so that:

$$PM_{it} = \frac{(\text{The sum of newsprint, tissue and liner and fluting board production})}{\text{Total paper and board production}} \tag{5}$$

Increases in PM_{it} should lead to increases in the utilization rate, since intensive use of waste paper is particularly high in the production of these three paper qualities. Thus, the coefficient β_2 ought to be positive.

SE_{it} and RR_{it} are both waste paper supply indicators, where SE_{it} (the structural effect) is the share of production to consumption of paper and board in country i at time period t. If, for example, consumption is significantly greater than production this is a reflection of the fact that supply of waste paper is relatively high in relation to demand for waste paper (in production). This reflects a high waste paper availability (price) and the ratio SE_{it} will be low while the utilization rate will be high. In other words, the β_3 coefficient is expected to have a negative sign. RR_{it}, on the other hand, measures instead the *intensity* with which the above supply is exploited. Naturally, if the recovery rate increases it would be relatively easier to obtain waste paper for use in paper and board production. Hence, β_4 should have a positive sign.

One drawback of the above waste paper availability measures, however, is that they take no explicit account of trade in waste paper although some countries are major importers of waste paper. To remedy this we also included the import price of waste paper (in US$ per metric ton) and estimated an alternative version of the utilization rate model (using a more limited data sample across 39 countries). However, in neither of the estimations changes in the import price had a statistically significant impact on the utilization rate (see Berglund and Söderholm (2003) for detailed results). As was noted above, one important reason for this result is that our focus is on inter-country differences in waste paper utilization, and these are determined mainly by long-term investments in capital-intensive paper and board mills. For these mills annual import prices for waste paper do not play a major role in the investment decision-making process, as these prices tend to be very volatile over time.[8] In addition, for some countries waste paper imports still play a very marginal role. For the above reasons we neglect the impact of import prices on utilization in the ensuing analyses.

4. Data and Model Estimation Issues

Our data are drawn from a panel of 49 countries for the period 1990-1996. Data sources and definitions are summarized in Table 2. Since waste paper price data only are available for a more limited set of countries, our recovery rate model as estimated using data across 39 countries but over the same time period. Apart from analyzing this data set as a whole, the panel data approach also provides us with enough degrees of freedom to divide the total sample into two categories; one “rich” country sample and one “middle-income” country sample. This permits us to test the hypothesis of structural differences in recycling behavior in

[8] This does not exclude, of course, that import prices may play an important role in short-term decisions, thus implying increased waste paper imports in years when the domestic market is strong and vice versa. See, for instance, Grace et al. (1981) and Yohe (1979).

rich and middle-income countries. In dividing the sample between “rich” and “middle-income” countries we relied on the World Bank’s income classification (World Bank, 2001), which states that “rich” countries have a GDP per capita of more than US$ 10 000 (1990 prices), while “middle-income” countries include countries with a GDP per capita between US$ 4000 and US$ 9999.[9] Appendix A lists some basic data for all countries that are included in the investigation.

Table 2: Data Sources and Definitions*

Variables	***Definitions***	***Sources***
Recovery rate (RR)	Waste paper recovery divided by paper and board consumption (%)	Pulp & Paper International (1998)
Utilization rate (UR)	Wastepaper consumption divided by paper and board production (%)	Pulp & Paper International (1998)
GDP per capita (GDP)	Purchasing power parity (PPP) estimates in international dollars**	World Bank (1998) and CIA (1999)
Population density (POP)	Number of inhabitants per square kilometer	World Bank (1998)
Urbanization rate (URB)	Percent of total population living in urban areas (%)	World Bank (1998)
Waste paper price (PWP)	Prices paid in US$ per metric ton	FAO (2001)
Structural effect (SE)	Paper and board production as a share of paper and board consumption (%)	Pulp & Paper International (1998)
Paper product mix (PM)	The sum of newsprint and liner and fluting board production as a share of total paper and board production (%)	Pulp & Paper International (1998)
Virgin forest supply (FR)	Growing stock of forest in million cubic meters divided by total population	FAO (1996), World Bank (1998) and CIA (1999)

* The data for the growing stock of forest are only for the year 1990. All other figures are for 1990-1997.

** The *international dollar*, developed by the World Bank, is the unit of account that equalizes price levels in all participating countries. It has the same purchasing power over total GNP as the U.S. dollar in a given year, but purchasing power over subaggregates is determined by average international prices at that level rather than by U.S. relative prices.

[9] Unfortunately we were not able to include what the World Bank refers to as “low-income” countries (with a 1990 GDP per capita lower than US$ 4000) in our investigation. Data on waste paper use and recovery in the less-developed countries, such as the African countries, are sparse and tend to be very unreliable.

In order to employ the regression models in equations (3) and (4) empirically we need to specify the choice of econometric technique and the stochastic framework. In this study we employ a one-way error component model in which the error term (ε_{it}) for each of the two equations can be decomposed into two parts so that:

$$\varepsilon_{it} = \lambda_t + \nu_{it} \tag{6}$$

where λ_t denotes the unobservable time effect and ν_{it} is the remainder stochastic disturbance term. The term λ_t is country-invariant and accounts for any time specific effects that are not included in the regression. A common approach is to assume that these effects are fixed over countries for a given time period, and then eliminate the time-specific disturbance component by introducing dummy variables for each time period. This is known as the fixed effects model or the least squares dummy variable model.[10] However, a Chow test of the null hypothesis that $\lambda_1 = \lambda_2 = ... = \lambda_{T-1} = 0$ was performed, and it suggested that the hypothesis of common intercept terms could not be rejected.[11]

Baltagi and Griffin (1984) suggest that if the variation between countries greatly exceeds the within variation (i.e., the variation over time), then ordinary least squares regression (OLS) with common intercept as well as slope coefficients becomes the preferred estimator. Our data include variables such as urbanization rate, population intensity, GDP per capita, paper product mix etc., which certainly vary much more between countries than within countries. For the above reasons, the results presented in this paper will be based on OLS. The reliance on cross-country variation can be assumed to capture long-run responses since the differences between variables in this case exhibit a wide range of variation and are the result of long-standing political and economic factors (e.g., Atkinson and Manning, 1995).

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[10] A more common approach in panel data applications is to assume the existence of cross-section (in our case country) specific effects, and introduce dummy variables for each cross-section unit (country). However, in our case we are primarily interested in the variation across countries (reflecting long-run changes), and the use of country-specific dummies would remove this variation.

[11] This is a simple *F*-test outlined in Baltagi (1995, p. 12). The *F*-statistic for the different models (i.e., recovery and utilization rate equations for “rich” and “middle-income” countries respectively) was never higher than 1.19, while the critical value of *F* at the one percent significance level is 2.80.

By performing a Hausman test we could reject the null hypothesis that the logarithm of RR_{it} is uncorrelated with the remaining error term (ν_{it}), and hence that $\ln RR$ is an exogenous variable in the utilization rate equation.[12] This problem of simultaneity was solved by the use of instrumental variables. We regressed ln*RR* on a set of variables considered exogenous (ln*GDP*, ln*FR*, ln*PM*, ln*SE*, ln*URB*, ln*POP*), and employed the fitted values from these first regressions as instruments in place of ln*RR* in equation (4). Similarly, we also found that the logarithm of the waste paper price ($\ln PWP$) was an endogenous variable in the recovery rate equation. Fitted values of ln*PWP* (employing the instrumental variables ln*GDP*, ln*FR*, ln*PM*, ln*SE*, ln*URB*, ln*POP* and the logarithm of the waste paper import prices), were thus used in the recovery rate estimations.

The models were also tested for the possible existence of heteroscedasticity using White's (1980) test, and in all cases the null hypothesis of homoscedasticity was rejected. The presented standard errors for the OLS models have therefore been calculated from the heteroscedastic-consistent variance-covariance matrix (MacKinnon and White, 1985).

Finally, the extensive reliance on country data involves a risk that some influential observations (i.e., outliers) may have a demonstrably larger impact on the calculated coefficients than is the case for most of the other observations. These outliers may in turn create large differences between sub-samples. We checked for the influence of outliers by: (a) plotting the residuals against the predicted values; and (b) introducing dummy variables for country observations with relatively high residuals (Baltagi, 1998; Cook and Weisberg, 1982). This procedure identified two countries with relatively high residuals and statistically significant dummy variable coefficients, New Zealand and Chile. Still, when these observations were removed from the regression runs, the changes in results were very modest and these countries are therefore included in the results presented below.

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[12] The underlying idea of the Hausman test is to compare two sets of estimates, one of which is consistent under both the null and the alternative hypothesis, and another, which is consistent only under the null hypothesis. We carried out the test by running an auxiliary regression and applying the standard *t*-test (Hausman, 1978). Detailed test results are available from the authors on request.

5. Empirical Results and Discussion

This section presents and discusses the results of the regression analyses, starting with the OLS estimates from the full sample of 49 and 39 countries, respectively, and proceeding with the corresponding estimates for the two sub-samples (“rich” and “middle-income” countries).

5.1 Full Sample Estimations

Table 3 presents the parameter estimates for the coefficients in the recovery rate and utilization rate equations together with adjusted *R*-square measures. Starting with the recovery rate regression model we note that all the four included variables tend to be important determinants of inter-country differences in recovery rates. *First*, the coefficient representing the impact of GDP per capita (α_1) has the expected sign, and it is statistically significant at the one percent level. A one percent increase in GDP per capita will, *ceteris paribus*, yield a 0.17 percent increase in the recovery rate. This result is consistent with the notion that waste paper recovery, and in particular the environmental benefits that are expected to follow from it, is a luxury good. As countries grow richer, the value that people assign to environmental issues will increase and there is likely to be a greater demand for waste reduction policies. We also hypothesized that growing GDP per capita implies a higher opportunity cost of labor, and since paper recovery and collection in many cases has been a relatively labor-intensive activity growth in per capita income would lead to decreases in the recovery rate. We are unable to separate the impacts of these two different effects, but we can nevertheless conclude that the *net effect* on the recovery rate of increases in GDP per capita appears to be positive. Apparently, increased willingness of people to support and participate in paper recovery programs tend to be more important than labor costs in the recovery sector. This may be explained by the fact that even though waste paper recovery is relatively labor intensive when compared to the use of virgin fibers, labor costs still represent a relatively small share of the total costs of paper recycling (e.g., Van Beukering, 1994).

Second, the coefficients for the demographic variables (α_2 and α_3) both have the expected positive signs and both are also statistically significant.[13] This indicates that the lower the costs of waste paper collection and recovery, in terms of transport (which often

[13] It should be noted that (the logarithmic forms) of the urbanization rate and the population density are not highly correlated. In our sample the correlation coefficient between the two is only 0.01. Thus, it seems not to be the case that a large part of the variation in URB is explicable by the variation of POP.

constitute more than 50 percent of total collection costs) etc., the higher the recovery rate. Hence, in countries with high population densities and urbanization rates, recovery rates will tend to be higher, this since waste paper collection will be cheaper and disposal will tend to be costly. *Third*, a higher price of waste paper induces more intensive paper recovery activities. This strengthens our case that – in addition to policy impacts – waste paper recovery is also affected largely by economic factors such as prices and costs.

Table 3: OLS Estimates for the Recovery Rate and Utilization Rate Equations: Full Sample*

Recovery Rate (RR)		***Utilization Rate*** (UR)	
Constant (α_0)	-2.156	Constant (β_0)	4.774
	(-3.695)		(10.816)
GDP (α_1)	0.171	*FR* (β_1)	0.005
	(2.784)		(0.253)
URB (α_2)	0.741	*PM* (β_2)	0.041
	(2.190)		(0.540)
POP (α_3)	0.069	SE (β_3)	-0.821
	(4.450)		(-12.089)
PWP (α_4)	0.735	*RR* (β_4)	0.687
	(4.754)		(5.797)
R-square (adj)	0.21	*R*-square (adj)	0.54

* *t*-statistics are given in parentheses.

However, our goodness-of-fit measure for the recovery equation, *R*-square (adj), is only 0.21. In other words, as much as 79 percent of the variation in recovery rates is left unexplained and is thus due to variation in the error term, ε_i, or to variations in other variables that implicitly form part of the error term.[14] As was noted above, one important variable that we have not included *explicitly* in the recovery rate equation is the impact of waste management policies on recovered paper supply. Parts of this impact, however, are likely to be reflected in the GDP measure and to some extent in the recovery cost variables. However, what is probably not reflected in these variables is the *efficiency* with which waste

[14] One should note, however, that low *R*-squares are very common for cross-section samples (Greene, 1993).

management policies are implemented.[15] Future research efforts should, therefore, explore in more detail such differences between countries.

If we proceed to analyze the parameter estimates for the utilization rate equation, we first note that the *R*-square (adj) is 0.54. We have thus been more successful in explaining the inter-country variation in utilization rates than was the case with recovery rates. All coefficients, except β_1 representing the impact of virgin fiber availability per capita, have the expected signs. However, the impact measured by β_1 is highly statistically insignificant.

Furthermore, as expected our model predicts that in countries in which the share of "intensively-recyclable" paper products is high, utilization rates tend to be high. This effect, though, is insignificant from both an economic and statistical point of view. Our two indicators of waste paper availability, *SE* and *RR*, tend to be the major determinants of inter-country differences in utilization rates, and the coefficients representing these impacts are also statistically significant. The two measures are correlated (with a correlation coefficient of 0.56), and it is thus hard to sort out in detail the exact contribution of each of these two variables. What is clear, however, is that countries endowed with abundant waste paper supplies (whatever the measure) overall have high utilization rates.

This suggests that waste paper utilization largely is market-driven. In other words, overall paper and board mills do not use waste paper primarily because public policy "mandate" them to do so; they use it because it is widely available and thus cheap. Of course, public policy *will* affect waste paper utilization indirectly through policies' impact on the recovery rate, and hence on the relative cheapness of waste paper.

As was noted in section 3 our reliance on inter-country variations primarily reflects long-run responses. In the short-run the link between increased recovery and utilization may not always be apparent in waste paper markets. For example, in many cases collection schemes have led to abundant waste paper supplies, but this has not induced a direct increase

[15] Additional explanatory variables were included in the recovery rate regression model, i.e., a proxy of the overall level of education, an infrastructure variable describing the extent of paved roads in the country, and an environmental sustainability index. However, none of these added any new information and were therefore removed from the final model. In addition, the population density variable was replaced by a variable defined as paper and board consumption divided by total land area. Still, these two models produced very similar results and the two variables were highly correlated with a correlation coefficient of 0.85. This was also the case when the GDP variable was replaced by the average wage level.

in use but instead substantial price slumps.[16] Again, this reflects the lumpiness (and thus the inflexibility) of the capital stock in the paper and board industries, and waste paper input choices therefore primarily derive from the investment in new capacity.

To sum up, a major part of the supply potential is not policy determined. It is instead largely an effect of national comparative advantage in foreign trade of paper and board products. Waste paper supply ultimately stems from recent paper and board consumption. As a consequence, in countries with a comparative advantage in producing and exporting paper products, paper production will be significantly greater than consumption implying net exports. In such cases the availability of waste paper will necessarily be comparatively low, not because poor waste management policies are in force but as a consequence of the competitive position in the world market for new paper and board products. It is not clear why policy makers should try to alter this situation only to achieve a higher waste paper utilization rate (see also Huhtala and Samakovlis, 1999).

5.2 Rich and Middle-Income Sample Estimations

For the recovery equation an *F*-test rejected the null hypothesis that the two sub-samples, "rich" and "middle-income" countries, could be combined into one single regression model.[17] In this section we therefore present and discuss the regression results from the two sub-samples. Table 4 displays the parameter estimates for each of the two recovery rate regressions.

The adjusted *R*-square measures are relatively low for both models (0.26 and 0.28, respectively). In spite of the overall low level of explanatory power the results provide some additional support for the notion that GDP per capita, population density, urbanization rate and waste paper prices are important determinants of waste paper recovery rates.

[16] See, for example, Browne (1996) for an analysis of such behavior in the U.S. newsprint market. To accurately investigate this issue, time series data on a monthly basis are required, and such an analysis is beyond the scope of this paper.

[17] The *F* statistic for this test was 8.27 while the critical value of *F* with 5 and 333 degrees of freedom is 3.02 at the one percent significance level. This is a simple Chow test outlined in Dougherty (1992, pp. 279-282).

Table 4: OLS Estimates for the Recovery Rate Equation: Rich and Middle-Income Country Samples*

Recovery Rate (RR)***: Middle-Income Countries***		***Recovery Rate*** (RR)***: Rich Countries***	
Constant (α_0)	-2.989	Constant (α_0)	-7.639
	(-2.546)		(-3.832)
GDP (α_1)	-0.035	*GDP* (α_1)	1.613
	(-0.182)		(4.843)
URB (α_2)	1.554	*URB* (α_2)	-0.018
	(5.160)		(-0.125)
POP (α_3)	0.244	*POP* (α_3)	0.065
	(3.205)		(3.971)
PWP (α_4)	0.674	*PWP* (α_4)	1.101
	(3.064)		(3.684)
R-square (adj)	0.26	*R*-square (adj)	0.28

* *t*-statistics are given in parentheses.

However, there exist three notable differences between the two sub-samples. *First*, the impact of GDP per capita on recovery rates is much more pronounced in the "rich" countries than is the case for "middle-income" countries. A one percent increase in GDP per capita will, ceteris paribus, lead to a 1.61 percent increase in the "rich" countries, while the corresponding impact in the "middle-income" countries is negative but very low (-0.035 percent). This result is entirely consistent with our notion that recycling activities tend to be more policy-driven in higher-income countries, and manifested through relatively strict environmental policies and mandatory recycling schemes. In the "middle-income" countries, however, increases in income tend not to be as highly correlated with increased concerns about waste management. Also, recovery activities are likely to be more labor-intensive in "middle-income" countries than in richer ones.

Second, the recovery of waste paper in the "middle-income" countries is instead more of an economically driven phenomenon in the sense that it tends to be more sensitive to cost factors than in the more wealthy countries. If population density increases by one percent the model projects a 0.24 percent increase in the recovery rates of the "middle-income" countries, but only a modest 0.06 percent increase in the "rich" countries. A similar pattern is found for the urbanization rate. For the "middle-income" sample we find a relatively large (an elasticity of 1.55), positive and statistically significant coefficient, while the corresponding coefficient for the "rich" sample is negative and highly statistically insignificant. This latter result is well in line with earlier studies, which primarily stress the importance of *urbanization* on wastepaper recovery, and especially in developing countries (e.g., Van Beukering and Sharma, 1996). Projections show that the share of total population living in cities will grow at a fast rate in the future (World Bank, 1995), and our results suggest that in many less-developed countries this will, ceteris paribus, have a substantially positive impact on waste paper recovery behavior.

Third, in both models increases in waste paper prices lead to higher recovery rates, and this impact is highly statistically significant. However, recovery activities tend to be more sensitive to waste paper price changes in the "rich" countries than in "middle-income" countries. The above may reflect the fact that the existing recycling infrastructure is more developed in the former countries, and this makes supply adjustments easier. However, in the less-developed countries recovery activities are – in the absence of effective recycling schemes – instead more dependent on high population densities and urbanization rates.

The estimated parameters for the respective utilization rate equations are reported in Table 5. Also in this case an *F*-test rejected the null hypothesis of a single, combined, regression for the two samples.[18] There exist some notable differences in the results from the two regressions. Most notably perhaps, the adjusted *R*-square is significantly higher for the "rich" country sample (0.68) than for the "middle-income" sample (0.27). This may suggest that other variables than those included in the regression are important determinants of utilization rates in "middle-income" countries. It may also be the case, however, that paper-recycling activities in these countries are more informal and are therefore not captured by the official statistics, creating noise in the results.

The regression results based on the two sub-samples support the notion that the utilization rate is largely driven by the degree of waste paper availability, measured by *RR* and *SE*. The large *SE*-coefficients indicate that, regardless of whether we consider "rich" or "middle-income" countries, the higher the proportion of paper and board production which is exported the more difficult it is to increase domestic utilization rate goals without interfering with established paper and board trade patterns (Huhtala and Samakovlis, 1999). As expected, in the case of the utilization rate equation the differences between the regression coefficients are not as pronounced as in the recovery rate case. Still, the impacts of the *RR* and *SE*

[18] The *F* statistic for this test was 2.44, while the critical value of *F* with 5 and 333 degrees of freedom is 2.21 at the five percent significance level. This is a simple Chow test outlined in Dougherty (1992, pp. 279-282).

variables are somewhat larger in the “middle-income” sample compared to the “rich” country case. This, again, is partial support for our hypothesis that waste paper utilization is a more market-driven phenomenon in countries with lower incomes, and the result is consistent with the notion that the input flexibility is higher in “middle-income” countries due to lower quality requirements and less capital-intensive production facilities.

Table 5: OLS Estimates for the Utilization Rate Equation: Rich and Middle-Income Country Samples*

Utilization Rate (UR)***: Middle-Income Countries***		***Utilization Rate*** (UR)***: Rich Countries***	
Constant (β_0)	4.532	Constant (β_0)	4.232
	(8.732)		(3.480)
FR (β_1)	0.002	*FR* (β_1)	-0.048
	(0.068)		(-1.815)
PM (β_2)	0.013	*PM* (β_2)	0.227
	(0.190)		(0.976)
SE (β_3)	-0.801	*SE* (β_3)	-0.703
	(-4.684)		(-8.598)
RR (β_4)	0.798	*RR* (β_4)	0.502
	(2.707)		(2.982)
R-square (adj)	0.27	*R*-square (adj)	0.68

* *t*-statistics are given in parentheses. The instruments used include the natural logarithms of *GDP*, *FR*, *PM*, *SE*, *URB* and *POP*.

The *PM*-coefficients are low and statistically insignificant, suggesting that the paper product mix also here is an unimportant determinant of cross-country differences in the waste paper utilization rate. The coefficients representing the impacts of virgin forest supply (*FR*) show that for the “middle-income” sample the impact of the relative availability of virgin forest resources has the wrong sign (positive) but is low and statistically insignificant. However, for the “rich” country sample we find that the corresponding coefficient is higher, negative and statistically significant (but only at the 7 percent level), partly reflecting the higher productivity of the wood in many developed countries for pulp and paper production. This latter result is very reasonable since waste paper and wood pulp ought to be substitutes.

6. Concluding Remarks

By employing a pooled data set of 39 and 49 countries, respectively, over the period 1990-1996, this paper has attempted to identify and analyze the main determinants of inter-country differences in waste paper recovery and utilization rates. The following general conclusions can be drawn from the analysis. Both economic and political factors affect recovery and utilization rates. Rich countries tend to experience high recovery rates, and in a "rich" country a given increase in income will lead to a greater impact on recovery behavior than is the case in a "middle-income" country. The maintained hypothesis in this paper is that this relationship can be explained by the fact that the avoided costs of waste management tend to be valued more highly in richer countries. As higher recovery rates imply higher waste paper availability, this will in turn positively affect utilization rates.

We also gain support for our hypothesis that the two recycling rates are largely economically (and not policy-) determined. For example, recovery rates are influenced by demographic features such as population density and urbanization rate, which together determine the cost of collection and recovery and especially so in "middle-income" countries. Furthermore, waste paper availability is the most important determinant (both statistically and economically) of waste paper utilization rates. Waste paper availability is in turn determined largely by the so-called structural (or trade) effect, which ultimately reflects the countries' comparative advantage in world paper and board markets.

In other words, differences in waste paper recovery and utilization rates are to a great extent due to economic characteristics about which very little can, or perhaps even should, be done. This indicates that the two recycling rates may be of limited use as policy measures. Policy efforts intended to affect the rates may turn out to be very costly or difficult to enforce as they run counter to other environmental goals (see also Huhtala and Samakovlis, 1999).

For example, attempts to mandate a certain utilization rate (in terms of recycled content standards for paper and board products) may lead to unintended behavior. If a country that is a major net exporter of paper and board faces a recycled content goal, it may have to import waste paper in order to comply since its domestic supply will be too scarce. Canada is one example where this has occurred. In the early 1990s the USA introduced recycled content standards for newsprint and the Canadian newsprint industry, which is a major exporter to the U.S. market, therefore found it necessary to import waste paper (from the USA) in order to meet the standards (Roberts and Johnstone, 1996). Also Swedish paper and board producers tend to import cheap waste paper (partly due to subsidized collection) from Germany to meet requirements. As transport over long distances are required it is not clear that this behavior favors the environment, which the policies primarily were put in place to do.[19]

The above suggests that policy makers need to recognize that any recycling policy instrument needs to take into account the specific features of each country and region. Harmonized standards, neither for recovery rates nor for utilization rates, appears desirable. More fundamentally, one needs also to recognize that waste paper recycling seldom is a desirable goal *in itself*. It may even be stated that a recycling policy that solely postulates quantitative targets such as "50 percent of all waste paper should be recovered" is good for nothing unless it is logically derivable by analysis of more basic objectives and values and of the relevant costs and constraints. People are primarily interested in the waste management and economic benefits of recycling, and our policy variables need to reflect this. Recovery and utilization rates appear to be only of limited use in this sense.

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[19] Of course, an alternative to increased waste paper imports would be to relocate the paper and board industry. Still, the attractiveness of this option can also be questioned from an environmental and waste management view (Huhtala and Samakovlis, 1999). See, for instance, Michael (1998) for an analysis of how increased U.S. paper recycling affects the distribution of pollution between the USA and Canada.

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Appendix A: Countries Included in the Econometric Analysis

Rich countries				Middle-income countries			
Country	**GDP/capita 1990/1996**	**RR (%) 1990/1996**	**UR (%) 1990/1996**	**Country**	**GDP/capita 1990/1996**	**RR (%) 1990/1996**	**UR (%) 1990/1996**
Australia	15438/20596	33/36	46/49	Algeria*	4554/4870	17/21	40/71
Austria	17520/21701	52/71	39/42	Argentina	6725/9652	41/31	40/44
Belgium	17968/22190	33/38	22/27	Brazil	5390/6490	34/37	32/36
Canada	18662/22104	23/42	9/24	Bulgaria	4855/4455	20/53	42/66
Denmark	17595/22695	35/54	80/123	Chile	7255/12013	45/9	39/8
Finland	16640/18837	31/35	5/6	Colombia*	5230/7000	44/39	56/61
France	17747/21585	35/42	47/50	Costa Rica	5048/6479	7/8	63/122
Germany	17198/21212	47/71	50/60	Czech rep.	8138/11116	33/38	27/36
Greece	11085/12476	38/29	75/74	Ecuador*	4094/5099	25/16	114/162
Hong Kong*	16770/24260	52/61	112/100	Hungary	6477/6952	52/61	49/78
Ireland	11820/18684	5/22	100/107	Iran*	4229/5351	11/11	31/36
Israel*	13442/18520	21/24	73/78	Lebanon*	5165/5928	60/63	100/113
Italy	16364/20139	27/31	46/51	Lithuania	5412/4442	54/40	33/29
Japan	18176/23158	50/52	52/54	Malaysia	6588/10905	14/18	37/44
Netherlands	15978/20503	51/64	65/67	Mexico	6721/7983	42/45	75/88
N. Zealand*	14004/17758	16/25	7/8	Panama	5150/7209	19/23	100/186
Norway	16542/23464	28/50	9/11	Poland	4623/6016	39/38	35/39
Portugal	10524/13535	40/40	43/31	Russia	5083/4269	25/29	22/18
Singapore	22880/26680	37/66	130/93	Slovak rep.	5919/7478	52/31	40/35
Spain	12291/15499	39/41	64/74	Slovenia	8904/12010	33/37	39/46
Sweden	16802/19588	46/52	12/17	South Africa	6749/7623	33/37	27/34
Switzerland	22609/24811	49/67	48/65	Thailand*	4000/6873	38/44	76/80
UK	16067/19917	33/40	59/69	Tunisia*	4319/4811	10/17	28/14
USA	22527/28023	34/45	29/39	Turkey	4591/5977	30/33	39/56
				Venezuela	7071/8321	34/45	61/66

* These countries are not included in the recovery rate estimations.

Sources: See Table 2.

Complementing Empirical Evidence on Global Recycling and Trade of Waste Paper

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Abstract

This comment provides a critical analysis of Van Beukering and Bouman's article on global paper recycling and trade. We first question their notion that developing countries specialize in waste paper utilization and developed countries in recovery activities primarily because of different patterns of waste paper trade. An increased focus on relative waste paper availability, we argue, provides us with a better understanding of global paper recycling. We also criticize some of the implicit assumptions made in their regression analysis of waste paper utilization rates. An alternative regression model is therefore derived and estimated. In contrast to the approach used by Van Beukering and Bouman our analysis: (a) is consistent with basic microeconomic theory; (b) distinguishes clearly between short- and long-run impacts; and (c) produces results that support our initial conjecture that waste paper availability is the most important determinant of waste paper use.

Keywords: waste paper, recycling, trade, developing countries, developed countries, utilization

Forthcoming in ***World Development***

1. Introduction

In a recent *World Development* article (Vol. 29, No. 10, 2001), Pieter Van Beukering and Mathijs Bouman (hereafter V/B) provide an interesting and instructive analysis of global recycling and trade of paper and lead. The main objective of their analysis is to understand what are the main determinants of recovery and utilization rates of lead and waste paper. V/B's primary hypothesis "is that developed and developing countries differ in terms of trade and recycling. Developed countries are characterized by high waste recovery and high export rates of secondary commodities while developing countries are characterized by high waste utilization and high import rates of secondary materials," (p. 1718). This hypothesis is tested in two basic sections, one qualitative and one quantitative.

The qualitative part of their article builds on graphical comparisons of trade and recycling patterns. Average data for a large set of developed and developing countries are employed. In the quantitative section V/B perform a regression analysis using pooled time series and cross-country data. This allows them not only to test their main hypothesis, but also to analyze additional determinants of paper and lead recycling. In both sections V/B present evidence that support their hypothesis, especially for the waste paper sector. "The main conclusion is that trade [of recyclable products] can be beneficial to some – the utilizers of secondary materials – and detrimental to others – the recoverers of recyclable materials," (p. 1733). They also conclude that a set of geographic variables (e.g., population density, primary commodity endowment) have noteworthy influences on recycling behavior, while the impacts of a number of "market-related" variables (e.g., relative price of waste paper and wood pulp) are referred to as "ambiguous".

Our comment focuses solely on V/B's analysis of waste paper recycling, although many of our arguments are general and therefore apply equally well to lead and to other recyclable products. We question the analysis of V/B on two related points. *First*, we do not believe that the different patterns of waste paper trade provide the main explanation for why recovery and utilization rates differ between countries. Instead there exist both theoretical and empirical reasons to focus explicitly on different measures of waste paper availability. These reasons and some supporting empirical evidence are laid out in section 2. *Second*, in section 3 we develop this discussion and focus on the regression analysis of V/B. Here a critical analysis of the (implicit) theoretical bases and assumptions underlying their model is provided. We then derive and estimate an alternative econometric model. In contrast to the approach used by V/B our analysis, we claim: (a) is consistent with basic microeconomic theory; (b) distinguishes clearly between short- and long-run impacts; and (c) produces results that support our initial conjecture that waste paper availability is the most important determinant of waste paper use. Finally, section 4 provides some concluding comments.

2. Why do Utilization Rates and Recovery Rates Differ?

2.1 Basic Definitions

V/B provide a detailed discussion on how to measure recovery and utilization rates for secondary commodities. Their definitions follow those of most other studies. For waste paper the recovery rate (*RR*) is expressed as follows:

$$RR = \frac{WP_{con} + \left(WP_{ex} - WP_{im}\right)}{PB_{con}} \qquad (1)$$

where WP_{con} is waste paper consumption, WP_{ex} is waste paper exports, WP_{im} denotes waste paper imports, and PB_{con} is total consumption of paper and board (the final commodity). Thus, *RR* will measure the extent to which a certain country is able to recover used paper and board products from the waste stream. In essence, *RR* therefore represents the supply side of the waste paper market. The utilization rate (*UR*), on the other hand, captures the demand side and reveals to what extent the recovered paper is actually being used in producing new final products. *UR* is therefore normally expressed as:

$$UR = \frac{WP_{con}}{PB_{prod}} \qquad (2)$$

where PB_{prod} denotes paper and board production.

2.2 The Approach of V/B

In the case of waste paper V/B analyze changes in the above recycling rates by employing time series and cross-section data over 27 years for 50 countries. The maintained hypothesis of their investigation is that differences in recovery and utilization rates are due to a process of waste paper trade specialization. They conclude: "[…] a country is specializing in either utilization by importing the additional secondary materials from abroad […] or in recovery by exporting the surplus of generated secondary materials," (p. 1720). This notion makes sense since the higher are the net exports of waste paper $\left(WP_{ex} - WP_{im}\right)$ the lower (higher) the utilization rate (the recovery rate) tends to be (see also below).

V/B then go on to suggest that there is a major difference between developing and developed countries in terms of such specialization. "High recovery and export rates of [waste paper] commodities characterize developed countries. High utilization and import rates of [waste paper] characterize developing countries," (p. 1721). Norway and Indonesia are used as illustrating examples. Indonesia has a high utilization rate and also a high waste paper import dependency (defined as the amount of waste paper imported as a share of total consumption of waste paper). Norway, on the other hand, is a net-exporter of waste paper and consequently its waste paper utilization rate is low. This is in spite of the fact that Norway lacks abundant supplies of primary resources and therefore is highly dependent on imports of wood pulp as well as final paper and board products.

When analyzing their entire data set, all 50 countries, V/B gain additional support for their hypothesis of specialization in the waste paper sector. They calculate aggregate five-year averages of utilization and recovery rates for developing and developed countries respectively. They also look at changes in the import- and export-dependencies of paper commodities, and conclude that overall "[d]eveloped countries have become relatively more export-dependent and developing countries have rapidly become more import-dependent," (p. 1722). This general pattern is particularly significant for waste paper trade, and this, V/B claim, explains why developing countries specialize in waste paper utilization and developed countries in recovery activities.

2.3 Our Approach

Our scrutiny of the analysis of V/B follows two steps. First we present some additional empirical facts to test the hypothesis that developing countries tend to have high waste paper utilization rates while developed countries instead have high recovery rates. Second, our major task in this paper will be to provide a critical analysis of the notion that waste paper trade patterns is the main determinant of this global pattern.

One problem with the analysis of V/B is the use of aggregate figures when comparing utilization and recovery trends. *On average* there seems to be a significant difference in waste paper specialization between developing and developed countries. It should however be clear that such average figures can be heavily determined by the influence of outliers and/or conceal important differences between countries. It therefore makes sense to test this hypothesis by also looking at individual country data. Figure 1 shows the waste paper utilization and recovery rates in 1997 for some selected developed and developing countries. With one minor exception these countries are identical to those used by V/B.[1]

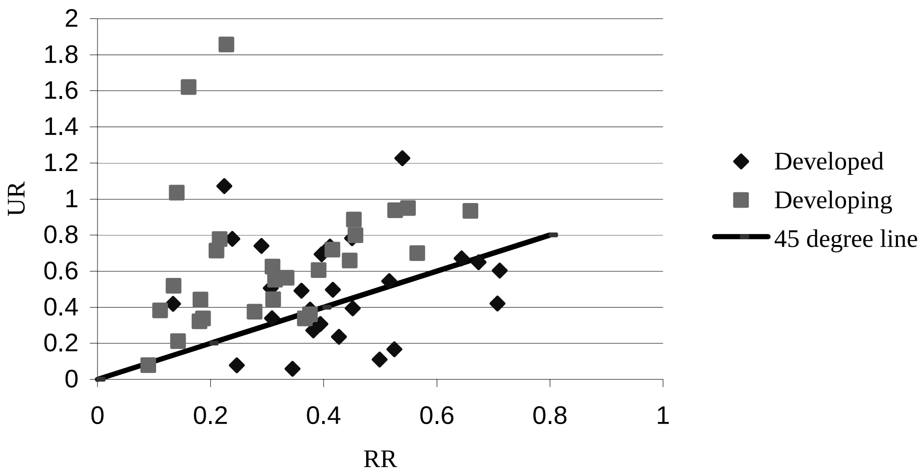

Figure 1. Waste paper recovery and utilization rates (1997)
Source: Pulp and Paper International (1998).

Figure 1 confirms V/B's claim that almost all developing countries tend to specialize in relative use of waste paper. However, for the developed countries the pattern is not that evident. It is certainly the case that many developed countries recover more waste paper than they use (in relative terms of course), but it is equally common that they are "net utilizers". In addition, many of the forest-rich countries, such as Canada and Sweden, who have very low utilization rates are also *net importers* of waste paper, a finding that is in contrast with the results of V/B.[2] In sum, even though we gain some limited support for V/B's hypothesis, the underlying reason for the documented global recycling patterns presented by them is not convincing. The remainder of this comment tries to explain why this is the case and based on this we also present some complementing empirical evidence on waste paper recycling.

To illustrate our points we will focus primarily on the utilization rate, although some remarks about the recovery rate naturally will follow. In the next section we develop a very simple theoretical model in order to derive the main determinants of the waste paper

[1] In order to construct Figure 1 we used data from Pulp & Paper International (1998), and in this publication 1997 recovery and utilization rate data for Honduras were not available.

[2] For example, in 1997 Canada imported 2088 kilotons (kt) of waste paper, while its exports only amounted to 688 kt. Sweden imported 559 kt of waste paper and exported 193 kt (Pulp & Paper International, 1998).

utilization rate. However, before proceeding it is useful to simply take a “mechanical” look at the two recycling rates. From equations (1) and (2) we can see that there are two reasons why *RR* and *UR* can differ. First, the *numerators* may differ in size. The difference between the numerator of the recovery rate definition and that of the utilization rate can be expressed as:

$$\left(WP_{con} + WP_{ex} - WP_{im}\right) - WP_{con} = \left(WP_{ex} - WP_{im}\right) \tag{3}$$

This shows that *one* reason for why the recovery rate may be higher (lower) than the utilization rate in a specific country is that that same country is a net exporter (importer) of waste paper. Clearly this is the approach taken by V/B in their analysis, and there is nothing inherently wrong in this. However, it is not sufficient since there exist yet another explanation. The difference between the *denominator* of the recovery rate definition and that of the utilization rate can namely be expressed as:

$$PB_{con} - PB_{prod} \tag{4}$$

Thus, a country may specialize in recovery simply because it consumes more final products than it produces, i.e., it is a net importer of paper and board products (and vice versa). Thus, the story of waste paper trade patterns provides us only with a partial answer to the question why we witness paper-recycling specialization in the world. It is important to tell the story on paper and board trade as well. In the following we will claim that this latter story is in fact the most relevant one when analyzing changes in utilization rates.

In Figure 2 the waste paper utilization rate for all countries are plotted against the ratio between WP_{ex} and WP_{im} (indicating the degree of net exports of waste paper), while Figure 3 plots the relationship between the utilization rate and the ratio between PB_{prod} and PB_{con} (indicating the degree of net exports of paper and board products).[3] According to the above both relationships ought to be negative. In the waste paper trade case this is the case

[3] In constructing Figures 2 and 3 (as well as Figure 1) we used data for 1997 only, while V/B use aggregate data over a longer time period. Still, V/B claim that the trend towards specialization in waste paper trade has become more and more prevalent over time. In other words, if we are unable to detect such a pattern for a relatively late year like 1997, it is unlikely to exist for earlier years. Furthermore, in constructing Figure 2 we used the same data source as V/B, but were still unable to find the relevant data for Colombia, Honduras, Morocco, Pakistan, Peru, Thailand and Romania. These countries are therefore not included in Figure 2.

(Figure 2), but the simple correlation coefficient over all countries is only –0.08. However, the degree of net exports of paper and board tend to be more negatively correlated with the waste paper utilization rate (Figure 3). In this case the correlation coefficient is –0.51. Thus, in countries that are net exporters of final paper and board products the utilization rate will tend to be low. Again, Canada and Sweden are two illustrating examples.

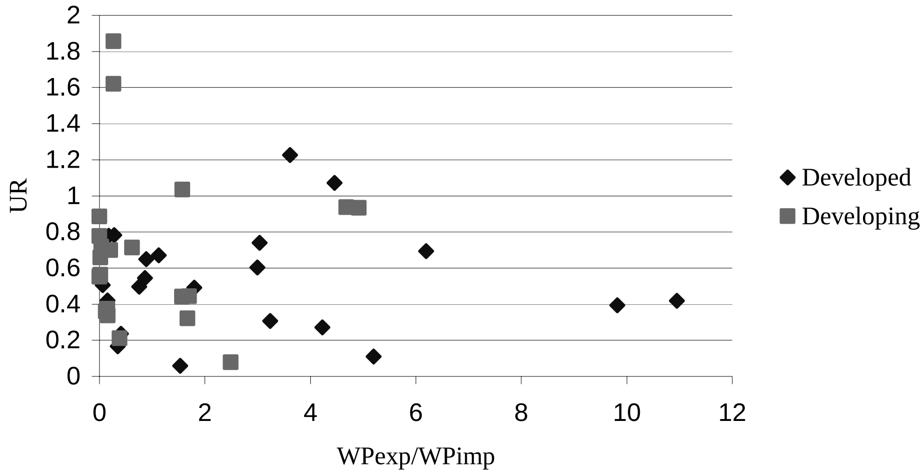

Figure 2. The correlation between utilization rate and net waste paper exports, 1997 (r = -0.08)

Sources: FAO (2001) and Pulp & Paper International (1998).

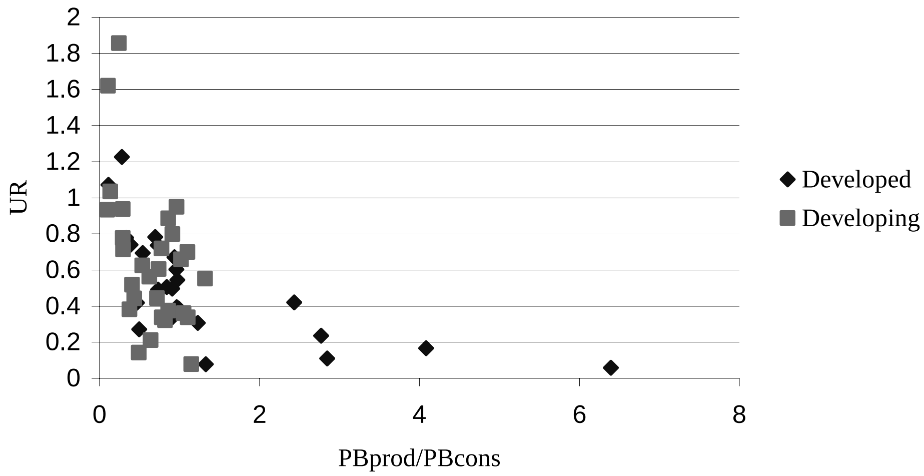

Figure 3. The correlation between utilization rate and waste paper availability, 1997 (r = -0.51)

Source: Pulp & Paper International (1998).

Why do paper and board trade patterns provide a better explanation to differences in waste paper utilization rates than do waste paper trade patterns? To understand this we need to remind ourselves that utilization rates are ultimately determined by the input demand choices of paper and board mills. Consider a mill that chooses its input factors so as to minimize production costs for a given level of output. Clearly this choice will be determined by the relative prices of input factors. The point here is simply that while the term $\left(PB_{prod} / PB_{con}\right)$ expresses a country's comparative advantage in the international paper and board industry, it is also in fact a proxy for the relative availability (and hence the price) of waste paper. Waste paper supplies are drawn directly from previous consumption levels of paper and board, i.e., the higher is PB_{con} the higher is the (absolute) level of waste paper supplies. In addition, the higher is PB_{prod} the higher is (absolute) waste paper demand. Thus, the ratio $\left(PB_{prod} / PB_{con}\right)$ essentially expresses the relative relationship between waste paper demand and waste paper supplies. For this reason high ratios are an indication of high waste paper prices, which in turn are likely to induce low utilization rates. Of course, this is a not a flawless proxy of the relative availability of waste paper (partly because it neglects the impact of waste paper trade). Nevertheless, as was noted above – and as will be illustrated in section 3 using econometric techniques – it appears to be a very important determinant of waste paper utilization patterns. This crucial role of the ratio between paper and board production and consumption has also been recognized in earlier work on waste paper recovery and use (e.g., OECD, 1976).

What meaning will the ratio $\left(WP_{ex} / WP_{im}\right)$ have to the domestic paper and board mill? No meaning at all since this ratio tells us virtually nothing about the relative price of the different input factors; instead it only indicates to what extent a country as a whole has a comparative advantage in the international waste paper market and this is determined by a number of factors such as the costs of recovery activities (both abroad and domestically), foreign and domestic demand for paper and board products, etc. In addition, waste paper imports are an endogenous variable from the mill's perspective, i.e., they are an outcome of the input decisions made by the mill. Imports will not determine these decisions, in the same way as "waste paper availability" will. Firms' input decisions are not determined by trade surpluses; they are determined by costs and prices.

Our approach to differences in utilization rates also adds a new perspective to the Indonesia-Norway example. The ratio between paper and board production and paper and board consumption is much higher in Norway (2.91 on average) for the period 1995-1997 than is the case for Indonesia (1.32). This explains the relatively low waste paper utilization rate in Norway. It is of course still true that Norway is a net exporter of waste paper, and that Indonesia is a net importer, but our point is simply that this tells us very little about the underlying economic reasons for the observed pattern.

3. What Explains the Differing Waste Paper Utilization Rates?

3.1 A Simple Microeconomic Model of Waste Paper Demand

In this section we develop the discussion initiated above on why waste paper utilization rates differ between countries. We outline a very simple microeconomic model, which provides the theoretical underpinnings for an econometric analysis that complements, and partly contradicts, the regression results presented by V/B.

Following the approach in earlier factor demand studies we employ duality theory, which states that if producers minimize input costs there exists a cost function that contains sufficient information to completely describe the underlying production technology (Varian, 1992). For a representative (cost minimizing) mill the average cost of paper and board output, AC, can be written as:[4]

$$AC \equiv \frac{1}{PB_{prod}}\left(P_{WP}WP + P_{VF}VF + \sum_{i=1}^{n} P_i Z_i\right) = f\left(P_{WP}, P_{VF}, P_i, PM\right) \tag{5}$$

$i = 1,\ldots,n$

where WP and VF represent waste paper and wood pulp (virgin fiber) consumption, respectively, while Z_i is a vector representing the consumption of all other input factors (such as energy, labor, capital etc.). P_{WP}, P_{VF} and P_i are the prices of waste paper, wood pulp and of the remaining inputs. Finally, we have a variable, *PM*, which reflects the fact that paper and board is not a homogenous product; paper and board instead consist of different qualities that have different waste paper coefficients. In our case *PM* is measured as the share of total paper and board production that constitutes newsprint, tissue and liner and fluting board. Since these products do not have to meet up to tough quality standards, intensive use of waste paper is likely to be relatively high in their production. Thus, in a country where the mix of paper and

[4] The model is based on Nestor (1991, 1992). For similar approaches to waste paper demand, see also Lee and Ma (2001) and Rehn (1995).

board products produced is in any way biased towards these three qualities, we would expect the utilization rate to be relatively high. The cost-minimizing factor demand equations can be obtained by applying Shephard's lemma and differentiating equation (5) with respect to input prices. We then obtain:

$$\frac{WP}{PB_{prod}} = g\left(P_{WP}, P_{VF}, P_i, PM\right) \tag{6a}$$

$$\frac{VF}{PB_{prod}} = h\left(P_{WP}, P_{VF}, P_i, PM\right) \tag{6b}$$

$$\frac{Z_i}{PB_{prod}} = k\left(P_{WP}, P_{VF}, P_i, PM\right) \tag{6c}$$

Equation (6a) is of particular importance to us since it expresses waste paper demand as a share of total paper and board output, the waste paper utilization rate. One advantage of this theoretical approach is thus that it recognizes explicitly that the utilization rate is tied closely to the demand side of the waste paper market.[5] According to the model this rate will be determined solely by the prices of all inputs and the share of "recycling-intensive" outputs. Naturally this is a simple model, and we do not claim that it in any way acknowledges all aspects of waste paper demand. Still, it represents an appropriate starting point for understanding changes in the utilization rate, and it is useful to compare the approach of this model with that employed by V/B.

3.2 The Approach of V/B

V/B estimate two regression models using the pooled data set mentioned above, one in which the recovery rate is the dependent variable and one in which the utilization rate serves as the dependent variable. For each of the two regression models two runs are conducted, one for developing countries and one for developed countries. In the following we concentrate solely on the waste paper utilization rate model.

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[5] Since waste paper recovery primarily is determined by supply behavior in the waste paper market the determinants of the recovery rate must be derived from another structural model. This questions the approach taken by V/B, who employ the same explanatory variables regardless of whether it is utilization or recovery patterns that are analyzed. Clearly, the recovery rate should be determined, for instance, by waste paper prices, recovery costs and the demand for waste management.

V/B use three different categories of independent variables; trade-related, demographic and market-related. As was discussed in the previous section, *trade-related* variables are of little interest as such when analyzing waste paper utilization. For example, in their analysis V/B suggest that "[t]he recycling industry will perform better if it can choose between domestic as well as foreign inputs," (p. 1728). For this reason, they argue, there will be a positive relationship between the "net-import dependency rate of waste paper" and the utilization rate. However, as was noted above imports are not an exogenous variable from a paper and board mill's perspective. This can be seen by noting that our model could easily be extended by introducing a distinction between imported and domestic input factors. In this case the mill's cost minimization decision would take into account the difference in price between imported and domestic waste paper and from this determine the relative size of each input source. Thus, it makes little sense to refer to the waste paper market as a "surplus market" (p. 1727); relative prices will determine waste paper demand! We therefore anticipate that the positive relationship between waste paper imports and utilization rate presented by V/B is likely to reflect the fact that some countries have access to low cost imports (and not the fact that the recycling industry has a "wider range of choice" simply because of the existence of high-volume imports). In the empirical section below we extend our model to also test for the impact of waste paper import prices on the utilization rate.

The second set of independent variables employed by V/B is referred to as *demographic* or *geographic* (even though they in essence are *economic*). The arguments put forward here are very much in line with the hypotheses that can be derived from our model and the results that are presented make much sense. The different geographic characteristics of each country will determine the relative price of energy, wood pulp as well as the cost of recovery and hence the use of waste paper.[6] One should note, however, that all the geographic/demographic variables introduced by V/B (e.g., population density, virgin resource endowment) are long-standing factors, which do not change much over time. They are thus particularly appropriate for understanding cross-country differences.

[6] It is a very plausible hypothesis that higher energy prices will favor waste paper use (instead of wood pulp), even though this relationship probably is much stronger for many metals. It may be questioned, though, whether a high import dependency of energy is a good proxy for high energy prices as is suggested by V/B. It should be noted that during the last three decades many countries have protected their energy industries with huge subsidies because they have been unable to compete with cheap imports (e.g., Radetzki, 1994; International Energy Agency, 1999). In addition, if high waste paper import dependency encourages waste paper use (as is also suggested by V/B), why would we experience the opposite for energy inputs?

The different *market-related* variables that are identified by V/B are all (perhaps with the exception of manufacturing wages) more volatile over time. Let us first consider the waste paper/wood pulp price ratio variable. By relying on average annual waste paper and wood pulp price data over 50 countries (brought together in a single diagram) V/B argue that there exists only meager support for the often-claimed notion that waste paper prices are more volatile than are wood pulp prices (e.g., Edgren and Moreland, 1989). However, by using average figures for a wide range of countries V/B essentially assume that the waste paper market is perfectly integrated globally. Even though international trade in waste paper is common, there exists little support for the assumption that domestic waste paper prices in all regions of the world move very closely together. For this reason one need instead to compare waste paper and wood pulp prices separately for each country. Figure 4 presents such a comparison for the U.S. case.

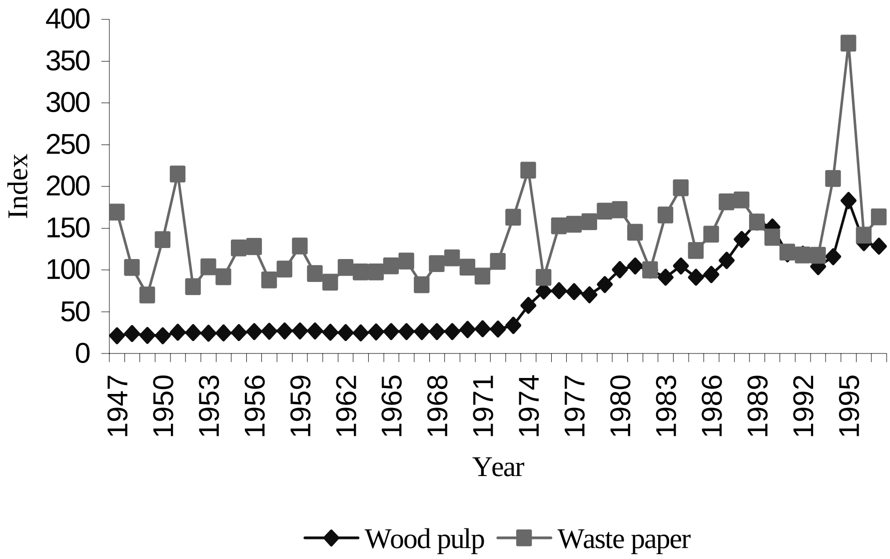

Figure 4. Annual waste paper and wood pulp price index for the USA, 1947-1998 (1982 = 100)
Source: Bureau of Labor Statistics (2002).

This figure is based on annual data and it shows that waste paper prices in the USA tend to be much more volatile over time than do the corresponding wood pulp prices.[7] In order to compare the degrees of variability for waste paper prices and wood pulp prices

[7] Of course, as also noted by V/B, these fluctuations come out even stronger when using data on a monthly basis.

respectively, we have calculated the coefficient of variation, *CV*, for each time series.[8] This analysis confirms that the variability in waste paper prices has been greater (*CV* = 71.1) than that for wood pulp (*CV* = 37.2). Another market-related variable used by V/B is the "growth rate of per capita consumption of paper". Clearly this variable will also fluctuate a lot over time as it is heavily related to the business cycle.

In sum, in contrast to the first part of their analysis some of V/B's choices of explanatory variables for the regression model are well in line with the simple theoretical framework outlined above. Most of the variables reflect the price (i.e., the relative availability) of the different input factors in the paper and board industry. However, some of these price-related variables (e.g., primary commodity endowment, population density) change very little over time while some (e.g., relative price of waste paper to wood pulp) fluctuate a lot even on annual basis. Why does this matter?

The decision to use waste paper as input for the production of paper and board is primarily determined by the rate of return of different production methods, which in turn are comparatively capital-intensive and have a long economic lifetime. Thus, the decision to invest in new production capacity will not depend only on current prices but also on broader measures of input factor availability (i.e., "long-run prices"). If the regression analysis aims at analyzing these long-run impacts it is best to rely solely on cross-country (or firm-specific) data, since these often exhibit a wide range of variation and tend to be the result of long-standing national policies. This is true in particular if the conditions at the cross-country observations have been stable for some time. For this reason the prevailing waste paper price will not be an appropriate long-run variable, since it tends to fluctuate much from one year to another. For example, the 1995 U.S. waste paper price (see Figure 4) will be a poor indication of the long-run availability of waste paper in the USA since this price spike was driven primarily by the presence of intense speculation in the U.S. market.[9] Thus, time series are more likely to reflect short-run responses, and especially so if there, as is thus the case with waste paper, have been substantial price variations during the period under study.[10] Clearly,

[8] The coefficient of variation expresses standard deviation as a percentage of the mean and is computed as follows: *CV* = (standard deviation/mean) x 100.

[9] See Ackerman and Gallagher (2002) for an analysis of the 1995 price spike in the U.S. waste paper market.

[10] These interpretations of the inferences that one can draw from the use of cross-country and time-series data, respectively, are standard practice in applied econometric work. See, for example, Griffin (1977) and Pindyck (1979). Empirical studies also confirm the notion that the elasticities derived from time-series data tend to be significantly lower than those resulting from cross-section studies (e.g., Atkinson and Manning, 1995).

the short-run responsiveness of waste paper demand to price changes will be much lower than that in the long-run.

Since V/B rely on both short- and long-run analysis it becomes hard to assess their results properly and in particular to compare the different responses. For example, V/B's (partial) reliance on time-series in combination with the use of annual waste paper prices is likely to explain their conlusion that "[p]aper recycling in developed countries is insensitive to developments in the price ratio of pulp and wastepaper," (p. 1731). However, the variable intended to capture the availability of virgin pulp ("primary commodity endowment"), which tends to be fairly stable over time but varies a lot over countries, comes out as an important determinant of waste paper utilization. These results create confusion. Should they really be interpreted as an indication that wood pulp availability is an important determinant of the waste paper utilization rate but that waste paper availability is not?

The short-run/long-run confusion is reinforced by the fact that V/B employ pooled data. While the use of either pure cross-section data or pure time-series facilitates the interpretation of the time dimension of the estimated relationships, the reliance on pooled data makes it harder to determine whether the estimates will reflect short- or long-run behavior (Stapleton, 1981). However, by introducing dummy variables it is possible to remove either (a) the within-country variation with the help of time-specific dummies; or (b) the between-country variation by employing country-specific dummies. In the former case it makes sense to interpret the results as long-run responses, while the latter method primarily captures short-run responses (Baltagi, 1995). Neither of these approaches is (as far as we can see) employed in V/B's analysis.

3.3 Our Approach

In this sub-section we present an alternative regression analysis of waste paper utilization rates. For at least two reasons, this new model represents an improvement of that presented by V/B. *First*, it is consistent with the theoretical framework outlined above. *Second*, it avoids the short-run/long-run confusion experienced in V/B's results and focuses instead solely on long-run impacts on waste paper use. For our purposes the waste paper utilization rate equation is expressed as:

$$\ln UR_{it} = \beta_0 + \beta_1 \ln PVF_{it} + \beta_2 \ln PM_{it} + \beta_3 \ln PWP_{it} + \beta_4 \ln RR_{it} + \varepsilon_{it} \qquad (7)$$

$i = 1,\ldots,N$; $t = 1,\ldots,T$

where UR_{it} is the utilization rate in country *i* and time period *t* defined as in equation (2), and ε_{it} is the error term. PVF_{it} denotes the availability (cheapness) of virgin fiber, and for our purposes it is measured as the growing stock of forests divided by the total population in country *i*. Thus, this variable is intended to measure the long-run relative availability of virgin fibers. Since virgin fibers are substitutes to waste paper in paper and board production we would expect the coefficient β_1 to be negative. The composition of paper and board production, PM_{it}, is measured by the share of total paper and board production (in tons) that constitutes newsprint, tissue and liner and fluting board. Increases in PM_{it} should lead to increases in the utilization rate, since intensive use of waste paper is particularly high in the production of these three paper and board qualities.

PWP_{it} and RR_{it} are both waste paper availability (price) indicators, where PWP_{it} is the share of production to consumption of paper and board in country *i* at time period *t*. As was suggested in section 2, if waste paper consumption is significantly greater than paper and board production the availability of waste paper is relatively high, while the demand for waste paper (in production) is relatively low. In this case the ratio PWP_{it} will be low and relative waste paper utilization will be high, i.e., the β_3 coefficient is expected to have a negative sign. RR_{it}, on the other hand, measures the *intensity* with which the above waste paper supplies are exploited. Naturally, if the recovery rate increases it will be easier to obtain waste paper for use in paper and board production. Hence, β_4 should have a positive sign.[11] As was noted above, we also estimate an extended model including the import price of waste paper (in US$ per metric ton) as an explanatory variable. Nevertheless, in line with the above discussion on short- and long-run impacts, we do not expect this variable to be a significant determinant of cross-country differences in waste paper utilization rates given the relatively high volatility of waste paper prices in combination with the capital intensive nature of the paper and board industry.

[11] It should be noted that our empirical specification of the model, equation (7), does not take into account the input prices of, for example, energy and labor. We therefore essentially assume that the underlying production function for paper and board is weakly separable in its aggregates, i.e., the mix of wood pulp and waste paper is assumed to be independent of the other inputs. We make this assumption since reliable prices for the other inputs are hard to obtain, especially for an international cross-section. In addition, they are not the focus of this investigation. Also due to data reasons we do not estimate the entire system of equations in (6a-c), using, for instance, seemingly unrelated regressions techniques.

Our data are drawn from a panel of 49 countries for the period 1990-1996.[12] In line with the approach of V/B we also divide the data set into one "high-income" country sample and one "middle-income" country sample. We follow the World Bank's income classification and classify "high-income" countries as those with a GDP per capita of more than USD 10 000 (1990 prices), while "middle-income" countries include countries with a GDP per capita between USD 4000 and USD 9999 (World Bank, 2001).[13]

In order to employ the regression model in equation (7) empirically we need to specify the choice of econometric technique and the stochastic framework. In this study we employ a one-way error component model so that the error term (ε_{it}) for the utilization rate equation can be decomposed into two parts:

$$\varepsilon_{it} = \lambda_t + v_{it} \tag{8}$$

where λ_t denotes the unobservable time effect and v_{it} is the remainder stochastic disturbance term. The term λ_t is country-invariant and accounts for any time specific effects that are not included in the regression. A common approach is to assume that these effects are fixed over countries for a given time period, and then eliminate the time-specific disturbance component by introducing dummy variables for each time period. This is known as the fixed effects model or the least squares dummy variable model, and it is the most appropriate approach when analyzing solely long-run (cross-country) impacts.

An *F*-test of the null hypothesis that $\lambda_1 = \lambda_2 = ... = \lambda_{T-1} = 0$ was performed, and it suggested that the null hypothesis of common intercept terms could not be rejected. The reason for this is that in our data set the variation between countries greatly exceeds the within-country variation (i.e., the variation over time). Our data include variables such as forest endowment, paper and board production and consumption patterns, paper product mix etc., which certainly vary much more between countries than within countries. For the above reasons, the results presented in this paper will be based on ordinary least squares with common intercept as well as slope coefficients (Baltagi and Griffin, 1984). Again, the strong

[12] Complete data sources and definitions can be found in Berglund and Söderholm (2002), which is available from the authors on request.

[13] Data for the poorest countries are generally very unreliable and hard to come by. A list of all included countries is available in Berglund and Söderholm (2002).

dominance of between-country variation over within-country variation in our data set allows us to interpret the estimated responses as long-run.

By performing a Hausman test we could reject the null hypothesis that the logarithm of RR_{it} is uncorrelated with the remaining error term (ν_{it}), and hence that $\ln RR_{it}$ is an exogenous variable in the utilization rate equation. This problem of simultaneity was solved by the use of instrumental variables. We regressed the logarithm of RR_{it} on a set of variables considered exogenous (the logarithms of GDP per capita, population density, urbanization rate, virgin forest availability (*PVF*), waste paper availability (*PWP*), and paper product mix (*PM*)), and employed the fitted values from these first regressions as instruments in place of $\ln RR_{it}$ in equation (7). This means that in contrast to the approach taken by V/B, the population density and the level of income (in our case GDP per capita and in V/B's case the manufacturing wage) will influence the utilization rate only indirectly via the recovery rate.[14]

Table 1 presents the parameter estimates for the coefficients in the waste paper utilization rate equation.[15] We present the results for all three samples ("total", "high-income", and "middle-income") but devote most of our attention to the two sub-samples.[16] Our two indicators of waste paper availability turn out as the most important determinants of inter-country differences in utilization rates, and the coefficients representing these impacts are also statistically significant. Thus, countries endowed with abundant waste paper supplies tend to have high utilization rates. This result strongly contradicts V/B's conclusion that relative waste paper utilization is unresponsive to changes in the waste paper price. Again, we believe that their finding is due to the short-run nature of their analysis, while we focus explicitly on long-run responses.

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[14] The regression models were tested for the possible existence of heteroscedasticity using White's (1980) test, and in all cases the null hypothesis of homoscedasticity was rejected. The presented standard errors for the OLS models have therefore been calculated from the heteroscedastic-consistent variance-covariance matrix (MacKinnon and White, 1985).

[15] We also performed separate regressions for the recovery rate, using waste paper prices, population density, urbanization and GDP per capita as explanatory variables. Our results support those of V/B. The income effect tends to be positive (and higher in developed countries compared to developing countries) and increases in the population density and urbanization rate induce increases in the recovery rate. See Berglund and Söderholm (2002) for details.

[16] An *F*-test rejected the null hypothesis that the two sub-samples could be combined into one single regression model.

Table 1: OLS Estimates for the Utilization Rate Equation*

Variables	*Total sample*	*High-income sample*	*Middle-income sample*
Constant (β_0)	4.774	4.233	4.532
	(10.816)	(3.480)	(8.732)
PVF (β_1)	0.005	-0.048	0.002
	(0.253)	(-1.815)	(0.068)
PM (β_2)	0.041	0.227	0.013
	(0.540)	(0.976)	(0.190)
PWP (β_3)	-0.821	-0.703	-0.801
	(-12.089)	(-8.560)	(-4.684)
RR (β_4)	0.687	0.502	0.798
	(5.780)	(2.982)	(2.707)
R-square (adj)	0.54	0.69	0.27

* t-statistics are given in parentheses.

Furthermore, according to our model countries in which the share of "intensively-recyclable" paper and board products is high, will also experience high utilization rates. However, this effect is insignificant from both economic and statistical points of view. The hypothesis that in countries blessed with substantial forestry resources the utilization rate will be high only gains partial support. Only for the "high-income" sample do we find this expected negative relationship. This may be due to the fact that many developing countries are primarily endowed with tropical forests, which often have a lower productivity in supplying wood fiber and also shorter fiber lengths that make them less appropriate for pulp and paper production (Van Beukering and Bouman, 2003).

Apart from the impact of virgin fiber availability, the differences between the two sub-samples are small. It should, however, be noted that increases in waste paper availability (whether measured by *RR* or *PWP*) have a greater impact on waste paper utilization in "middle-income" countries than in "high-income" countries. This suggests that the input flexibility is higher in the former countries, perhaps due to lower quality requirements and less capital-intensive production facilities.

Finally, Table 2 shows the parameter estimation results for the extended model including the waste paper import price as an independent variable. Due to limited data availability these estimations are based on a more limited sample of 39 countries (21 high-income countries and 18 middle-income countries) over the time period 1990-1996. As expected the coefficient representing the import price variable (β_5) is not significantly different from zero in any of the model runs, and it adds thus little to our understanding of cross-country differences in waste paper utilization. The parameter estimations for our original waste paper availability indicators, PWP_{it} and RR_{it}, appear robust, however, and they still indicate that these two proxies appear to explain a large degree of the variation in utilization rates. This does of course not imply that waste paper import prices are unimportant as such; it simply illustrates that short-run market-determined prices, given their high volatility,[17] are not a major decision variable for paper and board companies when deciding to invest in waste paper using capacity. We therefore need more stable proxies for 'long-run' prices if we are to investigate differences across countries.[18]

Table 2. OLS Estimates for the Utilization Rate Equation Including Import Prices*

Variables	*Total sample*	*High-income sample*	*Middle-income sample*
Constant (β_0)	1.053	1.356	1.767
	(2.950)	(3.042)	(2.061)
PVF (β_1)	0.013	-0.002	0.021
	(0.582)	(-0.059)	(0.436)
PM (β_2)	0.487	0.485	0.020
	(3.468)	(2.787)	(0.050)
PWP (β_3)	-0.760	-0.747	-0.670
	(-12.179)	(-13.013)	(-2.572)
RR (β_4)	0.711	0.575	0.623
	(5.965)	(4.203)	(1.709)
PWP-IMP (β_5)	0.008	-0.029	0.066
	(0.219)	(-0.468)	(1.481)
R-square (adj)	0.63	0.76	0.22

* t-statistics are given in parentheses.

[17] This short-run volatility is also apparent in our data set. Appendix A compares the development of waste paper import prices in six countries with the (more stable) development of the ratio between paper and board production and consumption.

[18] By using average price data over a longer time period (and thereby averaging out any short-term influences) for a large number of countries we would also be able to examine the long-run impact of waste paper prices. However, for data availability reasons this has not been possible, especially given our purpose of analyzing two groups of countries separately.

Table 2 also shows that the impact of the paper product mix on utilization is much more significant (both from an economic and a statistical point of view) than was the case for our previous sample. Apart from this the results are very similar to those presented above (see Table 1).

4. Concluding Comments

This comment has stressed the importance of waste paper availability as the main determinant of the relative utilization of waste paper. Empirical evidence has been presented in support of this notion. Our overall approach and results are in some contrast with the approach of V/B, especially since we reject the idea that waste paper trade is the main cause of observed differences between countries' utilization (and recovery) rates. In addition, our regression model involves an explicit focus on inter-country differences rather than a mix of "within-country" and "between-country" effects.

In spite of the above differences the policy recommendations that follow from our analysis are similar to those suggested by V/B. We note in particular that for many countries utilization (and indeed recovery) rates represent important policy targets. However, since differences in waste paper utilization are to a great extent due to long-standing economic characteristics about which very little can, or perhaps even should, be done, the degree of flexibility in affecting the utilization rate may be limited. For example, given the importance of waste paper availability in determining the relative utilization of waste paper, a strict utilization rate goal may be costly to enforce as it conflicts with prevailing paper and board patterns. If a country that is a major net exporter of paper and board faces a recycled content goal, it will have to import waste paper in order to comply since its domestic supplies are too scarce. It is not clear that this behavior is socially desirable, neither from an environmental nor economical point of view. Clearly this also speaks against the implementation of harmonized utilization targets across countries.[19]

Acknowledgements

Financial support from the Jan Wallander and Tom Hedelius Foundation, the Kempe Foundations, and the Marcus and Amalia Wallenberg Foundation is gratefully acknowledged, as are valuable comments on the work from Mikael Bask, Jim Griffin, Stefan Hellmer, Mats

[19] These conclusions are also drawn in Huhtala and Samakovlis (1999), who analyze the impacts of harmonized recycled content standards for waste paper in Europe.

Nilsson, David Pearce, Marian Radetzki, John Tilton and two anonymous referees. The views expressed in this paper and any remaining errors are, however, solely those of the authors.

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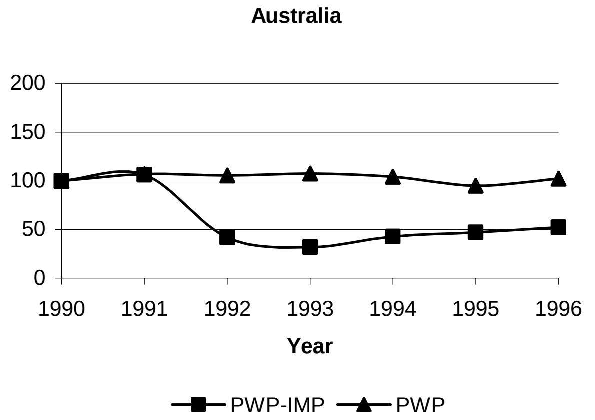
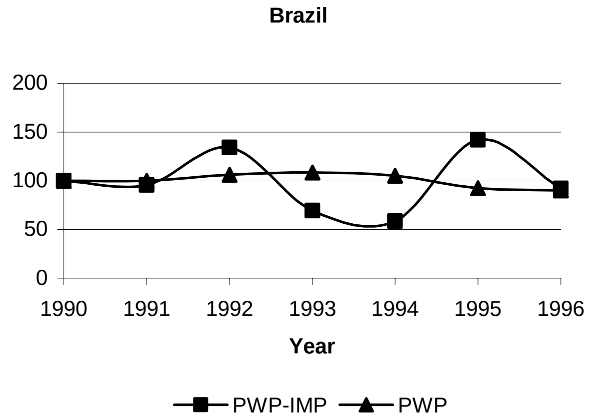
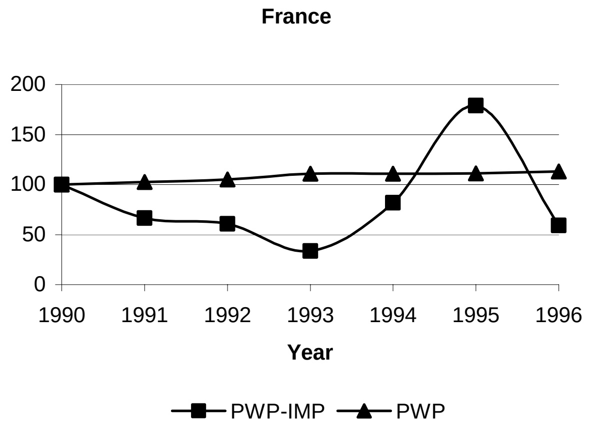
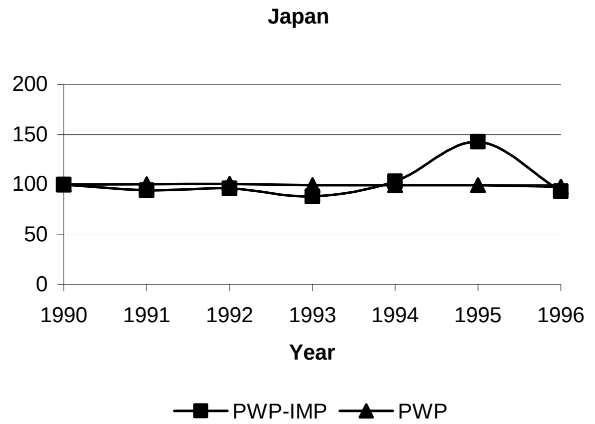
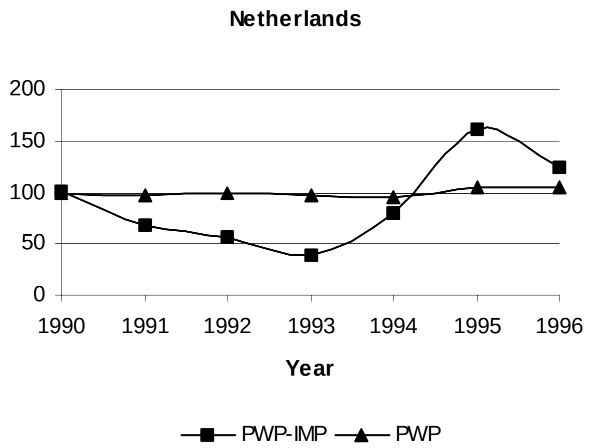
*Appendix A: A Comparison of Waste Paper Price Volatility in Selected Countries**

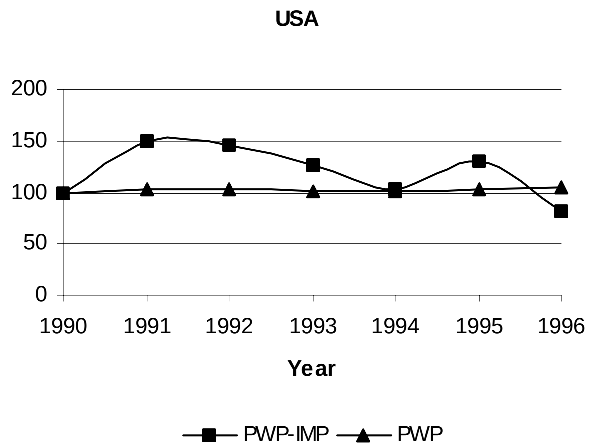












* *PWP* shows the ratio between paper and board production and paper and board consumption (%), while *PWP-IMP* is the import price of waste paper (in US$ per metric ton).

Sources: Pulp & Paper International (1998) and FAO (2001).

Spatial Cost Efficiency in Waste Paper Handling: The Case of Corrugated Board in Sweden

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Abstract

This paper analyzes the spatial cost efficiency of the Swedish legislation regarding waste disposal handling. We focus on the case of corrugated board and recognize that the different counties in Sweden possess different economic prerequisites in terms of waste paper recovery and utilization potential. We employ data for six corrugated board mills and 20 counties and a non-linear programming model to identify the least cost strategy for reaching the politically specified recycling target of a 65 percent recovery rate for corrugated board. That is, the total costs of recovering a minimum of 65 percent in each county are calculated and compared with the case when the country as a whole recovers 65 percent of all old corrugated board is collected but there exist no uniform target for each county. The conclusion is that from an efficiency point of view the recovery efforts should be concentrated to the highly populated and urbanized counties, and not be uniformly divided throughout the country. In the base case the results suggest that the cost efficient county-specific recovery rates should range from 51 percent to 72 percent.

Keywords: waste paper, corrugated board, recycling, cost effectiveness, transport costs, collection costs, Sweden, non-linear programming

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1. Introduction

For a long time, legislation has been the central tool with which principles of environmental policy have been transformed into practical measures. In Sweden, state and municipal authorities have undertaken numerous measures in the course of the 1990s, some of which have been quite elaborate, to promote recycling of raw materials. The general motivations have been that recycling is suppressed by a variety of market failures, and that its expansion is beneficial to society at large, primarily because it is environmentally benign and it contributes to the conservation of natural resources. Since 1994/95 the producer responsibility ordinance (PRO) has governed the collection and recycling of waste paper in Sweden. This responsibility is regulated for newspaper and similar paper qualities by the ordinance SFS 1994:1205, and for paper packaging materials by SFS 1997:185.[1]

The PRO *de jure* requires separation of paper from other waste products and that collected waste paper must proceed to material recycling, i.e., waste paper should be used in paper and board production and not for energy recovery or landfill (Swedish Environmental Protection Agency, 1999). However, *de facto* Swedish law also requires that waste paper should be collected throughout the country with no special account taken of regional differences such as transport network, distances to paper mills, and other demographic and geographic conditions. In other words, the law does not permit differentiated recovery targets across regions. Waste paper recovery in sparsely populated rural areas can therefore become prohibitively expensive (FAO, 1996). Transport is a major cost component in the production of pulp and paper and has significant negative environmental impacts. Still, little attention has been devoted to this aspect of the paper cycle.

There exists few previous economic research efforts on the PRO for paper and paper packaging materials in Sweden. However, two studies that do evaluate the PRO based on Swedish conditions are Marklund and Samakovlis (2001) and Radetzki (2000). Marklund and Samakovlis (2001) question the priority of material recycling over energy recovery as stated in the PRO by calculating and comparing shadow prices for waste paper for the Swedish paper and heating industries. They conclude that the welfare in Swedish society may improve if the PRO considered incineration as a possible alternative. Radetzki (2000) compares the net cost to society from recycling with the net costs from incineration and landfilling, and concludes that the PRO is very cost inefficient. Radetzki acknowledges that his empirical

[1] In 1997 the current law regarding paper packaging materials (SFS 1997:185) replaced the old one (SFS 1994:1235).

material is partly drawn from foreign studies, which makes it difficult to present entirely valid conclusions for Sweden. He therefore urge for additional studies that identify cost effective waste management measures and that use economic analysis as a tool to guide future policy making. Other studies have paid a lot of attention to life cycle analysis (e.g., Finnveden and Ekvall, 1998; Finnveden et al., 2000). Life cycle analysis enables all the impacts of a process or product on the environment to be evaluated. However, most life cycle analyses either ignore transport altogether or, when comparing different scenarios, assume that there will be little changes in transport modes or distances traveled. Although this notion might be valid in certain situations, transportation costs may turn out to be very important when analyzing the cost efficiency of the waste paper collection process. This is likely to be particularly valid for a sparsely populated country like Sweden. By identifying the least-cost means of meeting a particular standard or policy goal and using this cost as a benchmark case, we can estimate how much costs can be expected to increase from this minimum level if policies that are not cost effective are implemented. The above concerns provide the rationale for this paper, which aims at enriching the debate about policy failures with respect to waste paper handling.

The overall purpose of the paper is to evaluate the economic impacts of the Swedish policy formulation on waste paper recovery. A non-linear programming model is applied to identify the least cost strategy for reaching politically specified recycling targets in general and the 65 percent recovery rate for corrugated board (stipulated in SFS 1997:185) in particular. Specifically, the model is employed to calculate the total costs of recovering a minimum of 65 percent old corrugated board in each Swedish county, and then compare this cost with the case when the country as a whole recovers 65 percent of all corrugated board generated cost effectively. That is, for a given recovery target we identify the optimal amount of waste paper that should be recovered in each county in order to minimize the total costs of waste paper handling. The costs analyzed in the model include the recovery costs within each county and the additional transport costs incurred by bringing the board to the paper mills (which often are situated in other counties).

The present paper differs from earlier research efforts in at least two ways. *First*, it presents and applies a methodological framework in which cost-effective recovery rates for different spatially distributed regions can be calculated. *Second*, the model employed accounts explicitly for the fact that different counties/regions possess different economic prerequisites in terms of waste paper recovery and utilization potential.

Four important limitations of the present work need to be indicated before proceeding. *First*, the focus is solely on the spatial cost efficiency of Swedish waste paper handling, and we do not attempt to assess whether the present level of recovery itself is socially optimal. For instance, the model used ignores costs imposed on households as well as the benefit side of recycling, and it is thus an inappropriate tool for the economic assessment of different waste management alternatives (e.g., recycling versus incineration). *Second*, although imports of used corrugated board are acknowledged as an input alternative, they are not dealt with here since the PRO states that *domestic* waste paper must be used in paper production and not for energy recovery. We will revert to this issue in more detail in section 2.2. *Third*, although the emphasis of the analysis is on waste paper in general, corrugated board will be in special focus. The reason for this is that the corrugated board mills are located throughout the entire country, as opposed to e.g., newsprint production, which is geographically concentrated in the south of Sweden. Since paper and board consumption (and hence the supply of old corrugated board) is fairly concentrated in the southern part of the country, spatial cost efficiency should be a real concern for this product. *Fourth*, since no data on the marginal costs are readily available, no actual or true inefficiencies can be identified from the analysis but we can nevertheless provide some plausible and approximate effects of the uniform policy in force. By drawing on a number of empirical observations and a few assumptions about the cost structure we are nevertheless able to analyze the *relative* cost differences between different regions and scenarios (see section 3.2 for details). There are obviously no easy answers to the chosen research question and this paper does not provide a blueprint for a sustainable waste paper handling policy; further research will be necessary.

The paper proceeds as follows. Section 2 discusses how waste disposal targets tend to be formulated in Sweden and elsewhere, and outlines the theoretical underpinnings for how waste recovery policies should be analyzed if cost effectiveness is an important policy goal. Section 3 presents some important input data and specifications in the model employed. In section 4 we present the non-linear programming (NLP) model, which is used to analyze the impact of the producer responsibility ordinance in Swedish waste paper handling. The results from the model simulations are presented and discussed in section 5, while section 6 provides some concluding remarks and implications for policy.

2. Policy Approaches and Theoretical Underpinnings for Policy Analysis in Waste Paper Handling

2.1 Quantitative Requirements and Legislation in Recycling Policy

Policy makers in Sweden and elsewhere have often chosen to set waste disposal targets for various materials, usually with no attempt to determine the socially optimal level of the targets for the materials in question (e.g., Radetzki, 2000). The obligations stipulated in the EU directive of 1994 on packaging and packaging waste (94/62/EC) are realized in the Swedish national targets and the rules drawn up by the Council of State. The aim of these decisions and rules is to reduce the environmental impacts caused by packaging and packaging waste in Sweden. It is however difficult to determine to what extent the policy targets are based on empirical facts about of the environmental impacts of recycling and to what extent more or less arbitrary political concerns have dominated the decision process. The latter concerns are often rooted in what is politically achievable rather than what is environmentally sustainable and/or socially optimal from an economic point of view.

Recycling activities can be measured in various ways. The most common measure of the extent of recycling, in the case of paper and board, is the *recovery rate*. It is defined as the ratio of waste paper recycled to total paper consumption. The recovery rate thus measures the success with which one is able to recover waste paper from the waste stream. Table 1 provides some selected examples of waste management objectives in a number of countries.

Table 1: Selected Waste Management Objectives in Different Countries

Country	Objectives	Time table
France	Domestic packaging: 75% recovered.	1992-2002
Germany	Packaging: between 60% and 70% recycling. Strict hierarchy of treatment techniques.	1991-1998
Greece	Packaging recycling specific objectives of the 94/62/EC directive: 25% recycling.	1994-2000
Italy	Since the 1997 decree, between 50% and 65% recovery for the whole of packaging waste (between 25% and 45% recycling) and 35% of separate collection.	1997-2003
Netherlands	Packaging waste volume generated in 2000 should not be bigger than in 1986. Minimum of 60% recycling of packaging waste. No specific goals of municipal waste.	2000
Sweden	75% newsprint production should be recovered and corrugated board to 65% (with a minimum of 40% material recovery).	2001
USA	35% recycling rate.	2005

Sources: Buclet and Godard (2000); SFS 1994:1205 and SFS 1997:185; and EPA 530-N-96-008.

As can be noted, each country formulates domestic recycling goals for different fractions of the municipal waste flow, including paper, and for many countries the recovery rate constitutes an important policy variable.

Waste paper used by the paper and board industry is divided into separate categories, or grades, and different paper and board products require different grades of waste paper. Throughout Western economies in general and Sweden in particular the trend for recycling paper and board is on the increase. Figure 1 shows the recovery rate for paper and paperboard in Sweden between 1940 and 1997.

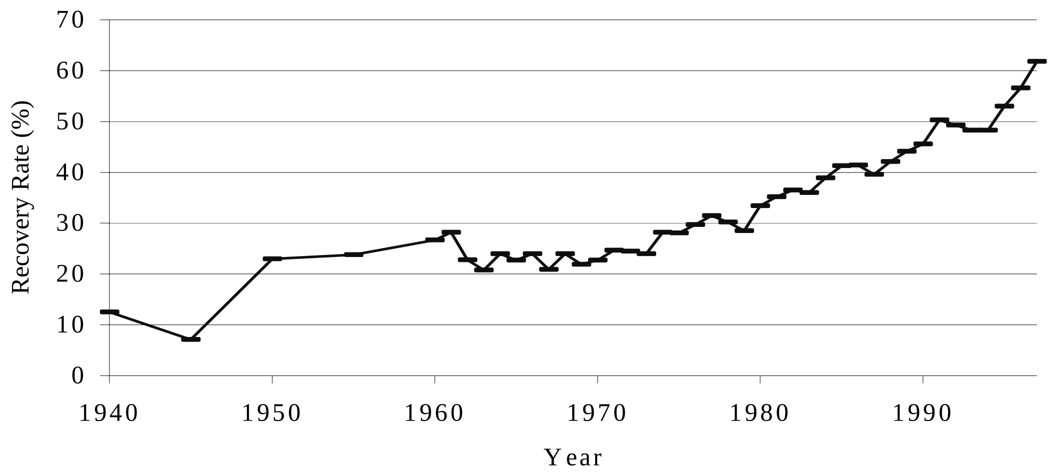

Figure 1: Paper and Paperboard Recovery Rates (in percent) in Sweden, 1940-1997
Source: Swedish Forest Industries Federation (2001).

In Sweden, the recovery rate has increased over time and during the 1980s and 1990s it more than doubled, to reach a level above 60 percent. In large part this has been a result of the policies in force. Two policy events in particular have contributed to the increase. *First*, in 1975 the Swedish government gave an advance warning of the producer responsibility in its preparatory work to introduce legislation regarding solid waste management (Prop. 1975:32). *Second*, the laws (of 1994 and 1997) regarding the producer responsibility for paper anchored this earlier premonitory. It is notable that in the short period after the 1994 legislation, the recovery rate increased by more than ten percentage points. The extent to which this is due to the legislation itself or just a rising trend prompted by changing market conditions is hard to assess without further information. Still, it is plausible to assume that the legislation has had a significant impact.

Policy differences between countries are intuitively apt since different countries are endowed with various demographic, economic and geographic features.[2] Within specific countries policy ineffectiveness also tends to be an issue of concern. For instance, it is questionable that some countries set equal recycling targets across different *fractions* of the materials. In the Netherlands, for example, the legislation states that 60 percent of the packaging waste is to be recovered for the purpose of recycling. Unless the recycling of different paper packaging types has equal marginal costs, setting one recycling policy goal for all fractions of paper will result in an ineffective outcome. Therefore, similar recycling rates for all fractions of waste paper would only in rare cases emerge as a cost effective policy tool. In many cases the recycling standards appear to have been established with very little grounding in economic theory or calculation (Radetzki, 2000). Furthermore, and as mentioned above, the Swedish laws *de facto* require that specific fractions of waste paper should be collected uniformly throughout the country, e.g., 65 percent of corrugated board is to be recovered in all counties. Still, unless these different counties have equal demographic, economic and geographic features, this policy is unlikely to be cost efficient. This latter question is the focus of this paper.

2.2 Marginal Cost Analysis as a Tool for Policy Evaluation

According to economic theory estimates of the marginal costs of recovery have to be employed in an analysis that aims at evaluating the impact of a policy that stipulates specific recovery targets.

Collection of waste paper is conducted in two ways. There is a direct collection system from companies while the collection of households' waste is made through recovery stations along with the collection of board. Since waste paper supply ultimately stems from recent paper and board consumption, the cost of waste paper recovery will be affected by demographic and geographic circumstances. Depending on population density, intensity of commercial activity etc., the costs of recovery will differ by region. Transportation costs will be low as long as the geographical area is limited in size and the economic activity is dense. The costs associated with transportation are, hence, likely to be highly variable *across* regions. Other things equal, actions to separate and collect waste paper will be more viable in

[2] Huhtala and Samakovlis (1999) support this notion empirically and conclude that uniform utilization rates within the European Union may be very costly to society. Berglund et al. (2002) present similar conclusions for both utilization and recovery rates.

regions that are densely populated and/or in which people live clustered in highly urbanized areas. Hence, the marginal cost curves of waste paper recovery (as well as those of other waste handling processes) are bound to be upward sloping due to large transport distances, and the increasingly inferior quality of the waste for the purpose at hand.

Marginal cost curves help us determine the cost effective division of the waste paper flows between alternative regions in the following way. Let us assume that waste paper recovery in the two regions, A and B, involve marginal costs as depicted in Figure 2, and that we wish to divide 65 percent of all waste paper flows between the two regions so as to minimize the cost to society as a whole. That is, our aggregate recovery goal is 65 percent. It is important to note that this "goal" of 65 percent does not necessarily represent socially optimal level of recycling activities. Still, if we assume that this is the level of waste paper to be recovered, cost-effectiveness analysis can be employed.

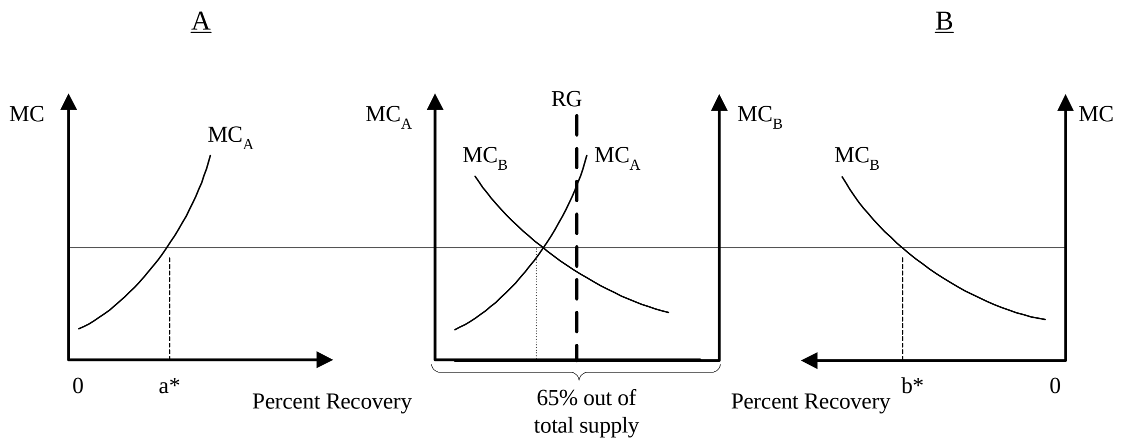

Figure 2: Cost-effectiveness Analysis in a Two Region Waste Recovery Scheme

Figure 2 demonstrates that a* percent should be recovered in region A and b* percent should be recovered in B, this since the marginal costs of recovery are the same at this level, and the two sum up to the recycling goal (65 percent), as illustrated in the middle graph. Any other subdivision will carry higher total costs, and will thus be sub-optimal. This simple theoretical framework can also help us to evaluate specific policy options, e.g., that 65 percent (which is more than a*) should be recovered in A, and 65 percent (less than b*) be recovered in B, here represented by RG in the middle graph. By calculating the difference between the higher costs of the additional recovery in region A and the lower costs of the reduced recovery in region B, we can assess the inefficiency cost to society incurred by implementing this policy rather than the cost-effective solution. The above theoretical considerations provide the basis of the NLP model presented in section 4. First, however, some important inputs into the model will be outlined.

3. Model Specifications

This section presents the supply and demand of old corrugated board in Sweden, and discusses how the marginal costs of recovery will be specified in our model. In doing this we make heavy use of the marginal cost analysis discussed in section 2.

3.1 Supply and Demand of Used Corrugated Board in Sweden

Figure 3 shows the locations of the seat of the provincial government cities in each Swedish county from where we assume the supply of used corrugated board is shipped,[3] and the major demand centers, i.e., the paper mills in Sweden that use corrugated board as main type of waste paper input in new production.

In order to calculate the level of old corrugated board recovery, information on the total quantity of packaging supplied to the Swedish market during the year has to be available. That is, if one wants to calculate the potential supply within a country one uses the measure of total corrugated board production plus imports minus exports for a given year. Given that the longevity of the product is short this will be a good approximation of total supply in that same year. However, although total supply in a country is relatively straightforward to calculate, the potential supply in each county is much harder to assess since no data are readily available of flows *within* Sweden (Georgsson, 2002). To overcome this problem, we assume that the consumption of corrugated board (and thus the total supply of used corrugated board) throughout the country is directly related to population. That is, it is assumed that regardless of region each person will on average consume the same amount of corrugated board. In 2001 the total supply of used corrugated board in Sweden was 411000 metric tons (Returwell, 2002). However, since the county of Gotland is exempt from the analysis 408339 metric tons will be employed as an estimate of the total supply in the analysis. Using population figures as weights the supply in each county will be divided as outlined in Table 2.

[3] The county of Gotland is exempt from the analysis due to its relatively small population and its geographical location (an island off east coast with no road connections with the mainland).

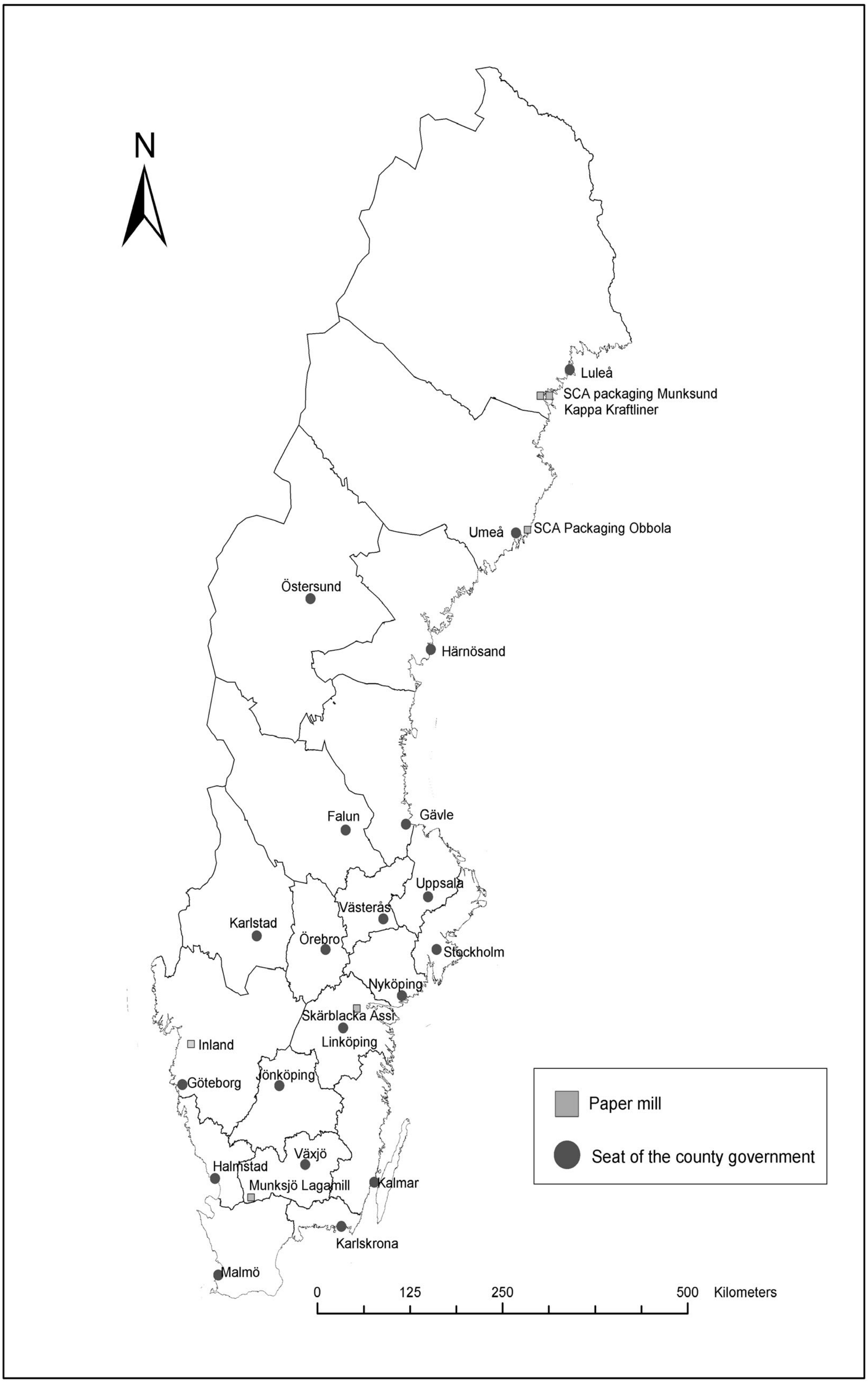

Figure 3: Supply and Demand Centers for Corrugated Board in Sweden

Table 2: Swedish Supply of Used Corrugated Board by County (metric tons)

County	Potential Supply of Waste Paper
Stockholm	83642
Uppsala	13562
Södermanland	11868
Östergötland	19077
Jönköping	15179
Kronoberg	8216
Kalmar	10969
Blekinge	6986
Skåne	52122
Halland	12687
Västra Götaland	69048
Värmland	12829
Örebro	12700
Västmanland	11915
Dalarna	13013
Gävleborg	13020
Västernorrland	11563
Jämtland	6062
Västerbotten	11906
Norrbotten	11971

Source: Returwell (2002).

Table 3 shows the paper and board mills in Sweden that use corrugated board as the main type of waste paper input in new production and the output they produce. The three largest paperboard mills, the two from SCA and Kappa, are all located in the northern part of Sweden, whereas the three smallest are located in the south (see also Figure 3).

Table 3: Paper Mills in Sweden that Use Corrugated Board as the Main Type of Waste Paper Input in New Production

Paper Mill	Waste Paper Capacity in 1999 (metric tons)	Main Output
SCA Packaging, Obbola	230 000	Kraftliner for Corrugated board
Kappa Kraftliner	112 000	Kraftliner for Corrugated board
SCA Packaging, Munksund	102 000	Kraftliner for Corrugated board
Munksjö Lagamill	92 000	Liner and fluting for Corrugated board
Inland	48 000	Wallboard
Skärblacka (AssiDomän)	32 000	Fluting for Corrugated board

Source: Swedish Forest Industries Federation (2001).

The waste paper capacities in Table 3 equal the demand restrictions used in the NLP model. However, since significant quantities of paper produced are exported every year, domestic supply of waste paper is at shortage. In order to meet the demand, the industry imports substantial amounts of used corrugated board from abroad. A profit-maximizing firm will choose the most inexpensive input in their production, i.e., the mill's cost minimization decision would take into account the differences in prices between imported and domestic waste paper and from this determine the optimal demand for each input source. However, since the PRO explicitly states that collected waste paper must go to material recycling and domestic paper must be used in paper production, and since this study analyzes efficiency *within* the prevalent domestic system, we will keep import out of our analysis by using the capacities only as upper constraints on domestic demand.

3.2 Marginal Cost Specification in the Model

All economic activities take place within geographical space. In our case, the costs of recycling, divided into collection costs (within a specific county) and transport costs (between counties) will differ across different regions. It is important to note that to a large extent collection costs in each county comprise of costs for transports (that is why sparsely populated areas have higher collection costs). Thus, a major component of marginal collection costs will be transport-related. Still, in the following when we use the term transport costs we refer solely to the costs of delivering board from supply centers to the demand points in other counties after collection have occurred.

Collection Costs within Counties

As was noted above the marginal costs of recovery (or collection) will depend on the relative size of the waste stream. Hence, the costs of recovery are expected to be relatively low in small but densely populated regions. Berglund and Söderholm (2002) present evidence that supports this notion and find that demographic features such as population density is a major determinant of the costs of collection and hence of recovery rates for a large sample of countries. There is no reason for why this would not be true *within* a country as well. For this reason, population density is used as the variable that determines the positions of the marginal cost functions in the different counties. Table 4 shows figures of population density in each county. They range from 2.61 people per square kilometer to 277.87 people per square kilometer. Hence, population density differs by a multiple of 100. In practice, however, the costs of collection between the high and low cost regions are not likely to differ by this much.

Therefore, the present study uses the logarithm of the population density as the main determinant of marginal recovery costs.

Table 4: Population and Population Density in the Counties of Sweden

County	**Population**	**Population density (people per square kilometer)**
Stockholm	1 803 377	277.87
Uppsala	292 415	41.84
Södermanland	255 890	42.21
Östergötland	411 320	38.94
Jönköping	327 266	31.24
Kronoberg	177 149	20.94
Kalmar	236 501	21.17
Blekinge	150 625	51.21
Skåne	1 123 786	101.91
Halland	273 537	50.15
Västra Götaland	1 488 709	62.18
Värmland	276 600	15.73
Örebro	273 822	32.15
Västmanland	256 901	40.77
Dalarna	280 575	9.95
Gävleborg	280 717	15.43
Västernorrland	249 299	11.5
Jämtland	130 705	2.64
Västerbotten	256 710	4.63
Norrbotten	258 094	2.61

Source: Statistics Sweden (2002).

By following an approach of distance costs developed by Samuelson (1952) and Mundell (1957), waste paper collection costs are here modelled in terms of ”iceberg” transport costs. The iceberg analogy stems from the idea that transport costs involved in towing an iceberg can be understood as causing an iceberg to melt away during the process. The costs of overcoming distance consequently decay the good being shipped thereby ensuring only a fraction of the good to be delivered (McCann, 2001). In our case, the good (waste paper) does not melt but its costs of collection shows similar exponential form due to widened “catchments areas”. Following the above, this study assumes that the total collection costs in each county, C_i, can be expressed as:

$$C_i = \int 2 e^{\alpha \left[\sum_{j=1}^{m} (q_{ij} / Q_i) * 100 \right]} \left(\frac{1}{\beta_i} \right) dq \qquad (1)$$

where the term $2e^{\alpha\left[\sum_{j=1}^{m}(q_{ij}/Q_i)*100\right]}$ represents the non-linearity of the marginal cost of collection, α is the slope parameter of the marginal cost curve, and $(q_{ij}/Q_i)*100$ expresses the recovery rate in percent in each county. The term q_{ij} represent the quantity of board that is recovered from county i and shipped to mill j. This implies that for a given county i the total volume of waste paper delivered to mills cannot exceed the total amount of paper and board consumption in that county (Q_i). That is, the amount recovered in each county cannot exceed the supply in each county. Furthermore, we do not know the "true" marginal costs of recovery but we know something about the relative difference between transport and collection costs. Feinberg (2002) reports that in Sweden the costs of collection roughly adds up to SEK 240 per metric ton, while the cost of transporting old corrugated board costs approximately SEK 60 per 100 kilometers. We used this knowledge to calibrate our marginal costs by setting the collection costs (measured in unit/tons) four times as large as the transport costs (measured in unit/tons). In the general setup we therefore assumed the cost of collection per ton to be four times the cost of transporting old corrugated board.[4] Given this, our slope parameter, α, is therefore assumed to be 0.2. Furthermore, since the marginal cost is assumed to differ between counties due to different cost factors such as population density, we need a "cost coefficient" that enables us to include these differences in the analysis. To do this we use β_i and it is calculated as the logarithm of the population density in county i as outlined above. The total costs, C_i, are thus obtained by the integral of all the marginal costs. Figure 4 shows the implied marginal cost curves of collection for the two counties Skåne and Stockholm.

Figure 4 illustrates the notion that the marginal costs are fairly low as long as the quantity recovered is small but increases substantially as the recovery rates become higher (and approach the total supply in each county). This is intuitive; as long as collection (and implicitly transport) is limited to geographical areas with dense economic activity the costs will be small, but as the collection area is widened to incorporate distant regions in each county the marginal costs of collection will increase substantially. It is also important to note that although it appears as if the marginal costs are equal in both regions up to about 40 thousand tons, the scaling in Figure 4 deceives us. Clearly, there are cost differences even at low recovery activities but the most significant appear only at higher levels of recovery. This is also intuitive; as long as we only collect small fractions we can do this in urban areas that

[4] We calibrated the model using the *de facto* PRO recovery rate requirements of 65 percent in each individual county since this roughly corresponds to the existing situation in the respective counties.

are present in all counties, but as we intensify the recycling activities, counties with denser populations will experience relatively lower collection costs than the more sparsely populated areas.

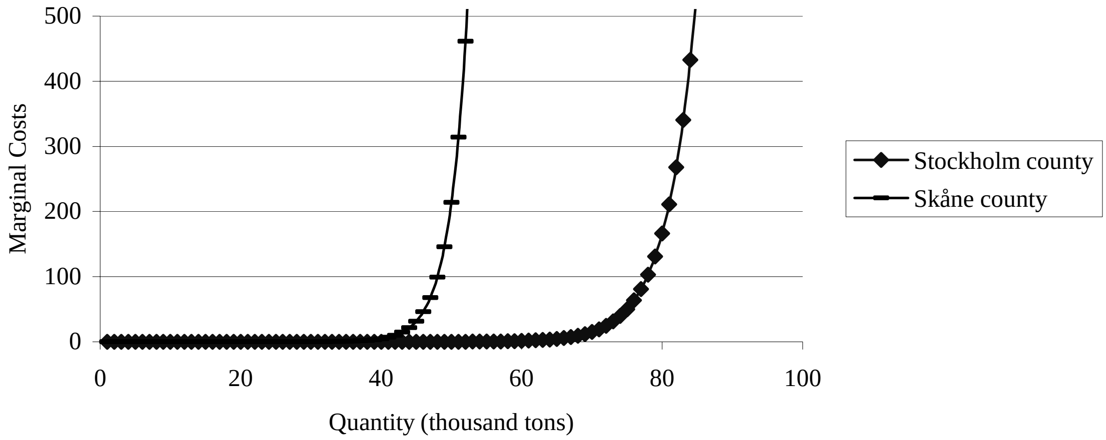

Figure 4: Implied Marginal Costs of Collection in the Skåne and Stockholm Counties

Transport Costs Between Counties

The assessment of the transport costs between the center of a county and a specific mill is relatively straightforward. The obvious and most studied determinant of transport costs is distance, and the general form of transport cost functions employed in location theory models are either linear or concave in the distance. This reflects common empirical observations of the relationship between transport costs and carrying distance (McCann, 2001). Hence, the greater the distance between two regions, the higher the expected transport cost will be. In our model the transport costs between counties and mills are assumed to be a linear function of distance and volume. The main bulk of waste paper is shipped via railways (Georgsson, 2002). The distances used in the present study are the shortest railway routes available as reported by Kjellberg (2002). The cost of transporting a specific quantity is assumed to be linearly increasing in distance. Thus, transport costs are not measured in monetary terms but rather as the product of quantity times the distance shipped. However, they are compatible with the costs of collection since they both measure quantity collected and shipped. Hence, this is not a problem since we only are interested in the relative cost differences between different counties and/or scenarios.

4. A Non-Linear Programming Model of Swedish Waste Paper Flows

This section develops a short-run model for the Swedish corrugated board industry within the framework of operations research. A key assumption of linear programming is that all its functions (objective function and constraints) are linear. Although this assumption may hold for many problems, it is often found that some degree of non-linearity is frequently present in many economic problems (e.g., Baumol and Bushnell, 1967). In the previous section we argued that waste paper recovery provides one such example. Thus, we need to use a non-linear objective function to make use of the marginal cost specification outlined above.

The purpose of the model is to simulate the optimal allocation of Swedish waste paper recovery between counties when unique costs for the individual counties are fully implemented. The model aims at minimizing the total costs of collection and transportation of used corrugated board subject to: (a) exogenous demand requirements at each individual mill; and (b) the supply constraints outlined in Table 2. The basic NLP problem can thus be expressed as:

$$\underset{q_{ij}}{Min}TC = \sum_{i=1}^{n} \int 2e^{0.2\left[\sum_{j=1}^{m}\left(q_{ij}/Q_i\right)\right]} \left(\frac{1}{\beta_i}\right) dq + \sum_{i=1}^{n}\sum_{j=1}^{m} T_{ij} q_{ij} \tag{2}$$

subject to:

$$Supply: \sum_{j=1}^{m} q_{ij} \leq Qi \qquad \forall i \tag{3}$$

$$Demand: \sum_{i=1}^{n} q_{ij} \leq Q_j \qquad \forall j \tag{4}$$

where *TC* is the total cost of collection and transport for Swedish society, *n* is the number of counties and *m* is the number of demand points (paper mills). The first part of the right-hand side of equation (2) is the total collection costs summed over all counties *i.* The second term is the total transportation costs incurred by shipping old corrugated board from the counties to the mills, i.e., T_{ij} represents the cost of transporting one ton of board from county *i* to mill *j*. The term q_{ij} represent the quantity of board that is shipped from county *i* to mill *j*. The constraint in equation (3) implies that for a given county *i* the total volume of waste paper delivered to mills cannot exceed the total amount of paper and board consumption (Q_i)

available in that same county. Equation (4), the second constraint, states that a given paper mill *m* cannot consume more old corrugated board than its total capacity (Q_j). Q_i and Q_j are therefore exogenously given. Equation (4) *de facto* implies that the demand for corrugated board is completely inelastic. This assumption is, though, partly supported by empirical research which reports very low own-price elasticities of waste paper demand (e.g., Gill and Lahiri, 1980; Edgren and Moreland, 1989; Nestor, 1992).

In addition, we need to distinguish between the policy outcome and the cost effective solution. In the former case the recovery goal set by Swedish authorities stipulates that the recovery rate should be the same across counties and equal 65 percent, so that:

$$\sum_{j=1}^{m} q_{ij} \geq 0.65 Q_i \qquad \forall i \tag{5}$$

In the cost-effective case, the only requirement is that the total recovery rate for the country as a whole equals 65 percent, so that:

$$\sum_{i=1}^{n} \sum_{j=1}^{m} q_{ij} \geq 0.65 \sum_{i=1}^{n} Q_i \tag{6}$$

Finally, all delivery levels must be non-negative, so that:

$$q_{ij} \geq 0 \tag{7}$$

In sum, the complete NLP problem is given by the condition that the total costs of collection and transportation, i.e., the delivery costs given in equation (2), should be minimized subject to the supply restrictions in equation (3), the demand restriction in equation (4) and the non-negativity conditions in (7). This general problem was solved for two distinct recovery rate requirements, one involving the *de facto* PRO requirement in equation (5) and one the cost effective solution in equation (6). The data used in the model are drawn from 20 spatially distributed supply centers of waste paper, and six spatially distributed mills as outlined in section 3. The software used to solve the NLP-problem was the so-called Premium Solver in Microsoft Excel (provided by Frontline Systems, Inc.).

5. Model Results and Analysis

In this section we first present the resulting delivery flows given the policy requirement that 65 percent should be recovered in each county. Then follows the cost effective county-specific recovery rates generated by the NLP-model. We continue with a sensitivity analysis of our results. In Tables 5 and 6, all numerical data represent the waste paper delivery flows (in tons), as predicted by the NLP-model. Table 5 shows the simulation results for the case when each and every county fulfills the 65 percent recovery goal.

Table 5: Estimated Solution of Flows when 65 percent in Each County is Collected

Solution (Tons delivered from county/to mill)	SCA Packaging Obbola (Umeå)	Kappa Kraftliner	SCA Packaging Munksund (Piteå)	Munksjö Lagamill	Inland	Skärblacka (AssiDomän)
Stockholm	25532	3841	3841	4239	300	16615
Uppsala	6706	0	0	2110	0	0
Södermanland	0	0	0	4224	2390	1100
Östergötland	0	0	0	6418	3912	2071
Jönköping	0	0	0	6886	2980	0
Kronoberg	0	0	0	5341	0	0
Kalmar	0	0	0	6509	621	0
Blekinge	0	0	0	4541	0	0
Skåne	0	0	1515	15223	11692	5449
Halland	0	0	0	7096	1151	0
Västra Götaland	7	2899	6417	15040	14293	6225
Värmland	0	0	0	4329	4010	0
Örebro	117	0	0	4255	3343	539
Västmanland	2716	0	0	2970	2058	0
Dalarna	5445	0	0	1826	1188	0
Gävleborg	7406	0	0	994	62	0
Västernorrland	7516	0	0	0	0	0
Jämtland	3940	0	0	0	0	0
Västerbotten	7739	0	0	0	0	0
Norrbotten	1986	2897	2897	0	0	0
Sum	69111	9637	14670	92000	48000	32000

In this case the results indicate that Munksjö, for instance, should receive deliveries from 16 out of 20 counties, while the mill Inland, should receive deliveries from 13 counties. Furthermore, the demand constraints are binding for Munksjö, Inland and Skärblacka, the three mills in the south of Sweden, whereas the corresponding constraints for the three northern mills are not. The reason for this is that the mills up north have relatively large capacities (see Table 3) while the supply of old corrugated board is relatively low in the northern counties in spite of a 65 percent required recovery rate.

Table 6 shows the cost effective flows between counties and mills, i.e., when 65 percent throughout the country as a whole is recovered but there exists no uniform recovery rate target for each county. In those cases where the cost effective model does not predict a positive waste paper delivery, but the policy solution (in Table 5) does this, is indicated by bold italic zeros.

Table 6: Estimated Solution of Flows when 65 percent in the Country as a Whole is Collected Cost Effectively

Solution (Tons delivered from county/to mill)	SCA Packaging Obbola (Umeå)	Kappa Kraftliner	SCA Packaging Munksund (Piteå)	Munksjö Lagamill	Inland	Skärblacka (AssiDomän)
Stockholm	39499	***0***	***0***	***0***	***0***	20782
Uppsala	8275	0	0	***0***	0	0
Södermanland	0	0	0	***0***	***0***	7169
Östergötland	0	0	0	7920	***0***	4049
Jönköping	0	0	0	9323	***0***	0
Kronoberg	0	0	0	4753	0	0
Kalmar	0	0	0	6490	***0***	0
Blekinge	0	0	0	4071	0	0
Skåne	0	0	***0***	36081	***0***	***0***
Halland	0	0	0	7791	***0***	0
Västra Götaland	***0***	***0***	***0***	7910	40395	***0***
Värmland	0	0	0	***0***	7605	0
Örebro	***0***	0	0	7661	***0***	***0***
Västmanland	7181	0	0	***0***	***0***	0
Dalarna	7602	0	0	***0***	***0***	0
Gävleborg	7735	0	0	***0***	***0***	0
Västernorrland	6778	0	0	0	0	0
Jämtland	3071	0	0	0	0	0
Västerbotten	6768	0	0	0	0	0
Norrbotten	***0***	3116	3394	0	0	0
Sum	86909	3116	3394	92000	48000	32000

A comparison of the two cases (Table 5 and 6) shows substantial differences in the allocation of waste paper deliveries. Noteworthy is that according to the effective solution SCA Packaging (Piteå) and Kappa Kraftliner in the north should only receive waste paper from their home county, Norrbotten. As in the policy solution, the demand constraints are binding for Munksjö, Inland and Skärblacka, whereas they are not binding for the remaining three mills. However, some of the amounts previously allocated to SCA Packaging Munksund and Kappa Kraftliner (the two mills farthest north) should be reallocated to SCA Packaging

Obbola. Again, the intuitive reason for this is that Obbola is located closer to the large quantities supply of old corrugated board. The former mills should only receive old corrugated board from its home county, Norrbotten. Furthermore, the simulated policy solution yields 52 delivery flows compared to 24 delivery flows in the cost effective setting. Munksjö, for instance, should according to the cost-effective solution receive deliveries from 9 out of 20 counties, while the mill Inland, only should receive deliveries from 2 counties. The county of Stockholm should increase its recovery of corrugated board by about ten percent and ship to only two mills, SCA Packaging (Obbola) and Skärblacka, compared to all six mills in the former case. The reason to this is that Stockholm has a high population density and thus lower costs of collection.

The simulations in Table 6 permit us to calculate the cost effective recovery rate for each county. These are presented in Table 7. Clearly, the policy of recovering 65 percent in each county is not cost effective since our model predicts recovery rates ranging from 51 to 72 percent. The counties of Jämtland and Stockholm are assigned the lowest and the highest recovery rate, respectively. Furthermore, only three counties, Skåne, Västra Götaland and Stockholm should recover more than the policy target of 65 percent, while the rest (17) should recover less. Not surprisingly, the former counties are the residence for the three largest cities in Sweden at present, with a total population that constitutes 50 percent of the total population in Sweden (see Table 4).

By using the cost structure for the two different cases under investigation, we can present an “efficiency loss”, which has been calculated as the cost savings incurred by the cost effective solution as a share of the total costs of the PRO solution. Recall from the discussion in section 3.2 that the absolute figures used are not “true” numbers. However, the interesting part is the relative size of the reported figures, which give us some hints about the potential inefficiencies of the uniform recovery policy. Employing the policy solution, compared to the cost effective one, implies according to our results a relative efficiency loss of 48 percent.[5] That is, the costs of employing the policy implied by the PRO is 48 percent higher than using the cost effective solution. Transport costs are similar while the costs of collection differ substantially between the solutions. The inefficiency thus stems almost only from the fact that collection in the policy solution is constrained to be equal (in relative terms)

[5] The total costs in policy solution were 24192418 units, while the cost-effective solution yielded a total cost of 16305854 units. This gives us a relative “efficiency loss” of 48 percent.

throughout the country regardless of regional differences in costs. In essence, efficiency gains can be obtained by concentrating recycling efforts to the dense population centers.

Table 7: Simulated Recovery Rates in a Cost Effective Setting

County	Recovery rates (in percent)
Stockholm	72
Uppsala	61
Södermanland	60
Östergötland	63
Jönköping	61
Kronoberg	58
Kalmar	59
Blekinge	58
Skåne	69
Halland	61
Västra Götaland	70
Värmland	59
Örebro	60
Västmanland	60
Dalarna	58
Gävleborg	59
Västernorrland	59
Jämtland	51
Västerbotten	57
Norrbotten	54

As was noted above, our analysis builds heavily on specific assumptions concerning, in particular, the collection costs at county level. Still, a sensitivity analysis permits us to assess how changes in these parameters affect the results. First, the coefficient that determines the curvature of the marginal costs may change the outcome of the analysis. We therefore allow the slope parameter (0.2 in the model) to change, using the relative cost share of transport and collection as a basis for the sensitivity analysis. Table 8 below summarizes the results from this sensitivity analysis.

From Table 8 we can conclude that as long as collection costs are *at least* 20 percent of total costs our results are robust. That is, the cost effective recovery rates will not differ much with an increase in the slope coefficient. Not surprisingly, if the transport costs dominate (constitutes more than 95 percent of total costs) the simulated recovery rates will be high in the counties that have, or are close to, mills.

The relative efficiency loss fluctuates between 5 to 100 percent. Noteworthy, though, is that as the relative collection cost share increases, so does the efficiency loss (apart from the 6.5 percent case for which the simulated recovery rates are very similar to the policy requirements).

Table 8: Sensitivity Analysis with Different Marginal Cost Slope Parameters

	Slope Parameter					
	0.25	0.22	0.2	0.18	0.15	0.10
County	**Recovery Rates (in percent)**					
Stockholm	71	72	72	72	61	13
Uppsala	62	62	61	60	58	89
Södermanland	61	61	60	59	59	92
Östergötland	63	63	63	62	65	100
Jönköping	62	62	61	61	64	0
Kronoberg	59	58	58	58	62	82
Kalmar	60	60	59	59	60	0
Blekinge	60	59	58	58	61	73
Skåne	68	69	69	70	77	100
Hallands	62	62	61	62	68	98
Västra Götaland	69	69	70	71	77	92
Värmland	61	60	59	58	53	0
Örebro	61	61	60	59	44	0
Västmanland	61	61	60	59	50	74
Dalarna	60	59	58	57	56	87
Gävleborg	61	60	59	59	61	95
Västernorrland	60	59	59	59	66	100
Jämtland	53	52	51	50	54	85
Västerbotten	58	57	57	58	67	100
Norrbotten	56	55	54	55	62	97
Total costs	207749499	36912298	16305998	10675059	8588428	6739352
Collection costs	199198214	28343027	7733849	2147950	564103	359722
Transport costs	8551285	8569272	8572149	8527108	8024325	6379630
CollCostShare	0.958	0.767	0.474	0.201	0.065	0.053
TransCostShare	0.042	0.233	0.526	0.799	0.935	0.947
Efficiency loss	100%	80%	48%	19%	5%	25%

By looking at the marginal cost curves in Figure 3 we can also conclude that as the authorities raise the preset recovery rate, i.e., above 65 percent, the discrepancies in collection costs between counties amplify.

Finally, changes in the marginal cost shift parameter, β_i, will also change the outcome of the analysis. Logically, if the differences in population densities between counties increase the individual cost effective recovery rates will diverge even more (assuming that the collection to transport costs shares are the same). If we, for instance, use population density without taking logarithms, Stockholm is simulated to recover 78 percent and Norrbotten only 46 percent. However, the main thesis still holds, and the challenge for future research efforts consist of to finding accurate measures of the "true" cost of collection in each county.

6. Concluding Remarks

In this paper we set out to compare the total costs of recovering a minimum of 65 percent of all used corrugated board in each county with the case when the country as a whole recovers 65 percent of all board cost effectively. This was achieved by using a non-linear programming model of the Swedish corrugated board industry.

Since regions are endowed with different sets of geographic and demographic features optimal waste paper recovery rates will differ between regions and thus counties. That is, each region exhibits unique features that need to be considered if efficiency in waste handling is to be attained. Clearly, shifts from high marginal cost collectors to low marginal cost ones will yield efficiency gains. It is also clear that if the preset policy goal increases above the present level of 65 percent, the inefficiency will amplify. Waste paper recovery activities should thus be concentrated to the highly populated and urbanized areas, and not be uniformly distributed across counties.

The main policy implication is therefore that county-specific recovery targets may increase the cost efficiency of the PRO. Still, instead of assigning different recovery rates to different counties one can accomplish this end by introducing a tradable permit system. In theory, the cost effective solution could be obtained by setting an overall recovery rate for old corrugated board and allow paper and board producers and importers to buy permits from the suppliers of old corrugated board. If the recovery rate of 65 percent was set, then the producers of paper and board would have to obtain 0.65 permits per ton produced, and each ton of old corrugated board produced would generate one permit. Paper and board producers who (for technical reasons) face difficulties in consuming large quantities of waste paper would be inclined to buy this permit. In this way, the production of old corrugated board is "subsidized" by the sale of permits, and the production of paper and board is "taxed" by the requirement to purchase permits (Dinan, 1992). Such a policy would, under certain conditions, yield efficiency gains to society. In practice, however, the system may be inappropriate since it may be subject to high transaction costs resulting from administrative complexity, compliance issues, and demand for increased enforcement. In addition, since the Swedish paper industry consists of only a few actors, the permit market would not be competitive and hence efficient. The main question therefore still holds: How should we design a system that ensures cost effectiveness in the waste paper handling process?

This paper only presents a first step in designing such policies. Future research should focus on: (a) finding better data; and (b) finding social efficient recovery targets since this procedure of cost effective analysis in general does not produce an efficient allocation because the predetermined objective of a 65 percent recovery rate may not be efficient. That is, all socially efficient policies are cost effective, but not all cost-effective policies are socially efficient. Policy makers will, therefore, not be helped in the process of designing welfare-enhancing policies unless they consider these issues as well.

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Households’ Perceptions of Recycling Efforts: The Role of Personal Motives

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Abstract

This paper analyzes households' perceptions of recycling activities in a municipality in northern Sweden, Piteå. The purposes of the paper are to analyze whether moral motives matter for: (a) the assessment of households' waste sorting costs; and (b) for the efficiency of introducing economic incentives for stimulating households' recycling efforts. Data were gathered using a mail-out survey to 850 randomly chosen individuals in the municipality of Piteå, Sweden. We employ an economic model of moral motivation with possible motivation crowding-out and econometric techniques. Two main results follow from the analysis. First, the results support the notion that moral motives significantly lower the costs associated with household recycling efforts. Specifically, the average hourly willingness to pay to let others sort household waste at source was found to be significantly lower than the corresponding income after tax (i.e., the opportunity cost of time). Second, moral motives are in some cases the cause of inefficient policy outcomes when introducing economic incentives to promote recycling efforts. Those who initially feel intrinsically motivated to sort waste at source are more inclined to be discouraged by the introduction of a pay-by-the-bag waste management scheme. However, those who sort waste because it is a requirement imposed on them by the authorities, have a positive perception of such as system since it permits a larger degree of flexibility in recycling efforts.

Keywords: recycling, households’ costs, time use, cost benefit, moral behavior, crowding theory

1. Introduction

We live in an age of escalating environmental consciousness, where recycling takes place on an increasing scale and in almost every area of the society. Recycling has traditionally occurred because it has been economical. From the 1970s and onwards, however, the perception in modern rich societies has been that we should recycle even more, something that is expressed by existing or proposed solid waste legislation.[1] In Sweden, since 1994/95, the producer responsibility ordinance has governed the collection and recycling of packaging materials, newsprint, cars and tires. The ordinance states that producers are responsible for the collection and handling of the ensuing waste and for establishing collection systems. The local authorities are therefore not responsible for this, but their conduct is governed by SFS 1998:808 and SFS 1998:811. *De jure* these laws also require households to sort waste at source. There exists thus an increased burden on the households to sort, clean and transport their waste to recycling centers. This has also encouraged some authorities to introduce new policy incentives to stimulate household recycling. The most commonly used waste handling strategy, a flat fee system, is unrelated to the volume of household waste that is discarded. As a result, an observable trend in municipal solid waste (MSW) policy reform is a gradual increase in the use of economic incentives, which is motivated due to its theoretical encouragement of both source reduction as well as waste diversion activity (e.g., Callan and Thomas, 1999). This notion of increased reliance on recycling that implies more sorting and cleaning efforts undertaken by households, as well as the introduction of economic incentives as means to promote household recycling will be the focus of the analysis in this paper.

Local and national authorities use both economic and political instruments in their attempts to induce households to contribute to sustainable development. However, there is a lack of understanding of how these tools interplay with the motives held by households and the daily constraints they face. There exist basically two strands of thinking with respect to this matter. First, many economists view recycling as any other activity, and recycling policies ought to, just as other policies, pass standard cost-benefit and efficiency tests (e.g., Baumol, 1977). In addition, economists usually argue that for the purpose of cost-benefit

[1] A result of this is that a "waste hierarchy" has emerged. For instance, the following waste management options are outlined: reduce, reuse, recycle, incinerate and landfill, where the first three are considered "good" options and the last two "bad" ones. This hierarchy was first put forward by environmental organizations such as *Friends of the Earth* and *Greenpeace*. The European Union uses the same order of preference in its directives for packaging waste (Ackerman, 1997).

analyses it does not matter why people value environmental goods (or any other good for that matter). As such, economists assume that all preferences are private, and they grant equal credibility to every motive that underlies these preferences (e.g., Harrison, 1992). A person's motives are unchallengeable and are thus irrelevant when evaluating recycling activities. For instance, for a given utility-maximizing individual the time spent on these activities should be valued as a pure opportunity cost of lost leisure to the individual. Another strand of thinking points out that motives do matter and that people undertake recycling activities for moral reasons (e.g., Ackerman, 1997; Frey, 1999). For instance, people may derive benefits from participating in recycling activities, e.g., gain positive feelings of "warm-glow" from contributing to a better environment. If this supposition is correct, we would expect households' willingness to pay (*WTP*) to avoid one hour of waste sorting and cleaning activities to be lower than their hourly net income. In addition, if households take a strong positive moral stance to waste sorting, the use of economic cost incentives may be an ineffective policy tool as it may undermine an individual's sense of civic duty (Frey and Oberholzer-Gee, 1997). Ackerman (1997, p. 32) support this notion and concludes: "unit pricing is remarkable not for how much it accomplishes, but for how little," and he go on to stress the role of non-market motives in household recycling. Thus, if people do not respond very much to price incentives for waste collection, economic incentives, such as e.g. a unit-pricing system, would only accomplish a moderate change in individuals' behavior.

The purposes of this paper are to analyze: (a) to what extent motives for undertaking recycling activities matter for the assessment of households' recycling costs; and (b) the role of motives in determining the efficiency of introducing economic incentives for recycling efforts. We accomplish this by gathering data using a mail-out survey to 850 randomly chosen individuals in the municipality of Piteå, Sweden, and by employing an economic model of moral motivation with possible motivation crowding-out as well as econometric techniques.

The paper proceeds as follows. Section 2 discusses the potential roles of motives in household recycling, while section 3 outlines the basic theoretical models of household recycling efforts used in the analyses. Section 4 discusses the development of the questionnaire that was used in the study and presents some general survey statistics. In sections 5 and 6 the empirical results are presented and discussed. Finally section 7 concludes the paper and provides some implications for recycling policy.

2. The Role of Motives in Household Recycling

From an economic point of view we posit that there are basically two reasons for why motives may be important in the analysis of household recycling efforts, viz., (a) they may matter for the assessment of households' sorting and cleaning costs; and (b) they may impact on the effectiveness of incentive based recycling policies.

2.1 The Assessment of Households' Sorting and Cleaning Costs

In a proper social cost-benefit analysis all costs and benefits associated with waste disposal activities have to be included. Only then will it be possible to assess the different options from society's point of view. But while some cost data on the waste handling process are relatively easy to extract the assessment of other costs, e.g., the time devoted by households to undertake recycling activities, regularly poses problems. All analyses that attempt at assessing the waste disposal costs imposed on society must make a serious effort to identify such costs, provide their quantitative dimensions, and, finally, to monetize them. In previous analyses of the social costs of waste treatment (e.g., Radetzki, 2000; Bruvoll, 1998), the value placed on the time households spend on sorting and cleaning waste constitutes a substantial share of the total cost of recovery. These studies have based the valuation of time on labor costs after tax (which we will refer to as the time-use approach).[2] In Bruvoll's (1998) study, households' collection costs for residential paper waste constitute over 40 percent of the total marginal recycling cost. The results presented by Radetzki (2000) show an even higher share for these collection costs.

Such conclusions have initiated a debate on whether households' sorting and cleaning costs should be considered a pure cost or perhaps even a benefit. One strand of thinking is that households' time devoted to clean and sort the waste on a daily basis should be seen as a cost to society, due to the opportunity costs of the time in terms of foregone leisure (e.g., Wiseman, 1990). However, according to an alternative view people may derive benefits from participating in recycling activities and, in addition, that recycling schemes may educate people to be more aware of resource use (Powell et al., 1996). For instance, people may gain a

[2] In these cases it is assumed that the opportunity costs for undertaking recycling activities is the value of lost leisure. The theoretical basis for this is outlined in Johansson (1993, p. 82): "Workers drawn from leisure activities should be valued at their reservation wages. If labour markets clear, the reservation wage is equal to the after-tax wage rate. This is the lowest amount of money inducing an individual to switch, at the margin, from leisure time to work time."

positive feeling of "warm-glow" from contributing to a better environment (Andreoni, 1990). If this latter conjecture is correct, we would expect households' willingness to pay (*WTP*) to avoid one hour of waste sorting and cleaning activities to be lower than their hourly net income.

Sterner and Bartelings (1999) employ a contingent valuation (CVM) study and estimate the annual average *WTP* of Swedish households to avoid recycling at approximately USD 38. In a similar fashion Bruvoll et al. (2002) find that the average Norwegian household is willing to pay about USD 20 annually to let others sort their waste. Furthermore, by dividing the average *WTP* by the hours individuals claim to spend on recycling activities Bruvoll et al. end up with a *WTP* per hour of about 45 US cents. This is clearly significantly lower than the corresponding net hourly wage assumed in Bruvoll (1998) and Radetzki (2000), which equals about USD 6. Sterner and Bartelings (1999) report that most people view the recycling activity as one of the most tangible actions that can be undertaken to contribute to a healthier environment, something which supports the notion that households generally take a moral stance in relation to their waste sorting activities. Hence, Sterner and Bartelings conclude that the time devoted to household recycling cannot entirely be considered a cost.

In this paper we follow the approaches employed by Sterner and Bartelings (1999) and Bruvoll et al. (2002), and compare Swedish households' *WTP* to avoid waste sorting and cleaning with their implied time valuation based on net income. However, in contrast to these earlier studies we analyze in more detail to what extent different recycling motives can explain the expressed willingness to pay.

2.2 The Effectiveness of Incentive-based Recycling Policies

Economists usually argue that the market is uniquely efficient in allocating resources. The key element in environmental policy is therefore to "get the prices right" and to ensure that all environmental externalities are accounted for in market mechanisms. Recently governments and local authorities have shown an increased interest in the practice of implementing schemes charging households by the pound they discard (i.e., a pay-by-the-bag scheme) as a way of attaining set recycling targets.[3] Hence, if households have to pay for additional waste disposal, they will reduce their waste generation and/or recycle more. In addition, such a scheme would, it is argued, also create more freedom for each household in deciding how

[3] Of course, the level set by the authorities may not necessarily be the economically efficient level of recycling.

much to sort than under a mandatory scheme. For instance, households whose opportunity cost of time is high are permitted to sort less as long as they pay for the extra (non-sorted) waste generated. This ensures a cost-efficient division of recycling activities across households.[4]

However, the standard economic approach to recycling incentives assumes that households' underlying motives for undertaking recycling activities are unaffected by the policy implemented. Recent research in the field of economic psychology suggests, though, that this is not necessarily the case. This literature analyzes the effect of price incentives by drawing a distinction between peoples': (a) intrinsic motivation, where the person likes the activity in itself, partly, due to moral considerations; and (b) their extrinsic motivation, where the person does it because of monetary payment or because of compulsion (e.g., Frey, 1992, 1999).[5] For example, we can sort waste either because our environmental morals tell us to do so or because we are told to do so by authorities or because it is financially beneficial. That is, moral reasons may come into play for people when undertaking a supposedly environmentally beneficial activity. One of the most important and empirically best-founded reasons for this is set up within the so-called crowding theory, which points to the possible destruction of intrinsic motivation.

When an external intervention in the form of a reward reduces individuals' intrinsic incentives to act it is referred to as "the hidden costs of reward" (e.g., Deci & Ryan, 1985; Pittman and Heller, 1987). Frey (1999, p. 399) generalizes this "hidden costs of reward" in two respects:

> (1) *All* outside interventions can affect intrinsic motivation; in addition to *rewards* the same effect can come about by external *regulation* (commands); and (2) External interventions *crowd out* intrinsic motivation if they are perceived to be *controlling* and *crowd in* intrinsic motivation if they are perceived to be *acknowledging*.

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[4] However, the empirical evidence of realized efficiency gains is not unambiguous. For instance, in the absence of illegal dumping opportunities unit-based pricing is the most efficient way of reducing waste (e.g., Jenkins, 1993), while Fullerton and Kinnaman (1995) show that in the presence of illegal dumping the optimal charge may be zero or even negative.

[5] The difference between the two motivational factors may not be clear-cut. However, for our purposes we distinguish between the two different motivations by following Frey (1997, p. 14): "for the purpose of explaining economically and socially relevant human behavior, it suffices that it makes sense to distinguish activities which individuals (mainly) do just because they like them, and others which they (mainly) do because they are induced to do so by monetary payment or by command."

These generalizations help us think productively about moral dimensions of policy problems and incorporate a dynamic dimension into our analysis. One illustrative example of the crowding theory is a case study of a "token economy" where old people living in asylums were exposed to different economic incentives such as making their beds in exchange for vouchers. After some time these people were no longer prepared to do anything if they were not paid for it, i.e., they were "demoralized" by the incentive structure presented to them (Frey, 1999).[6]

The lesson learned here is that "commercialization" can tear down moral values while support can reinforce them. This is useful information when evaluating and designing environmental policies. Let us use recycling schemes as an example. In Bruvoll et al. (2002) moral responsibility is presented as one important motive to why people choose to undertake time-consuming recycling activities. In other words, many households feel intrinsically motivated to sort and clean their waste at source. In this case the crowding theory postulates that if people undertake recycling activities for moral reasons, price-based waste management policies (e.g., a pay-by-the-bag system) may crowd out this intrinsic motivation and may on balance even lead to less recycling activity being undertaken. Thus, even though the new financial incentive in itself increases the extrinsic motivation to recycle, it may also be perceived as controlling and serves therefore not as a complement but rather as a substitute to the intrinsic motivation. This idea is in sharp contrast with the standard economic argument that price-based policies provide more flexibility in choices and are thus less (rather than more) controlling. Which of these views that dominate in practice remains an empirical question.

Previous empirical applications of the crowding theory focus mainly on labor supply performance, common pool resources, services, and constitutions and tax evasions (for a review of these studies, see Frey and Jegen, 2001). In this paper we add to this empirical research by testing the crowding hypothesis on households' recycling behavior. We do this by investigating households' perceptions of a pay-by-the-bag system, and whether this perception differs across households due to moral motivation.

[6] In addition, many personal relationships are valued exactly because they cannot be bought. Consider your own situation when you are having your best friends over for dinner. If they at the end of a (hopefully) nice evening started to insist on paying you for food, drinks, snacks, etc., your motivation for having them over again would probably diminish. On the other hand, if they ring you up the next day praising your cooking skills and cheering for an exceptionally lovely night, you probably are more motivated to have them over again (Frey, 1999).

3. Economic Models of Household Recycling and Motivation

In this section we present the theoretical framework used for analyzing households' recycling efforts. The recycling behavior modeled here is assumed to be both involuntary and voluntary. It is involuntary in the sense that every individual is *de jure* required to sort and clean his or her waste, but it is voluntary in the sense that the laws *de facto* seldom are enforced. The model outlined in sub-section 3.1 is derived from Bruvoll and Nyborg (2002). In this model recycling is observed even in the absence of monetary incentives, a common situation in many municipalities. Finally, in sub-section 3.2 we introduce monetary incentives into the model. The two models will be used to: (a) assess households' waste sorting costs; and (b) test the crowding hypothesis in the empirical context of a pay-by-the-bag waste management system.

3.1 Moral Motives and Households' Recycling Costs

The model in this section assumes that individuals' motives for recycling activities are in part due to a preference for a better environment and in part due to one's self-image as a responsible person. The self-image is in turn determined through a comparison of the individuals' own recycling efforts against a social norm. We will consider two different types of social norms. We first assume that the social norm is set by the authorities and is thus exogenously given. That is, what is considered a "fair share" of recycling activities is dependent on how much authorities require us to undertake. At the end of this sub-section we then consider the case when the social norm is endogenous (Bruvoll and Nyborg, 2002; Brekke et al., 2002).

Individuals' ethical compasses guide them in making decisions about environmental concerns. At the same time institutions prevailing in society affect this compass and are amenable to change (North, 1990). Economic activities, like all social phenomena, are therefore necessarily embedded in culture, which includes all kinds of social, political and moral value-systems and institutions. These value-systems and institutions profoundly shape and guide human behavior by imposing obligations, enabling and disabling particular choices, and creating social or communal identities, all of which may impact on economic behavior. We will first here focus on exogenously given norms, e.g., a feeling of shame if one does something illegal and unaccepted such as not sorting his or her waste (whether one is caught or not).

Let an individual's preferences be represented by the following general utility function:

$$U = u(c, l, E, S) \tag{1}$$

where c represents the individual's consumption of private goods, and l represents leisure time. The individual further derives utility from a pure public good, in this case environmental quality, E, and from his or her own self-image as a responsible person, S. The self-image responds to a person with a moral compass well matched with society's norms of responsible conduct (Brekke et al., 2002; Bruvoll and Nyborg, 2002). The utility function is increasing and quasi-concave in c, l, E, and S.

Labor supply and income are assumed to be exogenously given, this since the time devoted to recycling efforts and leisure are the primary focus of our analysis. The time constraint faced by the individual is thus given by:

$$l + r = T \tag{2}$$

where T is the total amount of time vacant for leisure and for recycling activities, r.

The utility derived from environmental quality, E, stems from two sources, the quality exogenously supplied by others, E_0, and the environmental quality improvement, e, arising from the individual's own recycling efforts:

$$E = E_0 + e \tag{3}$$

The contribution e is in turn determined solely by the recycling effort undertaken by the individual so that:

$$e = e(r) \tag{4}$$

The individual's contribution to a better environmental quality, e, is assumed to increase at a constant or diminishing rate, and will of course be zero if no recycling effort is undertaken.[7]

[7] Mathematically, $e_r>0$, $e_{rr}\leq 0$, and $e(0)=0$, where the subscripts denote derivatives.

Furthermore, each individual's self-image is assumed to relate to compliance to the prevailing norms in society. That is, the self-image increases if one contributes more than what is considered as minimum by society at large, and diminishes if one does not conform to the norm. For this reason we assume that the self-image is a function of the difference between the actual contribution and the social norm, *e**, so that:

$$S = S(e - e^*) \tag{5}$$

Put another way, we can consider the self-image as a distance; the further away in positive terms we are from the social norm (*e**), the higher is the self-image derived from the recycling effort undertaken, and vice versa.[8]

Maximizing utility, *U*, with respect to the recycling effort, *r*, for the individual subject to equations (2)-(5) yields the following first order condition:

$$\frac{\partial U}{\partial r} = -u_l + u_E e_r + u_S S' e_r = 0 \tag{6}$$

Rearranging we get:

$$e_r(u_E + u_S S') = u_l \tag{7}$$

Equation (7) shows that in optimum, the individual marginal benefits of recycling efforts equal the marginal costs of lost leisure for undertaking the activity. The benefits stem from two sources; the environmental benefits accruing to the individual herself (which are likely to be very small), and the improved self-image.

Let us now consider the scenario under the assumption that the social norm is endogenous to the individual. That is, the self-imposed norm stems from moral motives to recycle, which in turn may be influenced by public policy. Brekke et al. (2002) model the social norm (*e**) as endogenously determined through the individuals' own moral compass; *e** is now thus the *individuals'* own perception of the ideal contribution. We again assume that individuals make trade-offs between the wish to be socially responsible and the desire for consumption and leisure. In the model *e** is defined as the contribution that maximizes overall

[8] We assume that the derivatives $S'>0$ if $e<e^*$, and $S'=0$ if $e \geq e^*$. Furthermore, we assume that $S'' \leq 0$.

welfare only if made by everyone (otherwise no effort would be worth doing for the individual since his/her own contribution is insignificant). Assuming that every individual is identical, and then solving for social welfare maximum gives us the ideal contribution, e^*. Once e^* is determined the individual takes this into account maximizing his/her own utility as previously shown. The self-image will then alter the utility function regardless of whether the norm is exogenous or endogenous but through different sources (Bruvoll and Nyborg, 2002).

We will now turn to the theoretical underpinnings for the empirical estimation of the assessment of households' costs for recycling efforts. If individuals could give up their sorting by leaving it to someone else, thus keeping the environmental quality unchanged, i.e., abstracting from environmental benefits, we can analyze the costs of imposing recycling efforts on the households (i.e., on the individuals). That is, if the responsibility for sorting waste is shifted from individuals to a central sorting facility supplied by local authorities, the relationship in equation (3) is rewritten as:

$$E = E^c \tag{8}$$

where E^c is the level of environmental quality resulting from the central sorting process. However, since people may have moral motives to participate in recycling activities since it is one of the most tangible action that make them feel that they contribute to a better environment (Ackerman, 1997), they may experience a decreased self-image by leaving the sorting of waste to others. To allow for such a scenario in our model we therefore use superscripts for the self-image variable using 0 to indicate that individuals are sorting their waste themselves and 1 to indicate when sorting is done by local authorities.

Using our basic utility function in equation (1), and assuming that the environmental quality is unchanged by shifting the sorting from individuals to a central sorting facility (i.e., E^0=E^c), the change in utility that individual i experiences can thus be expressed as:

$$\Delta U_i = u_i\left(c_i, T - r_i^0, E^0, S_i^0\right) - u_i\left(c_i, T, E^c, S_i^1\right) \tag{9}$$

However, since utility is not directly observable and since individuals' recycling efforts are not part of market transactions we need to find alternative ways of assessing the economic value of the shift in responsibility. Theoretically a standard monetary measure for this would be the compensating surplus, which in optimum would be equal to the maximum willingness to pay for letting others sort their waste (e.g., Freeman, 1993). Therefore, a monetary measure for individual *i*'s welfare change due to the difference in utility caused by the shift of responsibility is given by the willingness to pay, WTP_i, and it is defined by the following equality:

$$u_i(c_i, T - r_i^0, E^0, S_i^0) = u_i(c_i - WTP_i, T, E^c, S_i^1) \tag{10}$$

where WTP_i equals the loss of consumption that keeps the individual's utility constant. Since c_i and E are unchanged, l_i increases and the change in S_i is ambiguous, WTP_i is ambiguous. The relative size of the utility derived from increased leisure and potentially decreased self-image will determine the sign of *WTP*. If the self-image is unchanged WTP_i must be positive. However, *WTP* might be lower than a pure opportunity cost of lost leisure due to diminished self-image.

In sum, the implications from the above discussion on exogenous and endogenous norms give us two possible outcomes of individuals' *WTP* for evading sorting waste at source. *First*, in the case that an individual perceives the social norm to be set by the authorities (and is thus exogenously given to the individual), i.e., what is considered a "fair share" of recycling activities is dependent on how much authorities require he/she to undertake, we would expect his/her *WTP* to be positive and relatively high. The reason is that the effort undertaken by the individual is perceived as compulsory due to a requirement imposed by authorities, and being able to leave the sorting activities to others thus would relieve the individual from the responsibility. *Second*, under the assumption that the individual perceives the social norm as endogenous, i.e., the self-imposed norm stems from his/her moral motives to recycle (and thus from the individual's own perception of the ideal contribution), and experiences increased self-image from undertaking recycling activities. Consequently, if someone else takes over these activities they might experience lower self-image. We would therefore expect his/her *WTP* to be lower than for those who feel it is a compulsory activity. If an individual looses a sufficient amount of self-image, it may even be the case that their *WTP* is negative. Finally, we need to consider a third option not outlined above. Regardless of whether the social norm is considered exogenous or endogenous, if recycling activities are perceived as pleasant activities in itself (i.e., the case when recycling activities, *r*, enters the utility function directly) we may also find that *WTP* becomes relatively low (or even negative).

3.2 Moral Motives and The Effect of Price Incentives

According to the traditional economic approach to household recycling, price-based waste management policies will, if monitored properly, provide the proper incentives for households to sort and clean their waste. It may even be the case that individuals are likely to be positive towards the introduction of, for instance, a pay-by-the-bag system. There are two reasons for this. *First*, the individual may receive financial compensation (i.e., reduced waste bill) for their recycling efforts. *Second*, and more importantly, some individuals feel that the norm (e^*) is not as prevalent any more. In line with the model above S will increase (since the new flexible system will ”loosen” up the norm). In other words, now it is legitimate not to sort as long as you pay for it. Hence, the individual faces more degrees of freedom in sorting activities, and can thus choose the optimal amount of recycling activities to undertake. We can therefore put forward the hypothesis that economic incentives such as pay-by-the-bag recycling scheme would be perceived as rather good, at least by those who previously felt that it was compulsory to sort. Thus, according to this approach extrinsic motivation is the primary determinant of recycling behavior.

However, other individuals may be intrinsically motivated to undertake recycling activities. This may be because they obtain a “warm-glow” from contributing to a better environment or simply because they perceive recycling as a pleasant activity in itself. According to the crowding theory, though, the introduction of price-based incentives reduces the intrinsic motivation for recycling. In order to model this situation we modify Bruvoll and Nyborg’s (2002) theoretical framework by: (a) introducing a monetary compensation (i.e., a cost reduction) for recycling activities, f; and (b) considering a slightly different formulation of the self-image component.

Let us therefore now assume that an individual’s preferences can be represented by a utility function of the following general form:

$$U = u(c, l, E, S, f) \qquad (11)$$

where c represents the individual’s consumption of private goods, and l represents leisure time.

In this formulation we assume that the monetary compensation, *f*, enters directly into the utility function.[9] The individual further derives utility from a pure public good, in this case environmental quality, denoted by *E*, from his or her own self-image as a responsible person denoted by *S*, and the monetary compensation directly derived from the sorting activities. The utility function is increasing and quasi-concave in *c*, *l*, *E*, *S* and *f*. Thus, according to this order of preference, increased financial benefits gives higher utility, i.e., $u_f > 0$. The constraints that the individual face are given by:

$$l + r = T \tag{2}$$

$$E = E_0 + e \tag{3}$$

$$e = e(r) \tag{4}$$

These equations equal those in the model in section 3.1. However, in this alternative model we assume that each individual's self-image, *S*, is a direct function of the recycling effort undertaken and the financial incentive. It can thus be expressed as:

$$S = s(r, f) \qquad \text{where } s_r > 0 \text{ and } s_f < 0 \tag{12}$$

where *S* is understood as the self-image the individual gains from contributing to the environment. The individual will experience higher utility from increased recycling activities since he/she thinks that it is a pleasant activity in itself and/or because he/she gets a positive feeling (i.e., warm-glow) from undertaking such activities. Equation (12) shows thus that the higher the individuals' recycling efforts the higher the self-image they experience from these activities, which in turn yields utility gains according to equation (11). However, equation (12) also shows that the higher the monetary compensation, the larger will be the destruction of the intrinsic motivation, and thus of ones self-image (Frey, 1999). Finally, we assume that:

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[9] Usually income, i.e., in our case monetary compensation, enters the utility function indirectly via the income constraint. However, specifying the nature of the income constraint for each type of transaction can quickly become cumbersome and analytically intractable. For analytical reasons and since our model excludes other incomes, we put the monetary compensation, *f*, directly into our utility function. See, for instance, Sidrauski (1967) for a similar approach.

$$f = f(r) \quad \text{where } f_r > 0 \tag{13}$$

Thus, the more the individual sorts at source and thus contributes to a better environment, the more financial benefits he/she will receive. Each individual maximizes utility (11) subject to the constraints (2) through (4), (12) and (13) by choosing how much recycling effort, *r*, he/she wishes to supply. The first order condition for this utility maximization problem is:

$$\frac{\partial U}{\partial r} = -u_l + u_E e_r + u_S S_r + u_S S_f f_r + u_f f_r = 0 \tag{14}$$

Rearranging we get:

$$u_E e_r + u_S S_r + u_f f_r = u_l - u_S S_f f_r \tag{15}$$

The optimal level of recycling activities, *r**, occurs at the point where the marginal benefits of recycling efforts equals the marginal costs. The left hand side of equation (15) shows that the individual marginal benefits of recycling efforts consist of increased utility from better environmental quality, increased self-image from the activity itself and the financial benefits. The right hand side shows the marginal costs stemming from lost leisure and diminished self-image from the economic incentives introduced (i.e., crowd-out effect) for undertaking the activity (Note that $S_f < 0$).

What if the authorities increase the monetary incentive (i.e., *f* increases)? The expression in equation (15) shows that in general, recycling efforts may increase or decrease when monetary incentives *f* increases, depending on whether the utility stemming from reduced waste bill outweighs the perceived disutility from the destruction of self-image.

In sum, this modified model recognizes that the extrinsic incentives, in this case the introduction of a price-based system, induce a person: (a) to supply more of recycling efforts due to the monetary incentives; but also (b) to contribute less since the monetary incentives crowd-out any existing intrinsic motivation. This is, therefore, not a tract against conventional economic wisdom; the relative price effect still is fully acknowledged. Higher price induce more recycling efforts supplied. However, the crowding out effect is an additional force working in the opposite direction (Frey, 1999). The net effect of the introduction of a price-by-the-bag system is therefore ambiguous, and remains an empirical question.

4. The Survey

4.1 Questionnaire Design Issues

One of the most influential reports in the area of contingent valuation methods (CVM) is the NOAA-panel report (Arrow et al., 1993), which questions the possibility of getting valid responses from mail-out surveys since they are likely to be sent to populations that are not representative of the relevant research population, and since the non-response rate may be high. The NOAA-panel therefore suggests that personal interviews should be employed instead. The panel points to the fact that the reported figures from a mail-out format usually have a built-in sample selection bias since the respondents get to review the questionnaire before deciding to answer. Therefore, in our case people who are active sorters may tend to be more inclined to respond than others. Nevertheless, in this study a mail-out survey was chosen over an interview approach since it was deemed cost efficient. That is, a larger group of respondents could be reached for the same amount of financial resources and time that were available. We also test explicitly for the possible existence of sample selection bias (see section 4.2).

The mail-out questionnaire was developed using information from various sources. First during 2001 a pilot study was conducted in the Swedish municipality of Luleå from which valuable lessons were learned (Karlsson, 2002).[10] In addition, the questionnaire developed by Bruvoll et al. (2002) was also reviewed and discussed.[11] These pre-study efforts resulted in some modifications of some questions in the final questionnaire and the design of the introductory letter.[12] The modifications were mainly semantically oriented.

The first part of the questionnaire included questions about the respondents' attitudes to recycling activities and about to which extent they sort and clean their waste at source. Among other things, the respondents were asked to state the average time per week they spend on recycling activities, and how much they were willing to pay (at the most) for letting someone else sort their waste. The best method for eliciting maximum willingness to pay (*WTP*) within a contingent valuation study is an issue of debate in the economic literature. For

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[10] Karlsson (2002) analyzes private and public preferences towards household recycling among 200 households in the municipality of Luleå by, among other things, asking the respondents about their willingness to pay for letting others sort their waste. The author estimates the *WTP* at USD 12.70 per year for evading sorting at source, and concludes that the respondents expressed both private and public preferences towards recycling activities.

[11] The author is indebted to Annegrete Bruvoll and Bente Halvorsen for gaining access to their questionnaire and for valuable suggestions.

[12] The introductory letter and the entire questionnaire sent out to respondents are included in Appendix A.

example, Loomis (1990) concludes that estimates from discrete choice studies are at least as reliable as when respondents directly state their willingness to pay in an open-ended question. Open-ended formats may also induce strategic bias problems. However, others (e.g., Jakobsson & Dragun, 1996) point out the difficulty in designing the bid vector, i.e., the range of prices given to respondents, in the discrete choice procedure. In addition, Kriström (1990) concludes that there is no clear consensus on the best way to derive summary statistics such as mean values for willingness to pay with discrete choice methods, and it has been found that discrete choice questions can lead to starting point bias (Mitchell and Carson 1989; McFadden 1994). This approach would not provide willingness to pay estimates in terms of consumer surplus and would therefore not provide data commensurate with standard cost-benefit analysis (Bateman and Willis, 1999). On the other hand, open-ended *WTP* questions, if stated correctly, should measure the "true value" individuals assign to the efforts undertaken, and will thus capture consumer surplus. In addition, in our case respondents may have compensating demands for letting others sort their waste, and open-ended questions allow respondents to state negative values. However, the unfamiliarity of respondents in determining, rather than reacting to, valuations seems likely to increase uncertainty and therefore the variance in responses (Ibid.). Nevertheless, in a recent study comparing open-ended and dichotomous choice values, Loomis et al. (1997) suggest that "it maybe premature to abandon use of open-ended *WTP* questions" (p 121).

In this study, we use the open-ended question approach basically for three reasons: (1) the starting bid was not clear; (2) it permits the respondents to state a negative value; and (3) the most common critique against open-ended questions is that they give rise to so-called strategic bias. In the case where there is no actual payment required, the respondent may be induced to state a relatively high *WTP* in order to increase the likelihood that the good is provided. However, in our case this is not really a problem. Our hypothesis is that the households' willingness to pay (*WTP*) to avoid one hour of waste sorting and cleaning activities is lower than their hourly net income. In this case it may therefore be motivated to use a method that overrates (rather than underrates) the individuals' *WTP*.

The respondents were also confronted with a number of statements regarding issues associated with household recycling and rural solid waste management, and were asked to state the degree to which they agreed or disagreed with these statements. The last part of the questionnaire contained socio-economic questions about, e.g., income, age, gender, living conditions, etc. At the end some space was reserved for the respondents to freely comment on the questionnaire or on other related issues. 177 respondents took this opportunity to comment on recycling activities (almost everyone commented on recycling in general, while only a handful commented on the questionnaire as such).

4.2 Survey Logistics and Sample

In February 2002, the questionnaire was mailed to 850 residents in a municipality in northeast Sweden, Piteå. The municipal waste management system in Piteå resembles closely those of other Swedish municipalities. Households are required to sort and clean their waste at source, and then transport it to recycling centers. All households pay a fixed fee for waste management, and there is thus no monetary incentive present. The response rate was approximately 71 percent after one reminder, corresponding to 603 responses. In Table 1 the socio-demographics of the survey respondents are presented and compared to a typical Piteå resident.

Table 1: Socio-demographic Comparison

Variable	Sample	Typical Piteå Resident
Mean Age	49.6	40.5
Mean Income (Monthly)	USD 1942-2427$^{\#\varphi}$	USD 1586$^{\#}$
Education (% university level)	25%	26%
Gender (% female)	53%	50%

$^{\#}$ Calculated using USD 1 exchanged for SEK 10.30.

$^{\varphi}$ Mean incomes for household including wage earnings and transfer payments.

Sources: Lindqvist (2002) and Piteå Kommun (2002).

As Table 1 shows, the average age of the respondents in the sample is slightly higher than that of the average Piteå resident. The over-representation of older people may arise from two sources. First, the mail-out survey was only sent to adults (with their own address) while Piteå of course is the residence of many children and teenagers as well. Second, senior people are more likely to respond to questionnaires than younger ones due to their relatively lower opportunity cost of time. Regarding income, note that mean income (including transfer payments) for *households* are compared with wage earnings only for a typical *individual.* A typical Swedish household consists of 1.55 adults (SCB, 2002). By using this figure for a household in Piteå we find that the sample mean income equals about USD 2458 and that compares fairly well with that of the typical resident. The level of education and the gender mix in the sample also compare well with the average Piteå resident. Overall the above implies that the results presented below can be considered reasonably representative for the Piteå population.

5. Empirical Estimates of Households' Recycling Costs

This section presents the results from comparing the traditional estimate of the opportunity cost of time (i.e., net income) with the individuals' willingness to pay to let someone else take over the time-consuming waste sorting activities. We end the section by outlining a regression model, which attempts at predicting the stated willingness to pay values. Before proceeding, however, it is useful to look at some of the key responses, and in particular those related to the individuals' motives for waste sorting. We first note that most respondents reported that they sort most of the types of waste fractions they generate at source. In other words, overall the respondents complied well with the requirement to sort and clean their waste.

Table 2: Motives for sorting waste

I sort partly because:	Agree	Partly agree	Partly disagree	Disagree	No conception	No answer
a) I want to think of myself as a responsible person.	49%	20%	6%	8%	7%	10%
b) I should do what I want others to do.	62%	13%	3%	6%	5%	11%
c) I want to contribute to a better environment.	73%	11%	4%	2%	3%	7%
d) It is economical for the society at large.	31%	14%	11%	10%	22%	12%
e) I want others to think of me as a responsible person.	28%	13%	9%	25%	13%	12%
f) It is a requirement imposed by the authorities (state or municipal).	50%	17%	10%	11%	3%	9%
g) It is a pleasant activity in itself that brings me satisfaction.	26%	14%	15%	23%	10%	12%
h) It is economical for me as a person.	11%	7%	9%	41%	20%	12%

Table 2 shows how the respondents perceived different statements about possible motives for undertaking waste sorting activities. It is worth noting that the respondents are heterogeneous with respect to most motives for sorting waste at source. For instance, roughly half of the respondents claim that they sort partly because they want to think of themselves as a responsible person, while the same share sort partly because it is a requirement imposed by the authorities.

Table 3 lists the variables that are used in the ensuing analyses.

Table 3: Description of Variables Used in the Empirical Analyses

Independent Variables	**Description**
INCOME	The households' monthly income (including transfer payments) measured in eleven levels (see Appendix A)
GMI	Green moral index ranging between 0 and 16
GENDER	1 if male, 0 if female
AGE	The respondent's age in years
EDUCATION	1 if university education, 0 if otherwise
LIVING	1 if house owner, 0 if otherwise
DISTANCE	Distance in meters to recycling center
COMPULSORY	The degree to which respondents feel that sorting at source is a requirement imposed by the authorities. 4 if fully agree, 1 if do not agree, 0 if no conception*
PROFITABLE	The degree to which respondents feel that they sort at source since it is economical for them as a person. 4 if fully agree, 1 if do not agree, 0 if no conception*
DO WHAT OTHERS SHOULD DO	The degree to which respondents feel that they sort at source because they should do what they want others to do. 4 if fully agree, 1 if do not agree, 0 if no conception*
PLEASANT	The degree to which respondents feel that they sort at source because it is a pleasant activity in itself. 4 if fully agree, 1 if do not agree, 0 if no conception*

* We did not exclude respondents who answered "no conception" to this question in order to keep the sample as large as possible. For each statement only a relatively low share of the respondents marked "no conception" in the sample. Of course, we performed a likelihood ratio test by including dummies of "no conception" for each statement in both regressions. However, we could not reject the null hypotheses that the statement dummies had no effect on the estimations. We therefore present results based on the variable descriptions as above.

In most cases these have been drawn directly from the survey responses. However, the so-called green moral index (*GMI*) requires some explanation. The calculation of the *GMI* is based on the responses to the first four statements in Table 2. When creating the *GMI* we ranked the degree to which the respondents agreed to these statements. "Agree" was assigned a four (4), "Partly agree" three (3), "Partly disagree" one (1), and "Disagree" zero (0). The response "No conception" was given the mean value so as to not to bias the index in any way.

By aggregating these values for each respondent, we obtained the *GMI*, which thus ranges between 0 and 16. This *GMI* is associated with the self-image under an endogenous social norm, and is therefore consistent with any warm-glow feelings that individuals may hold when actively taking part in recycling activities (see section 3).

5.1 The Opportunity Cost of Time Spent on Waste Sorting, Cleaning and Transport

The time-use approach assumes that the time used to sort, clean and transport waste represents a pure cost to the individual. In other words, the individual does not by any means derive any direct benefits, or satisfaction, from participating in environmental work. This approach therefore aggregates the time spent by each individual and assigns a monetary value to each hour.

We asked the respondents how much time per week they (as individuals) spend on average on sorting, cleaning and, finally, transporting the waste to the recycling depot. The responses to this question are displayed in Table 4.

Table 4: Respondents' Time Spent on Sorting Waste

How many minutes do you on average use per week for…	Mean (minutes)	Median (minutes)
cleaning?	14	10
sorting and carrying out the sorted waste?	15	10
transporting the sorted waste to central depot?	20	20
Total	49	40

On average each respondent spends a total of 49 minutes each week sorting, cleaning and transporting his or her household waste. Out of this, 14 minutes are used for cleaning, 15 minutes for sorting and carrying, and 20 minutes for transporting the sorted waste to the central reception depot. However, it has been argued that the transporting of waste often is done in conjunction with other activities (e.g., shopping for food), and therefore the resulting transport costs should not be attributed solely to recycling purposes.[13] In this case, the time-use per individual for pure in-house activities of sorting and cleaning only (the first two rows in Table 4) add up to an average of 29 minutes in our sample. In comparison to Bruvoll et al.

[13] For this reason we asked the respondents whether they purposely make trips to the central depot for depositing sorted waste. 37 percent claimed that this was the case "most often/always" and 27 percent did it "sometimes". Only 16 percent claimed that they never did extra trips for the sole purpose of depositing sorted waste.

(2002) that report 23 minutes per week, our figures of time use seem high. One explanation could be that there might not exist a linear relationship between time spent and the amount of waste for every adult to sort. That is, the reported average time spent represent the *household*, not the *individual*. In addition, we find that many respondents reported “round” figures like 5, 10, 15 minutes etc., which indicate some uncertainty and we should hence use caution when analyzing the figures.

By using a labor costs after tax of USD 5.82 per hour, as used in the Radetzki (2000) study, we obtain the total monetary costs per week of the individual time spent on recycling. The results from this exercise are presented in Table 5.

Table 5: Estimates of Time-use Costs, per Individual

	Minutes	**Cost**
Average total time spend	49	USD 4.75
Average time spend on in-house activities	29	USD 2.81

5.2 Households’ Willingness to Pay for Leaving Recycling Activities to Others

In this sub-section we analyze how much the sample households were willing to pay to let someone else take over the waste sorting activities. This willingness to pay (*WTP*) estimate provides an indication of to what extent sorting waste at source is perceived as a cost for the household, i.e., it grants a monetary value to the recycling effort made by individuals. However, the reported value is not only influenced by the time factor. The respondents can, for instance, put a value on not having used milk cartons drying on the sink and/or perceive that the recycling efforts undertaken are a source of increased self-image (see sub-section 3.1). However, even when respondents experience benefits of increased self-image from “saving the environment”, no “free lunch” is involved. The *WTP* measure should hence in this case be interpreted as an indication of the opportunity costs for households to undertake recycling efforts.

In the survey we asked the respondents the following open-ended contingent valuation question:

> Assume that the local authorities decide that you no longer need to sort your waste and that you only need to leave it in the trash can. Hired workers collect the unsorted waste and take care of the waste for further cleaning and sorting. How much would you at most be willing to pay per year for such a system where someone else sorts your waste?

The answers to this valuation question can be divided into four categories: *positive values*, *zeroes*, *blanks* and *question marks*. We received 374 positive and zero values and they are the ones employed in the analysis. Since both blanks and question marks probably correspond to "do not know" or "protest" answers, we cannot draw any inferences about these respondents' *WTP*. Some of these responses, however, may reflect that individuals' moral compasses are not compatible with this type of question. The average willingness to pay for this offer was USD 25 per year. If we look only at those respondents who stated a strictly positive value in response to the hypothetical recycling arrangement (168 people), we obtain an average *WTP* of USD 56.25 per year. If we divide the average *WTP* of all respondents with the average number of hours they spend on recycling activities annually, we get a *WTP* per hour of about 38 US cents.[14] This is substantially lower than the average hourly wages after tax (USD 5.82). Bruvoll and Nyborg (2002) as well as Sterner and Bartelings (1999) find similar results. Sterner and Bartelings estimate through a CVM study the annual *WTP* of a household to recycle at approximately USD 38, and Bruvoll and Nyborg determine the annual *WTP* of households in Norway to pay for others to sort their waste at USD 20. They further find by dividing the average *WTP* by the hours individuals claim to spend on recycling activities, a *WTP* per hour of only about 45 cent, which is very similar to the results find in this present study. This discrepancy between average *WTP* per hour and the average hourly wages after tax can be explained in a number of ways. For instance, the labor market may be inflexible and/or the times reported as devoted to recycling activities may be biased. The relatively low *WTP* estimates may of course also stem from some or all of the factors outlined in section 3, e.g., moral motives. In order to bring some clarity into this matter, we proceed by estimating an econometric model of *WTP*.

5.3 Determinants of *WTP*

This sub-section presents an econometric model, which attempts to explain the individual willingness to pay to be spared from sorting and cleaning the waste generated within ones household. Whilst a simple averaging of open-ended responses provides an estimate of mean *WTP* (section 5.2), bid curves can also be estimated from open-ended contingent valuation data. In essence, this procedure will produce a measure of the willingness to pay for having others sort their waste. This approach facilitates an explanation of the underlying behavior of

[14] If we do this only for those who stated a strictly positive value we get a *WTP* per hour of about 85 US cents.

the respondents, and hence the factors driving the average *WTP* value. We use the following general *WTP* equation for the individual:

$$WTP_i = f(INCOME, GMI, GENDER, AGE, EDUCATION, LIVING, \\ DISTANCE, COMPULSORY, PROFITABLE, PLEASANT) \tag{16}$$

where all variables are defined as in Table 3.

In our case the empirical version of the function in equation (16) may take the form of a censored regression model. Equation (16) will therefore be estimated using a tobit method since our data set is likely to be censored from below. That is, in theory individuals could have compensating demands for letting others sort their waste and hence state negative values. The open-ended approach permits such answers, yet no one stated a negative value. Still, a large proportion of the respondents reported zero *WTP*s, so least squares regression would produce biased and inconsistent estimates and would hence be inappropriate (Greene, 1997). In order to employ the tobit model on equation (16) empirically, we assume a linear function and add an additive stochastic disturbance term, $\boldsymbol{\varepsilon}_{\mathbf{i}}$. The empirical specification of the willingness to pay equation is then given by:

$$\begin{aligned} y_i^* &= \boldsymbol{\beta}'\mathbf{x}_\mathbf{i} + \boldsymbol{\varepsilon}_\mathbf{i} \\ y_i &= 0 \text{ if } y_i^* \leq 0 \\ y_i &= y_i^* \text{ if } y_i^* > 0 \end{aligned} \tag{17}$$

Following the discussion above our *a priori* expectations about the signs of the regression coefficients are as follows. *INCOME* should be positively correlated with *WTP* as richer people face less binding budget constraints than do people with lower income, and could hence pay more for the service. In addition, people with high incomes often tend to have a higher opportunity cost of time, which should imply a higher *WTP* to avoid waste sorting. Respondents with higher *GMI* are expected to have a lower *WTP* due to the decreased self-image following the introduction of a central sorting system; we therefore expect this coefficient to be negative. There are no *a priori* expectations about the signs of the coefficients for *GENDER*, *AGE* and *EDUCATION*. However, Sterner and Bartelings (1999) find that females generally have a higher *WTP*, while education and age where negatively correlated with *WTP*. Regarding the variable *LIVING* we expect that respondents living in apartments are willing to pay more for letting others sort their waste since one of the motives put forward in the literature is that limited space incurs an extra cost for sorting waste, and people living in apartments plausibly perceive an extra burden due to less living area (e.g., Jakus et al, 1997). *DISTANCE*, in turn, we hypothesize to be positively correlated with *WTP*. That is, the further away respondents are from the recycling center the more they are willing to pay for letting others sort and transport their waste. Moreover, the coefficient representing the variable *COMPULSORY* is assumed to have a positive sign. That is, if respondents feel that waste sorting etc. is a requirement imposed by the authorities, their willingness to pay should be higher than if they do not. On the other hand, if respondents feel that they undertake these activities since it is economical for them as a person, their willingness to pay for letting others sort their waste should be lower than for those who do not. The *PROFITABLE* coefficient is thus expected to have a negative sign. Finally, if some respondents feel that recycling activities are pleasant in itself, we expect these to have a lower willingness to pay compared to the ones who do not feel this way. Therefore, the coefficient representing *PLEASANT* is expected to have a negative sign.

Table 6 presents the parameter estimates for the coefficients in the regression model, and the log-likelihood value. Overall the results are consistent with our theoretical expectations, with all coefficients except *PLEASANT* having the expected signs and nearly all being statistically significant. The R^2 value[15] is however low, 0.09, indicating that most of the variation in willingness to pay could not be explained.[16] The estimated mean *WTP* is USD 27.12, which is slightly higher than the mean of the total sample (USD 25).

Direct interpretation of the regression coefficients in a Tobit model is not easy (Gujarati, 1995). However, from the estimated coefficients we can assess marginal effects, i.e., the impact of a change of one standard deviation unit in the value of the regressor on the regressand (Ibid.). We note that some factors in particular tend to be important determinants of willingness to pay. The coefficient representing the impact of the individual's moral compass for recycling activities, *GMI*, has the expected sign, and it is statistically significant

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[15] The econometric literature presents several alternative R^2 measures. We use the squared correlation between the actual and estimated dependent variable, i.e., $R^2 = r^2_{y,\hat{y}}$. For a discussion on different goodness-of-fit measures, see Baltagi (1999).

[16] One should note, however, that low *R*-squares are very common for cross-section samples (Greene, 1993). Furthermore, other studies on the economics of household recycling behavior also frequently report low *R*-squares (e.g., Fullerton and Kinnaman, 2002).

at the 5 percent level. This result suggests that those people who largely undertake recycling activities for moral reasons, do not feel comfortable in buying themselves free from such activities. The coefficient representing *AGE* is statistically significant at the 1 percent level, suggesting that older people are willing to pay less to avoid waste sorting. This is in line with the results of Sterner and Bartelings (1999).

Table 6: Parameter Estimates for the Willingness to Pay Equation

Variable	Coefficient	Marginal Effects
CONSTANT	863.281*** (2.725)	379.719
INCOME	16.621 (0.602)	7.311
GMI	-38.470** (-2.268)	-16.921
GENDER	237.753** (2.155)	104.577
AGE	-18.210*** (-4.265)	-8.009
EDUCATION	109.877 (0.939)	48.330
LIVING	-382.099** (-2.133)	-168.069
DISTANCE	0.021** (2.174)	0.009
COMPULSORY	102.761** (2.140)	45.200
PROFITABLE	-34.918 (-0.679)	-15.359
PLEASANT	59.950 (1.245)	26.369
Log L	-1125.359	
Estimated mean WTP	USD 27.12	

Note: *t*-statistics are given in parentheses. *, **, and *** indicate statistical significance at the ten, five, and one percent level, respectively.

The coefficients representing the variables *GENDER*, *LIVING* and *DISTANCE* are all statistically significant at the 5 percent level. We had no *a priori* expectations about *GENDER* but our results indicate that men have a higher willingness to pay than women (which is in contrast to what Sterner and Bartelings (1999) find). People living in apartments are willing to pay more to let others handle their waste as previously hypothesized. Furthermore, *DISTANCE* we hypothesized to be positively correlated with *WTP*, and this is supported by our model estimation.

The coefficients representing *INCOME* and *EDUCATION* are not statistically significant. The positive sign of the income coefficient is as expected, and higher educated people seem to be more willing to pay for letting someone else sort their waste. As with the income variable the reason for the latter result could be that highly educated people have less leisure time and will hence have a higher opportunity costs for undertaking recycling activities.[17] The variable *COMPULSORY* is statistically significant at the 5 percent level and has the expected sign, meaning that if the respondents feel that sorting at source is a requirement imposed by the authorities, their willingness to pay to avoid this activity is higher than for those who do not feel this way. *PROFITABLE* has the expected sign, while *PLEASANT* has not, but both are insignificant from a statistical point of view so we cannot draw any definite conclusions about these impacts.

To sum up, moral motives seem to matter when evaluating the cost that individuals' assign to their recycling efforts undertaken. Many people feel highly morally committed to recycling activities, and as such they gain benefits (and face thus not only costs) when engaging in such activities. As a consequence, the real costs associated with individuals' recycling efforts tend to be lower than what is indicated when using the time-use approach.

6. Personal Motives and Economic Incentives: Empirical Results

This section analyzes whether economic incentives tend to crowd-out the individuals' intrinsic motivation for recycling, or whether price-based systems rather result in a larger perceived freedom for the individual. In order to find out how the respondents perceive the presence of economic incentives in the waste management field (here represented by a pay-by-the-bag system) we confronted them with the following scenario:

> Some municipalities in Sweden have introduced a new system to finance the municipality's waste handling costs. Such a system is designed so that the households themselves choose how much they want to sort at source, and the local authority then weighs the non-sorted waste.

[17] It should be noted that the income and education variables are not highly correlated. In our sample the correlation coefficient between the two variables is 0.31.

Households then pay per kilo waste they have chosen not to sort. Thus, households that sort more pay a lower total fee than those who sort less.

We then let the respondents specify to what extent they agreed with a number of statements related to this pay-by-the-bag system. The current waste handling system in the municipality of Piteå is not weight-based in this sense but relies instead on a fixed monthly fee. Table 7 summarizes the responses to these questions.

Table 7: Attitudes Towards Priced-based Waste Management Policy

In my opinion such a price-based system is:	Agree	Partly agree	Partly disagree	Disagree	No conception	No answer
a) Good, since it gives me a greater ”freedom” in sorting. That is, I can choose the amounts that I wish to sort while I can dispose of the rest and pay for it.	23%	12%	6%	27%	14%	18%
b) Bad, because I wish to choose to sort myself, and not be “forced” to do so by paying a fee.	36%	9%	8%	13%	13%	21%
c) Good, since those who sort little are “punished” in the form of higher fees.	29%	12%	10%	13%	16%	20%
d) Good, since it provides a clearer economic motive for sorting.	31%	14%	8%	15%	15%	17%
e) Bad, since it decreases my motivation for sorting more.	18%	8%	8%	26%	16%	24%
f) It is a public duty to sort and sorting should be encouraged in other ways than through financial “carrots”.	37%	11%	10%	12%	13%	17%

It is worth noting that the respondents are spread “all over the field” with respect to most of the statements. For instance, regarding statement (a), 35 percent agreed or partly agreed to the view that this price-based system would give them more freedom since they then can choose which fractions to sort (if any) and haul the rest. Almost exactly the same share (34 percent) does not agree (or disagree partly) to this, indicating that the sample population is heterogeneous with respect to this question. The same story goes for statement (e), i.e., that a price-based system would diminish their motivation to sort, with 26 percent agreeing with the statement and 34 percent not. In sum, we gain some support for the hypotheses that a price-based system can induce both a sense of more freedom in recycling activities as well as crowd out any intrinsic motivation.

However, in order to explore this issue in more detail we attempt at testing two specific hypotheses, both related to statements (a) and (e) above. These hypotheses are:

- Respondents that sort waste because they consider it to be a requirement imposed on them by the authorities are more likely to be positive towards the introduction of a pay-by-the-bag scheme. The new system provides these persons with more flexibility, as it is now legitimate to sort less as long as you pay for the non-sorted waste.
- Respondents that feel intrinsically motivated to sort waste are more likely to be negative towards a pay-by-the-bag scheme since it crowds out their motivation and does not induce them to sort more (perhaps even less). This hypothesis is thus consistent with the postulates of the so-called crowding theory.

We use two different equations to test these hypotheses, one “traditional” and one “crowd out”, with different dependent variables in each but with the same set of explanatory variables:

$$\begin{aligned} \textit{Dependent variable}\,(0/1) = f(&\textit{GENDER}, \textit{AGE}, \textit{EDUCATION}, \textit{INCOME}, \\ &\textit{COMPULSORY}, \textit{PLEASANT}, \\ &\textit{DO WHAT OTHERS SHOULD DO}) \end{aligned} \tag{18}$$

Thus, to test our hypotheses we estimated two binary logit models, where the responses to statements (a) and (e) in Table 7 serve as the dependent variables. Statement (a) tries to capture the “traditional” view that economic incentives create more freedom in respondents’ recycling behavior, while statement (e) captures the “crowding out” effect. We hypothesize that the responses to statements (a) and (e) to be influenced by socio-economic factors, and by individual motives for undertaking recycling activities. Specifically, we consider three motives to sorting waste at source: *COMPULSORY*, *DO WHAT OTHERS SHOULD DO* and *PLEASANT* (see Table 3 for definitions). These potential determinants of the attitudes towards a pay-by-the-bag system are scrutinized by estimating the following binominal logit model for each of the two alternative models:[18]

[18] The logistic distribution is similar to the normal distribution except in the tails, which are heavier than in the normal distribution. For a comprehensive discussion on logit models, see Greene (1993) and Maddala (1983).

$$P[Y=1]=\frac{e^{\boldsymbol{\beta}'\mathbf{X}}}{1+e^{\boldsymbol{\beta}'\mathbf{X}}}=\Omega(\boldsymbol{\beta}'\mathbf{X}) \tag{19}$$

where $\boldsymbol{\beta}$ is a vector of parameters to be estimated. When creating the "traditional" dependent variable we categorized the degree to which the respondents agreed to the statement: "Good because it gives me more freedom. I can choose what fractions of the waste I want to sort and haul the rest and pay for it." The responses "Agree" and "Partly agree" were grouped together and assigned a one (1), and "Partly disagree" and "Disagree" a zero (0). The response "No conception" was omitted from the sample since we cannot draw any valid inferences from this. When creating the "crowding out" dependent variable we used the responses to the statement: "Bad since it diminishes my motivation to sort more." The responses "Agree" and "Partly agree" were assigned a one, while "Partly disagree" and "Disagree" were assigned a zero. Again, "No conception" was omitted from the sample.[19]

Employing the same set of variables as explanatory variables in each of the two regression models, we have no *a priori* expectations about the impact of the socio-demographic variables *GENDER*, *AGE*, *EDUCATION* and *INCOME* in neither model. However, for the "traditional" equation we expect that respondents who sort because they feel it is a requirement imposed by authorities should experience a larger "freedom" if a pay-by-the-bag scheme is introduced, and they are thus likely to be positive towards such a system. However, respondents who sort because they feel that they should contribute to recycling to the same extent that they expect others to contribute, are expected to express a negative attitude towards statement (a). One reason for this would be that these people are likely to be of the opinion that one should not be able to "buy oneself free" from an activity that is considered to be a public duty. In the case of respondents who sort because they think it is a pleasant activity in itself, it is harder to form any *a priori* expectations about their attitude. On the one hand we could expect these people to be positive towards the introduction of a pay-by-the-bag scheme. People who enjoy sorting waste are likely to be active recyclers, and they thus probably gain financially from this since they will pay comparatively low waste management fees. On the other hand, however, the introduction of such a system may (for some) also crowd-out any intrinsic motivation to sort.

[19] Since "no conception" responses were removed, 321 and 227 observations, respectively, were used in the final regression estimations.

For our “crowd out” equation we expect that respondents who sort because they feel it is a requirement imposed by the authorities, will experience that their motivation is not diminished by a pay-by-the-bag scheme. On the contrary, these people will rather experience increased motivation since they can earn money by recycling. Furthermore, if the respondents’ sort because they feel that they should do what they think others should do, their motivation should be reduced by such a scheme. Again, this reflects a situation in which externally introduced incentives “crowd-out” intrinsic motivation. Similarly, if respondents sort because it is a pleasant activity in itself we may expect their motivation to diminish with the introduction of economic incentives. However, as in the “traditional” sense, people who enjoy sorting waste are likely to be active recyclers, and they thus probably gain financially from this since they will face comparatively low waste management costs, and they may thus not necessarily feel any destruction of their intrinsic motivation to sort.

Table 8 presents the parameter estimates for the two binary logit regression models, as well as the corresponding log-likelihood measures. The third and fifth column of Table 8 presents the marginal effects of a change in any independent variable on the probability that a respondent agrees (or partly agrees) with the statement under investigation. For instance, the probability that a respondent agrees with the statement “Good because it gives me more freedom. I can choose what fractions of the waste I want to sort and haul the rest and pay for it” is estimated to decrease by about 0.5 percent for an extra year of age of the respondents and increase 3.8 percent for a 1-point increase in the percentage of respondents with university degrees.

Likelihood ratio tests of the hypothesis that all coefficients are zero were conducted for both models, and they resulted in chi-squared values of 21.59 and 28.24, respectively. With 7 degrees of freedom the critical value of chi-squared at 1% significance level is 18.47. Thus, it is possible to reject the joint hypothesis that all coefficients are zero in both estimations.

We begin by analyzing the results from the “traditional” model (the two left columns in Table 8). Regarding our categorization of individuals’ perceived, increased freedom following the introduction of a pay-by-the-bag recycling scheme we note that only one socioeconomic factor tends to be an important determinant. The older the respondent is the more likely is it that he/she is negative towards a pay-by-the-bag scheme, and this effect is statistically significant at the 5 percent level. The coefficients representing the variables *GENDER*, *EDUCATION* and *INCOME* are however not statistically significant.

Table 8: Parameter Estimates for the Binary Crowding Out Variable

Variable	"Traditional" Coefficient (t-statistic)	"Traditional" Marginal Effects	"Crowd out" Coefficient (t-statistic)	"Crowd out" Marginal Effects
CONSTANT	1.603** (2.082)		-1.655 (-1.874)	
GENDER	-0.381 (-1.533)	-0.088	-0.169 (-0.580)	-0.042
AGE	-0.024** (-2.453)	-0.005	0.031*** (2.831)	0.008
EDUCATION	0.169 (0.605)	0.038	-0.724** (-2.150)	-0.178
INCOME	-0.062 (-0.974)	-0.014	0.178** (2.301)	0.044
COMPULSORY	0.364*** (3.370)	0.084	-0.255* (-1.721)	-0.063
DO WHAT OTHERS SHOULD DO	-0.239* (-1.902)	-0.055	-0.082 (-0.608)	-0.020
PLEASANT	0.149 (1.431)	0.034	0.292** (2.507)	0.072
Summary Statistics				
Number of observations	321		227	
Log likelihood	-200.32		-142.59	
Restricted log likelihood	-211.11		-156.71	
Chi-squared (7 d.f.)	21.59***		28.24***	

Note: *t*-statistics are given in parentheses. *, **, and *** indicate statistical significance at the ten, five, and one percent level, respectively.

As expected the coefficient representing *COMPULSORY* is statistically significant at the 1 percent level, i.e., people who sort because that is something the authorities force them to do feel an increased freedom in their recycling activities if a pay-by-the-bag scheme is introduced. The coefficient representing *DO WHAT OTHERS SHOULD DO* is negative and statistically significant at the 10 percent level, indicating that respondents who take part in recycling activities since they want to do what others should do, do not subscribe to the statement that a pay-by-the-bag system creates more freedom in their sorting behavior. According to these people, recycling is a public duty and not so much a matter of individual choice. Finally, there is no clear relationship between respondents who undertake recycling activities because it is a pleasant activity in itself (*PLEASANT*) and the perceived increased freedom the monetary incentives introduced to them would generate. The sign of the coefficient, however, indicates that respondents who enjoy sorting waste perceive a pay-by-the-bag system as positive. The explanation could be as outlined in section 3.2; their efforts simply result in a reduced waste bill.

When considering the “crowd-out” model we first note that three socio-economic factors tend to be important determinants (the two right columns in Table 8). Older people feel a bigger destruction of their motivation following a pay-by-the-bag system than do younger ones, and this effect is statistically significant at the 1 percent level. This is not a surprising finding; North (1990) finds that different generations form different institutional arrangements (e.g., social norms and rules). In this specific case older people are not in favor of traditional economic thought while younger ones subscribe to both increased flexibility and less perceived destruction of intrinsic motivation. The coefficient representing *EDUCATION* is statistically significant at the 5 percent level, and reveals that higher educated people tend to feel less destruction of their motivation than do those with lower education. The marginal effect of a 1-point increase in the percentage of respondents with university degrees is estimated to decrease the probability that the respondent felt less motivated by about 18 percent. One explanation for this might be that higher educated people work more and will thus have a higher opportunity costs for leisure and will then find it convenient to pay-by-the-bag and let someone else sort their waste. The coefficient for *INCOME* is statistically significant at the 5 percent level and has a positive sign, which suggests that people with higher income feel less destruction of their motivation. Moreover, people who sort because that is something the authorities “force them” to, do not feel less motivated to recycling activities if a pay-by-the-bag scheme is introduced. This finding is statistically significant at the 10 percent level. Furthermore, the coefficient representing *PLEASANT* is statistically significant at 5 percent level, indicating that people that feel that recycling is a pleasant activity in itself feel less motivated by a pay-by-the-bag scheme than people who do not. We *a priori* had no clear conjecture about this impact, but it seems that our empirical results is in line with the postulates of the so-called crowding theory. That is, that monetary incentives crowd out respondents intrinsic motivation to sort.

In sum, we do find some support for both of our hypotheses. *First*, economic incentives, represented by a pay-by-the-bag recycling scheme, create more freedom for respondents that previously felt it compulsory to sort. *Second*, monetary incentives tend to crowd-out individuals’ intrinsic motivation for recycling. It is important to note that this was a first attempt to empirically test the hypothesis of crowding out behavior in the field of recycling. Nevertheless, the findings in this section do point to the fact that intrinsic motivation matter for individuals when undertaking recycling activities, and this has important implication for recycling policy. Still, equally important is the finding that other individuals perceive a price-by-the-bag system as a flexible one making room for more freedom in their everyday lives. However, this means that the respondents are heterogeneous in their view of recycling efforts, which may be troubling for policymakers in their choice of policy instruments to change behavior in an intended way.

7. Concluding Remarks

This paper has used original data gathered from individuals to analyze: (a) to what extent motives for undertaking recycling activities matter for the assessment of households' recycling costs; and (b) the role of motives in determining the efficiency of introducing economic incentives for recycling efforts. Specifically, for the former purpose, we compared the average hourly willingness to pay to let others sort household waste at source with the corresponding income after tax (i.e., the opportunity cost of time); and used an econometric model, which attempted to explain the individual willingness to pay to let someone else take over the time-consuming waste sorting activities. We find that the average hourly willingness to pay to let others sort household waste at source was significantly lower than the corresponding income after tax. Furthermore, the perception of ones self-image and moral concerns can explain some of this finding. Specifically, we find that individuals' moral compass for recycling activities, expressed by the green moral index, is a major determinant of individual's willingness to pay for evading sorting waste at source. In addition, peoples' view on whether sorting at source is a requirement imposed by the authorities or not does also play an important role for respondents *WTP*. To sum up, since households often express moral motives for many environmental activities (some even find it a pleasant activity in itself) it is not convincingly the case that one should place a cost on all activities involved.

For the latter purpose we tested the hypotheses that economic incentives in recycling schemes may create more freedom for respondents that previously felt it compulsory to sort, and that monetary incentives may crowd-out the individual's intrinsic motivation for recycling. Again we find support for both these hypotheses in our empirical analysis. We find in particular that those who sort waste because it is a requirement imposed on them by the authorities have a positive perception of such as system since it permits a larger degree of flexibility in recycling efforts. In addition, those who initially feel an intrinsic motivation to sort waste at source are more inclined to be discouraged by the introduction of a pay-by-the-bag waste management scheme. Apart from personal motives we also find that age where of relevance in determining the respondents' view of perceived "freedom" as well as crowding out intrinsic motivation with the introduction of a pay-by-the-bag scheme. In sum, older people go against the grain in traditional economic thought while younger ones subscribe to both increased flexibility and that they do not feel their motivation to be crowded out by the economic incentives introduced to them.

The above gives rise to an important policy implication; motives do matter for recycling activities in general. The population is heterogeneous in their view of recycling efforts, i.e., there is no single *Recycling Man* out there to be guided. Hence, policies aimed at changing recycling behavior are not straightforward. Some individuals have learned to appreciate the reward of economic incentives, while some feel that environmental morale is crowded out by the same means. This insight is vital if the use of such instruments is to be expanded. The goal for policymakers will thus be to find a way of guiding both categories of people in their strive for a sustainable society. However, our research also indicates that in the future economic incentives may become relatively more effective than today due to the younger generation's perception of such means.

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Appendix A to Paper 4: Letter and Questionnaire

Swedish Original:

PITEÅ KOMMUN
Miljö- och Byggkontoret

LULEÅ UNIVERSITY OF TECHNOLOGY

Luleå, 16 februari 2002

Till Er som källsorterar eller inte källsorterar,

Piteå Kommun genomför, tillsammans med Luleå tekniska universitet, ett forskningsprojekt som handlar om hushållens inställning till källsortering och återvinning. Sådan kunskap är viktig för att göra avfallshanteringen i kommunen mer effektiv.

Ni är en av de 850 hushåll som genom ett slumpmässigt urval har utsetts att besvara denna enkät. **Vi hoppas att du vill hjälpa oss genom att ta dig tid att fylla i enkäten och vi är tacksamma om du så snart som möjligt besvarar det frågeformulär som du fått och återsänder det i det frankerade kuvertet.** Var god och svara inom två veckor.

I en vetenskaplig undersökning som denna är det viktigt att människor med olika uppfattning får tillfälle att delta, även de som kanske inte har ett direkt intresse av källsortering. **Därför är det är av yttersta vikt att även Ni som inte källsorterar och/eller är negativt inställd till källsortering också tar er tid att fylla i enkäten.** Naturligtvis behandlas ditt svar helt konfidentiellt, numret längst upp till höger på frågeformuläret är endast till för att vi inte ska påminna dem som redan svarat.

Om ni vill fråga om något i formuläret eller om undersökningen i allmänhet, tveka inte att kontakta **Christer Berglund** på telefon 0920-492977 eller via e-post: Christer.Berglund@ies.luth.se.

Ett stort tack på förhand för din medverkan!

Med vänliga hälsningar,

Christer Berglund	*Ulf Hedman*	*Per-Erik Granlund*
Fil.Lic.	*Miljö- & byggchef*	*VD Renhållningen*
Luleå tekniska universitet	*Piteå Kommun*	*Piteå Kommun*

Frågeformulär

Nedan följer frågor rörande källsortering i hemmet. Frågorna avser sortering, rengöring och frakt av avfall ämnat för återvinning. Vi avser här inte avfall som ni i hushållet sorterar för eget bruk, exempelvis papper till kaminen eller avfall till den egna komposten.

1. Källsorterar du allt, mestadels, något eller inget av följande typer av avfall? Ringa in lämpligt alternativ.

Brännbart avfall.

4	3	2	1
Allt	Mestadels	Något	Inget

Matavfall/kompost.

4	3	2	1
Allt	Mestadels	Något	Inget

Pappersförpackningar.

4	3	2	1
Allt	Mestadels	Något	Inget

Plastförpackningar utan returpant.

4	3	2	1
Allt	Mestadels	Något	Inget

Glasförpackningar utan returpant.

4	3	2	1
Allt	Mestadels	Något	Inget

Metallförpackningar utan returpant.

4	3	2	1
Allt	Mestadels	Något	Inget

Papper i tidningar och wellpapp.

4	3	2	1
Allt	Mestadels	Något	Inget

Farligt avfall (ex. olja, lösningsmedel, medicin).

4	3	2	1
Allt	Mestadels	Något	Inget

Batterier.

4	3	2	1
Allt	Mestadels	Något	Inget

Elektriskt/elektroniskt avfall (ex. glödlampor, TV, leksaker med elektronik).

4	3	2	1
Allt	Mestadels	Något	Inget

Om du svarat ”inget (1)” på alla ovanstående alternativ, gå direkt till fråga 11 på sidan 4

2. **I vilken utsträckning håller du med om nedanstående påståenden? Ringa in lämpligt alternativ.**

Jag källsorterar därför att….

a) det är något som myndigheterna (stat, kommun) ålagt mig att göra.

4	3	2	1	0
Instämmer helt			Instämmer inte alls	Har ingen uppfattning

b) jag vill att andra människor skall uppfatta mig som en ansvarsfull person.

4	3	2	1	0
Instämmer helt			Instämmer inte alls	Har ingen uppfattning

c) jag vill se mig själv som en ansvarsfull person.

4	3	2	1	0
Instämmer helt			Instämmer inte alls	Har ingen uppfattning

d) jag bör själv göra sådant som jag anser att andra bör göra.

4	3	2	1	0
Instämmer helt			Instämmer inte alls	Har ingen uppfattning

e) källsortering är en önskvärd aktivitet i *sig själv* som skänker mig tillfredsställelse.

4	3	2	1	0
Instämmer helt			Instämmer inte alls	Har ingen uppfattning

f) källsortering gynnar miljön, och jag vill bidra till en bättre miljö i Sverige.

4	3	2	1	0
Instämmer helt			Instämmer inte alls	Har ingen uppfattning

g) källsortering är gynnsamt för den svenska ekonomin som helhet, och jag vill bidra till detta.

4	3	2	1	0
Instämmer helt			Instämmer inte alls	Har ingen uppfattning

h) min källsortering gynnar mitt hushålls privata ekonomi.

4	3	2	1	0
Instämmer helt			Instämmer inte alls	Har ingen uppfattning

3. **Hur upplever du tillgängligheten att lämna grovt avfall, farligt avfall, trädgårdsavfall, el-avfall, restavfall, etc. på Bredviksbergets återvinningscentral avseende…**

a) öppettider?

4	3	2	1	0
Bra	Mindre bra	Ganska dåligt	Dåligt	Har ingen uppfattning

b) avstånd?

4	3	2	1	0
Bra	Mindre bra	Ganska dåligt	Dåligt	Har ingen uppfattning

Eventuella kommentarer: ____
____

4. Hur lämnar du ditt restavfall (ex. keramik, porslin, säkringar, kastruller, etc.)?

☐ Lämnar själv (Bredviksberget)
☐ Till sopbilen (2 ggr per år för villa/särskilt kärl för lägenheter)
☐ I brännbart och/eller komposterbart kärl hemma
☐ Annat,_____________

5. Ungefär hur många <u>minuter</u> spenderar du i genomsnitt <u>under en vecka</u> till:

- att rengöra källsorterat avfall? ________minuter
- att sortera och bära ut sorterat avfall i hemmet? ________minuter
- att frakta källsorterat avfall till återvinningsstationer? ________minuter

6. Hur långt har du till närmaste återvinningsstation?

_____________meter

7. Åker du extra/enkom resor till återvinningsstationen för att lämna sorterat avfall?

☐ Ja, oftast eller alltid ☐ Ja, ibland ☐ Nja, någon gång ☐ Nej, aldrig

8. Hur upplever du din återvinningsstation avseende utformning och färg?

☐ Snygg ☐ OK ☐ Ful ☐ Ingen uppfattning

Eventuella kommentarer: ____
____

9. Hur upplever du din återvinningsstation avseende städning/renlighet?

4	3	2	1	0
Bra	Mindre bra	Ganska dåligt	Dåligt	Har ingen uppfattning

10. Händer det att du får ta hem/lämna utanför stationen för att behållaren är full?

☐ Ofta ☐ Sällan ☐ Aldrig

11. Idag är det hämtning var fjärde vecka av <u>brännbart</u> avfall. Om kommunen istället införde en generell 14-dagarshämtning (dvs. hämtning varannan vecka), skulle du vara villig att betala 350 kronor/år istället för nuvarande 288kr/år för den servicen?

☐ Ja ☐ Nej ☐ Vet ej

12. Idag är det hämtning varannan vecka av det <u>komposterbara</u> avfallet. Om kommunen införde en generell veckohämtning (dvs. hämtning varje vecka) av det komposterbara under *sommartid (juni, juli och augusti)*, skulle du vara villig att betala 200 kronor <u>extra</u> per år för den servicen utöver de 371kr/år ni betalar nu?

☐ Ja ☐ Nej ☐ Har ingen hämtning, komposterar själv ☐ Vet ej

Nedan följer några frågor rörande alternativa avfallshanteringssystem till det som finns i Piteå idag. Kom ihåg när ni besvarar dessa frågor att ni inte skulle ha den nuvarande kostnaden för hämtning etc. eftersom dessa alternativa system <u>helt</u> skulle kunna ersätta det nuvarande. I nuläget finns det dock inte några planer på att ändra systemet i Piteå utan frågorna nedan är ”bara” till för att få information om hushållens allmänna inställning till källsortering.

13. Om du mot en avgift på 400 kronor/år fick möjlighet till hämtning av utsorterade förpackningar (som du sorterat själv) vid ditt hushåll som skulle ersätta nuvarande återvinningsstationer, skulle du anta detta erbjudande?

☐ Ja ☐ Nej ☐ Vet ej

14. Antag att kommunen beslutar att hushållen inte längre behöver källsortera själva, utan att de kan lämna sitt osorterade avfall direkt i soprummet/soptunnan. Kommunanställda eller inhyrda arbetare tar sedan hand om upphämtning och sortering. Hur mycket skulle ditt hushåll <u>maximalt</u> kunna tänka sig att betala <u>per år</u> för att någon annan källsorterar ert hushållsavfall enligt ett sådant system?

Kronor per år:________

15. Vilket/vilka av följande påståenden förklarar bäst hur du resonerade när du besvarade fråga 14? Kryssa i ett eller flera alternativ.

☐ Jag tycker att det är min plikt att källsortera och tycker inte att man skall kunna ”köpa sig fri” från det ansvaret.
☐ Jag har inte råd att betala något mer/mycket mer för att slippa källsortera.
☐ Jag är inte intresserad av källsortering.
☐ Avfall är något personligt och jag vill inte att någon annan ska sortera mitt avfall.
☐ Jag betalar gärna för att någon annan källsorterar åt mig. Då får jag mer tid att göra annat.
☐ Annat, nämligen____

16. I vissa kommuner i Sverige har man infört ett nytt betalningssystem för att bekosta kommunens avfallshanteringskostnader. Ett sådant system innebär att hushållen själva väljer hur mycket de vill sortera, men kommunen väger alltid det icke-sorterade avfallet. Hushållen får sedan betala en avgift per kilo efter hur mycket de har valt att inte sortera. Med andra ord, de hushåll som sorterar mer får en lägre sophämtningskostnad än de som väljer att sortera mindre. I vilken utsträckning håller du med om följande påståenden om ett sådant viktbaserat system?

Jag tycker att ett sådant system är….

a) bra för det ger mig en större ”frihet” i källsorteringen. Det vill säga, jag kan själv välja vilka fraktioner av avfallet som jag vill sortera medan jag kan slänga resten och betala för det.

4	3	2	1	0
Instämmer helt			Instämmer inte alls	Har ingen uppfattning

b) dåligt för att jag vill själv välja att sortera och inte bli ”tvingad” i form av avgifter.

4	3	2	1	0
Instämmer helt			Instämmer inte alls	Har ingen uppfattning

c) bra för de som slarvar med sorteringen ”bestraffas” i form av högre avgifter.

4	3	2	1	0
Instämmer helt			Instämmer inte alls	Har ingen uppfattning

d) bra eftersom det ger ett tydligare ekonomiskt skäl till att källsortera..

4	3	2	1	0
Instämmer helt			Instämmer inte alls	Har ingen uppfattning

e) dåligt eftersom det minskar min motivation till att källsortera mer.

4	3	2	1	0
Instämmer helt			Instämmer inte alls	Har ingen uppfattning

f) det är en samhällelig plikt att källsortera och källsortering bör uppmuntras på andra sätt och inte genom ekonomiska ”morötter”.

4	3	2	1	0
Instämmer helt			Instämmer inte alls	Har ingen uppfattning

17. Antag att kommunen skulle bestämma sig för att bemanna återvinningsstationerna för att säkerhetsställa att sorteringen sker på ”rätt sätt”, dvs. att rätt avfall läggs i rätt behållare. Hur skulle du uppleva en sådan bemanning?

Det….

a) skulle vara bra för då får man hjälp med att lägga rätt saker på rätt ställe.

4	3	2	1	0
Instämmer helt			Instämmer inte alls	Har ingen uppfattning

b) skulle kännas som ett kontrollerande av mina vanor från myndigheternas sida.

4	3	2	1	0
Instämmer helt			Instämmer inte alls	Har ingen uppfattning

c) skulle motivera mig att källsortera mer än tidigare.

4	3	2	1	0
Instämmer helt			Instämmer inte alls	Har ingen uppfattning

18. Tycker du att du är tillräckligt informerad om gällande taxor och bestämmelser för avfallshanteringen i Piteå Kommun?

- ☐ Ja, tillräckligt informerad.
- ☐ Nej, ej tillräckligt informerad.
- ☐ Vet ej.

Eventuell kommentar:____

19. Det jag vet om Renhållningen i Piteå har jag fått genom (fler alternativ än ett kan kryssas för)…

- ☐ kontakt med Renhållningen
- ☐ kontakt med Miljö och Bygg
- ☐ Internet
- ☐ media
- ☐ informationsblad från kommunen
- ☐ granne/kamrater/etc.

20. Har du varit i kontakt med Renhållningen i Piteå om frågor rörande sophämtning/avfallshantering under de senaste 12 månaderna?

- ☐ Ja.
- ☐ Nej.

21. Var du nöjd/missnöjd med denna kontakt?

- ☐ Nöjd.
- ☐ Missnöjd.
- ☐ Varken nöjd eller missnöjd.
- ☐ Vet ej.

B: Bakgrundsfakta

22. Är du kvinna eller man?

☐ Kvinna ☐ Man

23. Hur gammal är du?

_______ år

24. Vilken utbildning har du?
(kryssa endast i ett alternativ)

☐ Grundskole- eller folkskolebildning
☐ Gymnasieutbildning
☐ Folkhögskoleutbildning
☐ Högskole- eller universitetsutbildning
☐ Annan, nämligen_________________________

25. Vilken typ av boende har du?

☐ Villa, radhus, kedjehus.
☐ Lägenhet
☐ Annat, nämligen:__________________

26. Ungefär hur stor är ditt hushålls sammanlagda inkomst per månad, före skatt?
(Inkludera även inkomster, såsom exempelvis eventuell sjukpenning, föräldrapenning, studiemedel eller arbetslöshetsersättning)

☐ Mindre än 5 000 kronor
☐ Mellan 5 001 och 10 000 kronor
☐ Mellan 10 001 och 15 000 kronor
☐ Mellan 15 001 och 20 000 kronor
☐ Mellan 20 001 och 25 000 kronor
☐ Mellan 25 001 och 30 001 kronor
☐ Mellan 30 001 och 40 000 kronor
☐ Mellan 40 001 och 50 000 kronor
☐ Mellan 50 001 och 60 000 kronor
☐ Mellan än 60 001 och 70 000 kronor
☐ Mer än 70 000 kronor

C: Övriga kommentarer

Tack för din hjälp –Här finns plats för dina egna kommentarer om du tycker det finns något annat viktigt att tillägga om avfallshanteringen i Piteå, det här frågeformuläret eller något annat rörande den här studien.

English translation:

PITEÅ KOMMUN
Miljö- och Byggkontoret

LULEÅ
UNIVERSITY
OF TECHNOLOGY

Luleå February 16, 2002

To you who sort or do not sort waste at source,

The municipality of Piteå is together with Luleå University of Technology carrying out a survey into people's attitudes towards households' recycling activities. Improved knowledge about these attitudes is important for the design of efficient policies.

You are one out of 850 households that have been randomly selected to answer this survey. **We hope that you will take time to fill out this survey and we would be grateful if you could fill in the questionnaire and return it to us in the enclosed pre-paid envelope as soon as possible**. Please respond within two weeks.

In scientific investigations of this kind it is important that people with different attitudes express their views; that is, also those who have little or no interest in sorting and recycling activities. **It is therefore important that also you who do not sort waste at source and/or are unenthusiastic about recycling activities take time to fill in the questionnaire.** The answers that you give will be completely confidential and anonymous. It will not be possible to distinguish what you or any other respondent have answered from the results that will be published.

If there is anything in the questionnaire that is unclear or if you have any questions about the project, please feel free to contact **Christer Berglund,** telephone 0920-492977 or via e-mail: Christer.Berglund@ies.luth.se.

Thanking you in advance for participating!

Yours sincerely,

Christer Berglund	*Ulf Hedman*	*Per-Erik Granlund*
Fil.Lic.	*Miljö- & byggchef*	*VD Renhållningen*
Luleå University of tech.	*Piteå Kommun*	*Piteå Kommun*

Questionnaire

Below you will find questions regarding waste sorting activities at source. Here we are not interested in waste that you possibly sort for your own use, for example, paper for use in your fireplace, etc.

1. Do you sort all, mostly, some or none of the following type of waste fractions? Circle the appropriate alternative.

Combustible waste.

4	3	2	1
All	Mostly	Some	None

Food waste/compost.

4	3	2	1
All	Mostly	Some	None

Paper covering.

4	3	2	1
All	Mostly	Some	None

Plastic covering without refund.

4	3	2	1
All	Mostly	Some	None

Glass without refund.

4	3	2	1
All	Mostly	Some	None

Metal covering without refund.

4	3	2	1
All	Mostly	Some	None

Newsprint and corrugated board.

4	3	2	1
All	Mostly	Some	None

Hazardous waste (e.g., oil, solvent, medicine).

4	3	2	1
All	Mostly	Some	None

Batteries.

4	3	2	1
All	Mostly	Some	None

Electric/electronic waste (e.g., light bulbs, TV, toys with electronic parts).

4	3	2	1
All	Mostly	Some	None

If you answered ”none (1)” on all of the above questions, continue directly to question 11 on page 4.

2. How well do you agree with the following statements? Circle the appropriate alternative.

I sort because….

a) it is a requirement imposed by the authorities (state or municipal).

4	3	2	1	0
Agree			Disagree	No conception

b) I want others to think of me as a responsible person.

4	3	2	1	0
Agree			Disagree	No conception

c) I want to think of myself as a responsible person.

4	3	2	1	0
Agree			Disagree	No conception

d) I should do what I want others to do.

4	3	2	1	0
Agree			Disagree	No conception

e) it is a pleasant activity *in itself* that brings me satisfaction.

4	3	2	1	0
Agree			Disagree	No conception

f) I want to contribute to a better environment.

4	3	2	1	0
Agree			Disagree	No conception

g) it is economical for the society at large.

4	3	2	1	0
Agree			Disagree	No conception

h) it is economical for me as a person.

4	3	2	1	0
Agree			Disagree	No conception

3. How do you experience the availability of disposing heavy waste, hazardous waste, garden waste, electronic waste etc. at Bredviksbergets depot regarding…

a) opening hours?

4	3	2	1	0
Well	Fairly well	Fairly poor	Poor	No conception

b) distance?

4	3	2	1	0
Well	Fairly well	Fairly Poor	Poor	No conception

Comments: ____

____

4. How do you dispose of your "other waste" (e.g. ceramics, porcelain, fuses, pots, etc.)?

☐ By myself (Bredviksberget)
☐ To the garbage truck (2 times a year for houses/specific containers for apartments)
☐ In container for combustible and/or in container for composting at home
☐ Other way, namely:_____________

5. Approximately how many <u>minutes</u> do you on average spend <u>per week</u> for:

- cleaning sorted waste? ________minutes
- sorting and carrying out the sorted waste? ________minutes
- transporting sorted waste to a central depot? ________minutes

6. How far is it from your home to the nearest situated central depot?

_____________meters

7. Do you often travel extra/purposely to the central depot for depositing sorted waste?

☐ Yes, often/always ☐ Yes, sometimes ☐ Only once in a while ☐ No, never

8. How do you find your central depot regarding color and shape?

☐ Pretty ☐ OK ☐ Ugly ☐ No conception

Comments: ____

____

9. How do you find your central depot regarding cleaning/neatness?

4	3	2	1	0
Well	Fairly well	Fairly poor	Poor	No conception

10. Is it often the case that you have to return the waste or leave it outside because the container is full?

☐ Often ☐ Rarely ☐ Never

11. Today you have pick-up of combustible waste every fourth week. If the municipality introduced a two-week interval instead, would you be willing to pay SEK 350/year instead of the current SEK 288/year for this service?

☐ Yes ☐ No ☐ Don't know

12. Today you have pick-up of compost waste every second week. If the municipality instead introduced a weekly interval of this waste during the summer (June, July and August), would you be willing to pay an <u>additional</u> SEK 200/year on top of the current SEK 371/year?

☐ Yes ☐ No ☐ Have own compost ☐ Don't know

Below you find questions regarding alternative waste management systems to the one we have today in Piteå. When you answer these questions remember that you would not have the current charges for pick-up, etc. since these system would <u>fully replace</u> the prevailing system. As of today there are no plans of changing the current system, so these hypothetical questions ”only” help us to gain information about your attitudes toward waste management.

13. If the municipality offered you to pick up sorted waste (that you had sorted) at your household for a fee of SEK 400/year, thus replacing the current recycling centers, would you accept this offer?

☐ Yes ☐ No ☐ Don't know

14. Assume that the local authorities decide that you no longer need to sort your waste and that you only need to leave it in the trash can. Hired workers collect the unsorted waste and take care of the waste for further cleaning and sorting. How much would you <u>at most</u> be willing to pay <u>per year</u> for such a system where someone else sorts your waste?

SEK per annum:________

15. Which of the following statements best explains the reasoning behind your choice in question 14? Tick appropriate boxes.

☐ I think that it is my duty to sort and I don't believe that one should be able to "buy one selves free" from that responsibility.
☐ I can't afford to pay more/much more in order to be relieved from sorting.
☐ I am not interested in sorting
☐ Waste is something personal and I do not want anyone else to sort my waste.
☐ I will gladly pay for someone else to sort for me. Then I would have more time for other things.
☐ Other, namely: ____

16. Some municipalities in Sweden have introduced a new system to finance the municipality's waste handling costs. Such a system is designed so that the households themselves choose how much they want to sort at source, and the local authority then weighs the non-sorted waste. Households then pay per kilo waste they have chosen not to sort. Thus, households that sort more pay a lower total fee than those who sort less. To what extent do you agree with the following statements regarding such a weight based system?

I think that such a system is…

a) good, since it gives me a greater "freedom" in sorting. That is, I can choose the amounts that I wish to sort while I can dispose of the rest and pay for it.

4	3	2	1	0
Agree			Disagree	No conception

b) bad, because I wish to choose to sort myself, and not be "forced" to do so by paying a fee.

4	3	2	1	0
Agree			Disagree	No conception

c) good, for those that sort less are "punished" in the form of higher fees.

4	3	2	1	0
Agree			Disagree	No conception

d) good, since it provides a more distinct economic motive for sorting.

4	3	2	1	0
Agree			Disagree	No conception

e) bad, since it decreases my motivation for sorting more.

4	3	2	1	0
Agree			Disagree	No conception

f) it is a public duty to sort and sorting should be encouraged in other ways than through financial "carrots".

4	3	2	1	0
Agree			Disagree	No conception

17. Assume that the municipally decides to crew the recycling stations in order to grant that sorting is carried out in a "right fashion", i.e. the waste is disposed of in the correct containers. How would you feel about such a crew?

It….

a) would be useful as one receive instructions about how to dispose of the waste in an appropriate manner.

4	3	2	1	0
Agree			Disagree	No conception

b) would feel as if the authorities are monitoring my habits

4	3	2	1	0
Agree			Disagree	No conception

c) would induce me to sort even more than before.

4	3	2	1	0
Agree			Disagree	No conception

18. Do you think that you have received sufficient information regarding fares and regulation for waste handling in the municipally of Piteå?

☐ Yes, sufficient knowledge
☐ No, lack of knowledge
☐ Don't know

Comments:____

19. What I know about the Sanitary Department in Piteå I have learned through (more than one box may be checked)…

☐ Communication with the Sanitary Department
☐ Communication with the Environmental and Construction Department
☐ Internet
☐ Media
☐ Information pamphlets from the municipality
☐ Neighbors/friends/etc.

20. Have you been in contact with the Sanitary Department in Piteå regarding refuse collection/dispose handling during the past 12 months?

☐ Yes
☐ No

21. Were you satisfied/dissatisfied with this contact?

☐ Satisfied
☐ Dissatisfied
☐ Neither satisfied nor dissatisfied
☐ Don't know

B: Socioeconomic questions

22. Are you a woman or a man?

☐ Woman ☐ Man

23. How old are you?

_______years

24. What is your highest level of education?
(check only one alternative)

☐ Elementary school or equivalent
☐ Senior high school or equivalent
☐ Residential college
☐ College or university
☐ Other, namely:__________________________

25. What type of living do you have?

☐ House, row house, link house
☐ Apartment
☐ Other, namely:___________________

26. Which of the following alternatives best describes your total monthly household income before taxes? (Also include other forms of income, such as housing allowance, parent's allowance, study support, unemployment benefits etc).

☐ Less than 5 000 SEK per month
☐ Between 5 001 and 10 000 SEK per month
☐ Between 10 001 and 15 000 SEK per month
☐ Between 15 001 and 20 000 SEK per month
☐ Between 20 001 and 25 000 SEK per month
☐ Between 25 001 and 30 001 SEK per month
☐ Between 30 001 and 40 000 SEK per month
☐ Between 40 001 and 50 000 SEK per month
☐ Between 50 001 and 60 000 SEK per month
☐ Between 60 001 and 70 000 SEK per month
☐ More than 70 000 SEK per month

C: Comments

Thank you for your assistance – Please use the space below for additional comments regarding the questionnaire, recycling activities in Piteå or anything else that you think is important for this survey.