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A Comparison of Household Recycling Behaviors in Norway and the United States

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Abstract. This paper investigates empirically the determinants of household recycling in Norway and compares the results with a similar, recently published, study of households in the United States. The comparison focuses on the relative importance of user fees on waste disposal, community recycling programs, and socioeconomic factors. Both data sources are nationwide, material-specific, and at the household level. One major finding is that a disposal fee provides a significant economic incentive to Norwegian households, whereas its effectiveness in the United States is still up for debate. Providing households with convenient recycling options, such as curbside and drop-off recycling, appears generally effective, but less so in Norway than in the United States. Socioeconomic characteristics are less important predictors of behavior in Norway than in the United States. Qualifications on the comparison are provided throughout and two extensions for future research are suggested at the end.

Key words: community recycling programs, cross-country comparison, environmental policy, household waste management, user fees on waste disposal

1. Introduction

Virtually all production and consumption activities that humans engage in create byproducts, or waste materials, that are socially costly to handle. Historically, production and consumption byproducts have typically been burned, put in landfills, or sometimes dumped into oceans and other water bodies. These handling methods generate public "bads" through negative impacts on environmental quality and human health. Moreover, the so-called waste materials are derived from scarce natural resources that are costly to extract and that could have secondary-use values. Perhaps for these reasons, the last few decades have seen growing citizen and government concern over waste issues. To conserve resources, reduce reliance on landfills, and combat environmental problems associated with traditional waste handling methods, nations have turned to aggressive pursuit of recycling and other waste reduction policies.

Following a line of research into the effectiveness of these policies, this paper investigates the determinants of recycling behavior in Norway, using rich material-specific, household-level data.[1] Understanding people's responsiveness to specific policy options, and more generally, establishing which factors influence household recycling behavior, is important for evaluating the performance of current policies as well designing future policies. Community recycling programs (CRPs) are costly to implement and operate, and whether they can be justified economically depends critically on the extent to which households participate in them.

Despite the importance of this topic, the empirical literature employing highly disaggregated data to investigate policy effectiveness is sparse. A recent study of households in the United States in Jenkins et al. (2003), henceforth referred to as JMPP, constitutes arguably the most comprehensive analysis of the determinants of recycling behavior so far.[2] With similar data for Norwegian households this paper is able to provide a direct cross-country comparison. Such comparison is interesting because though standard economic theory offers unambiguous *qualitative* predictions of the effects of most policies, these predictions are not always born out in practice. Moreover, underlying idiosyncrasies in population preferences intuitively suggest that policies could give rise to significantly different *quantitative* responses in different social contexts. The comparison is focused on the role of user fees on waste disposal and provision of convenient recycling options, the two most wide-spread policies. Another point of comparison is the effects of socioeconomic and demographic factors. Finally, in addition to presenting results for several material types commonly targeted by policies in both countries, this paper presents a unique analysis for food waste, a material that has not been studied before.

The next section sets the stage with background information on waste management in the two countries. Section 3, compares the Norwegian data that we use with the U.S. data employed in the primary reference study, taking account of differences in the definition and measurement of variables, as well as sample characteristics. The conceptual framework and econometric specification are explained in Section 4, with estimation results following in Section 5. Summary of findings and suggestions for future research are given in Section 6.

2. Waste Management in Norway and the United States

Regulations of waste, and institutions for its management, have evolved similarly in the two countries. In the United States, public concern about waste related issues manifested itself in Subtitle D of the Resource Conservation and Recovery Act of 1976. In Norway, waste policies are based on the Pollution Control Act of 1981. The broad policy agenda and waste diversion goals are set at the national or state level, with implementation of specific policy options left to local jurisdictions. In Norway, the Ministry of the Environment oversees implementation of the nation's waste management goals. In the United States, this function is served by the U.S. Environmental Protection Agency (USEPA), though individual states sometimes formulate their own, more stringent, policy goals. In both countries, specific policies are implemented at the municipal level, where, typically, local utility agencies contract with private waste and recycling companies to offer waste services to households. In recent years, both countries have seen a shift from an initial concern about the number of landfills and volumes of waste, to concerns about the environmental impact of landfills, in particular, the contribution of waste to emissions of greenhouse gases.

The most common policy instruments used to combat waste problems are fees levied on traditional waste disposal and subsidization of CRPs.[3] User fees on waste disposal, so-called pay-as-you-throw schemes, give households an incentive to change their purchase patterns to reduce the waste they generate in the first place as well as an incentive to increase their recycling. CRPs, such as drop-off and curbside recycling options, operate primarily through households' time budgets, reducing the effort it takes to separate recyclable materials from other trash, thereby improving the relative attractiveness of recycling as a waste handling alternative. In the United States, user fees and CRPs started gaining popularity in the 1980s, whereas in Norway these policy tools came into prominence about a decade later. By late 1990's, about 200 out of the 435 Norwegian municipalities (covering approximately 70% of the population) had implemented some type of disposal fee scheme, while 10% of the U.S. population were subject to user fees on waste disposal (Miranda and Bynum 1999; Statistics Norway 2001).[4] When it comes to the most popular CRP, curbside recycling, 75% of the Norwegian population and roughly 50% of the U.S. population had access to this type of program (Glenn 1998; Statistics Norway 2001).[5]

Believed in large part due to pursuit of these policy options (in both the residential (household) sector as well as the commercial (business) sector of the economy), the two countries have achieved significant improvements in material recovery.[6] The overall U.S. material recovery rate went from less than 10% in 1980 to about 30% in 1999 (EPA 1999). Norway experienced a similar increase in material recovery in the 1990's, with the recovery rate reaching 57% in 1998, up from 46% in 1993 (Statistics Norway 2001).[7]

Despite these trends however, both countries continue to produce vast amounts of waste and sending large amounts of recyclable materials to landfills, with daily waste generation per capita at about 2.5–3 pounds in the United States and approximately 2 pounds in Norway (USEPA 1999, 2001; Statistics Norway 2001).[8] Moreover, recent data suggest that waste generation is rising (Statistics Norway 2001; Kaufman et al. 2004). For these reasons, policy officials in both countries continue to focus on waste as a major environmental issue, with new policies being devised at both the regional and national levels.[9] Hence, research on the effectiveness of waste management policies remains important to policy makers and planners who face the task of allocating limited budgets among a wide range of public programs and services.

3. Descriptive Data Discussion

The Norwegian data that we employ come from Statistics Norway's fourth quarter 1999 economic survey, which included a series of questions related to household recycling behavior, in addition to standard questions on socio-economic and demographic status (Statistics Norway 1999). In total, 1162 individuals completed survey interviews, with about 3/4 of the interviews conducted in person and 1/4 conducted over the telephone, resulting in a participation rate of 58%.[10] This data source is linked with municipality-level information on waste management policies, also compiled by Statistics Norway (on bi-yearly basis). In comparison, the survey data used in JMPP came from a mail-mode survey administered primarily to households in metropolitan areas, resulting in a dataset weighted more towards urban households than the Norwegian dataset. The U.S. survey focused specifically on recycling behavior, which may have lead to an over-sampling of so-called avid recyclers.[11] Table I compares definitions and measurement of variables used in our econometric analysis with those of the analysis in JMPP, Table II compares reported recycling behaviors across the two survey samples, and Table III gives descriptive statistics for exogenous variables used in the analysis of recycling in Norway (with similar descriptive statistics given in Table IV of JMPP for the U.S. data).

Recycling behavior is represented by recycling intensities or fractions of various material types that are recycled.[12] The Norwegian survey asked the participants to indicate whether they recycle *none*, *some*, *most*, or *all* of the recyclables in five material types: paper, glass, metals, plastics, and food waste, while the U.S. survey asked the participants to indicate household recycling percentages for newspaper, glass bottles, aluminum, plastic bottles, and yard waste.[13] For their analysis, JMPP fold seven recycling rate intervals into three new categories: 0–10%, 11–95%, and 96–100%, whereas we combine the *some* and *most* responses into one category (*some/most*), which gives us a comparable classification of recycling efforts.[14] Both surveys asked the participants to exclude deposit-refund materials from their recycling intensity assessments. The first four material types are similar, permitting us to make direct cross-country comparisons (with the caveat that the Norwegian material categories are more inclusive than the U.S. types; for example, paper

Table I. Comparison of variables

Variables	Norwegian data	U.S. data
Dependent variable		
Fraction of materials recycled	Qualitative indicators (none, some, most, all):	Percentage intervals (0–10%, 11–95%, >95%)
	Paper	Newspaper
	Glass	Glass bottles
	Metals	Aluminum
	Plastic	Plastic bottles
	Food waste	Yard waste
Policy variables		
Waste disposal fee	(0,1) Indicators for mandatory and voluntary fee differentiation	$/Gallon for 2nd container
Curbside recycling program	(0,1) Indicator	(0,1) Indicator
Drop-off recycling program	(0,1) Indicator	(0,1) Indicator
Curbside and mandatory	(0,1) Indicator	(0,1) Indicator
Age of recycling program	Not available	(0,1) Indicators for 1–2 years, and >2 years
Number of curbside materials	Count of materials	Count of materials
Socioeconomic and demographic variables		
Household income	Continuous	(0,1) Indicators for 10-$14.99K, 15-$24.99K, 25-$34.99K, 35-$49.99K, 40-$74.99K, >$75K
Household size	Count of members	Count of members
Age of household head	Survey participant's age	Average male and female
Single or detached house	(0,1) Indicator	(0,1) Indicator
Home owner	(0,1) Indicator	(0,1) Indicator
Education level	(0,1) Indicator for college or above	(0,1) Indicators for high school, college, and graduate degree
Population density	(0,1) Indicator for more than 100,000 people	Number of people per square mile
Other controls		
Fixed effects	(0,1) Indicators for state	(0,1) Indicators for metropolitan statistical area
Multiplicative heteroscedasticity specification	Curbside recycling and number of curbside materials	Curbside recycling and age of program

Table II. Distribution of recycling behavior

Material type	Recycling intensity category		
Norwegian sample	None	Some/most	All
Paper	10.87%	20.07%	69.06%
Glass	18.52%	21.27%	60.21%
Metals	47.32%	19.47%	33.21%
Plastics	66.07%	13.86%	20.07%
Food waste	53.05%	7.88%	39.07%
U.S. Sample in reference study:[a]	0–10%	11–95%	96–100%
Newspaper	8.8%	16.6%	74.6%
Glass bottles	11.3%	22.2%	66.5%
Aluminum	15.0%	21.8%	63.2%
Plastic bottles	17.8%	28.0%	54.2%
Yard waste	43.3%	22.8%	33.9%

[a]From Table 5 of JMPP.

Table III. Descriptive statistics for the Norwegian sample

Variable	Sample mean	St. dev.
Policy variables		
"Voluntary" waste disposal fee system	0.56	0.50
"Mandatory" waste disposal fee system	0.18	0.38
Curbside collection of paper	0.89	0.32
Curbside collection of glass	0.03	0.17
Curbside collection of metals	0.02	0.13
Curbside collection of plastics	0.02	0.13
Curbside collection of food waste	0.60	0.49
Number of curbside materials	1.55	0.70
Drop-off program for paper	0.45	0.50
Drop-off program for glass	0.99	0.10
Drop-off program for metals	0.68	0.47
Drop-off program for plastics	0.26	0.44
Drop-off program for food waste	0.09	0.28
Perceived mandatory recycling	0.35	0.48
Socioeconomic and demographic variables		
Household income (in 1000 NOK)	390.97	263.40
Household size	2.78	1.43
Age of head of household	43.06	16.60
Single or detached house	0.58	0.49
Home ownership	0.76	0.43
College degree or above	0.24	0.43
Population $>100{,}000$	0.27	0.44

encompasses newspapers as well as other recyclable paper items). The last material category, yard waste in the U.S. data and food waste in the Norwegian data, are both wet-organic materials, but cannot be directly compared as people are likely to view recycling these materials as two substantially different activities. As seen in Table II, paper (newspaper) and glass (glass bottles) are the materials subject to most recycling in both the Norwegian sample and the U.S. sample, whereas recycling efforts for metals and plastics (aluminum and plastic bottles) are significantly lower in Norway than in the United States. Surprisingly, the profiles for the wet-organic materials look similar, with the exception that food waste recycling appears to be more an "all or nothing" decision in Norway than is the yard waste recycling decision in the United States.

The primary policy information incorporated into the econometric estimations is presence of a user fee on waste disposal, presence of curbside recycling, and presence of a drop-off recycling option. We distinguish between two types of user fee schemes, mandatory versus voluntary (see note 5 for an explanation of this distinction). The curbside and drop-off recycling variables are material-specific and can be regarded as binary attributes of the CRP. As observed in Table III, close to 3/4 of the Norwegian households lived in communities with waste disposal fees (56% voluntary and 18% mandatory). In the U.S. sample, 10% lived in communities with disposal fees and the mean marginal price faced was about $0.06 per gallon per month, according to JMPP. With respect to availability of curbside recycling, 89% of Norwegian households had collection of paper, less than 5% had access to curbside recycling of glass, metals, and plastics, respectively, while curbside collection of food waste was available to 60% of them. In comparison, JMPP report that 92% of the U.S. sample had access to newspaper curbside recycling, over 75% lived in municipal areas with curbside collection of glass bottles, aluminum, and plastic bottles, respectively, and 53% had access to curbside collection of yard waste. The most common drop-off material in Norway is glass, with 99% of the sample living in communities with this option. In contrast, less than 1% of the U.S. households had drop-off glass recycling. In general, drop-off options appear much more common in Norway. However, a comparison based on these two surveys alone is somewhat skewed, as drop-off recycling is more common in rural areas and the U.S. survey was weighted towards urban areas.

Other CRP attributes may also be of importance to households' recycling decisions. We construct a variable representing the number of materials covered by the curbside recycling program, as a way to capture economies of scale and familiarity with this program type. JMPP also include information on the age of the CRP to capture familiarity effect. This information is not available on Norwegian CRPs. However, it is unlikely that this omission will invalidate the comparison as CRPs were introduced only recently in Norway, with subsequent little variation in length of existence across communities. The average number of materials collected in the curbside recycling program was 1.55 in the Norwegian data and 3.9 in the U.S. data (as reported by JMPP). Another CRP attribute, specific to curbside programs, is whether participation is voluntary or mandatory. For this attribute, both our analysis and that of JMPP rely on self-reported information. About 35% of the Norwegian survey participants either partially agreed or agreed with the statement "I recycle because I consider it a mandate from the government", despite the fact that, to the best of our knowledge, recycling is not mandatory in any of the Norwegian municipalities. In the U.S. sample, over 1/3 stated that recycling is mandatory. Other CRP attributes that neither our study nor JMPP had data on was frequency of curbside collection, the extent to which different recyclables had to be sorted into separate material streams, travel distances to drop-off centers, and the extent to which guidance was provided to households on how to handle various recyclable items. The unavailability of data on these attributes is likely to make parameter estimates noisier, particularly on variables capturing presence of curbside and drop-off recycling options.

Socioeconomic and demographic effects are captured with data on household income, household size, education, age, home-type and homeownership, and population density, with the difference in how we incorporate this information versus how it was incorporated in JMPP given in Table I. As seen in Table III, the profile for the Norwegian sample is as follows: the average household income was about NOK 391,000 (roughly $55,000) per year, the average household size was 2.7, 24% of the respondents had more than a high school degree, the average age was 43 years, 58% lived in single or detached homes, 76% were homeowners, and 27% lived in communities with more than 100,000 people. According to JMPP, the most common income category among the U.S. survey participants was $50,000 to $75,000, with over half the sample reporting income over $35,000. The average household size was 2.8, over half the respondents had education beyond high school, the average age of the household head(s) was 48 years, 76% lived in detached homes, 79% were homeowners, and the average population density of the respondents' communities was 5,800 persons per square mile.

4. Conceptual Framework and Econometric Model

The intensity of recycling, or the fraction of each material recycled, is assumed to be a function of waste disposal fees, CRP attributes, and socioeconomic and demographic factors. The recycling data with categorical responses lead naturally to an ordered model specification in which the true or actual fraction is latent.[15] Instead of observing actual recycling intensities, we observe whether households recycle *none*, *some/most*, or *all* of each material. Following JMPP, let y^* represent the true recycling intensity, assumed to arise from utility maximization, and let X represent a matrix of exogenous variables with a conformable vector of parameters β. Furthermore, let subscript i track households and subscript j track material types. A reduced form behavioral relationship can thus be expressed as

$$y_{ij}^* = X_{ij}\beta_j + \varepsilon_{ij}, \tag{1}$$

where ε_{ij} is a stochastic error term representing unmeasured factors, assumed to be distributed logistically. Define $y \in \{0, 1, 2\}$, where 0 represents the recycle *none* response category, 1 represents *some* or *most*, and 2 stands for the recycle *most* response category, then the probabilities of observing behaviors in the three response categories are

$$\begin{aligned}
\Pr(y_{ij} = 0) &= \frac{1}{1 + e^{X_{ij}\beta_j}}, \\
\Pr(y_{ij} = 1) &= \frac{1}{1 + e^{-\mu_j + X_{ij}\beta_j}} - \frac{1}{1 + e^{X_{ij}\beta_j}}, \\
\Pr(y_{ij} = 2) &= 1 - \frac{1}{1 + e^{-\mu_j + X_{ij}\beta_j}},
\end{aligned} \tag{2}$$

where μ_j is a material-specific ordered logit threshold parameter to be estimated along with β_j. In our case, the relationship between the latent recycling intensity, the response categories, and the threshold parameter is as follows: $y = 0$ if $y^* = 0$, $y = 1$ if $0 < y^* < \mu$, and $y = 2$ if $y^* \geq \mu$.

5. Estimation Results and Discussion

The ordered response model was estimated by maximum likelihood in LIMDEP version 7.0 and the main results are reported in Table IV. The sign of estimated parameters gives the qualitative effects of the variables on recycling intensities. The high chi-square statistics indicate that the estimated material equations are overall highly significant.

Multiplicative heteroscedasticity in presence of curbside recycling and number of curbside materials was tested and subsequently corrected for where appropriate, the underlying idea being that there may be less variation in recycling intensities when recycling is more convenient and when households are more familiar with the activity. However, likelihood ratio tests fail to reject constant error variance in all cases, except in the case of food waste. In comparison, JMPP tested for heteroscedasticity in presence and age of the

Table IV. Ordered logit results for Norwegian household recycling behavior

Variable name	Paper	Glass	Metals	Plastics	Food waste
Constant	0.1568 (0.3132)	0.7253** (0.3306)	−0.5549* (0.2887)	−0.7696*** (0.2972)	−0.7268*** (0.2459)
Policy variables:					
Disposal fee 1 (voluntary)	0.4066*** (0.1351)	−0.1084 (0.1402)	0.1215 (0.1301)	0.3880*** (0.1287)	0.2564*** (0.0984)
Disposal fee 2 (mandatory)	0.2656 (0.2257)	0.0491 (0.2324)	0.4399** (0.2124)	0.155 (0.2215)	0.6423*** (0.1973)
Drop-off recycling program	0.0720 (0.1364)	0.2646* (0.157)	0.0411 (0.1289)	0.1027 (0.1844)	−0.1652 (0.3779)
Curbside recycling program	0.4014* (0.2205)	−0.0789 (0.6738)	−0.0760 (0.9154)	1.5036* (0.8934)	0.8636*** (0.1421)
Number of curbside materials	0.2246* (0.1217)	0.0048 (0.1103)	0.2191** (0.0934)	0.3023*** (0.0916)	0.0853 (0.0999)
Mandatory recycling and curbside	−0.1479 (0.1126)	0.0075 (0.5034)	0.7604 (1.0127)	0.2312 (1.0705)	0.1378* (0.0804)
Socioeconomic and demographic variables:					
Household income[a]	0.0111 (0.0183)	0.0311 (0.0200)	0.0131 (0.0209)	0.0012 (0.0198)	0.0054 (0.0136)
Age of head of household	0.0140*** (0.0032)	0.0104*** (0.0028)	0.0103*** (0.0028)	0.0046 (0.0030)	0.0031 (0.0021)
Household size	−0.0076 (0.0386)	0.0000 (0.0368)	0.0046 (0.0352)	−0.0357 (0.0367)	−0.0018 (0.0231)
Single or detached house	−0.0132 (0.1382)	−0.1135 (0.1357)	0.0446 (0.1208)	0.0218 (0.1317)	0.1252 (0.0815)
Home ownership	−0.1199 (0.1591)	0.0102 (0.1543)	0.0308 (0.1393)	0.0230 (0.1534)	0.0052 (0.0974)
College degree or above	0.0677 (0.1274)	−0.0728 (0.1034)	0.0109 (0.0986)	−0.1764 (0.1134)	−0.0079 (0.0718)
Population $>$ 100,000	−0.0679 (0.2015)	−0.1156 (0.1607)	−0.2824* (0.1662)	−0.4025** (0.1839)	−0.6052*** (0.1573)
Ordered logit threshold parameter	1.0437*** (0.0832)	0.7762*** (0.0531)	0.6209*** (0.0423)	0.5198*** (0.0424)	0.2193*** (0.0436)
County controls	Yes	Yes	Yes	Yes	Yes
Heteroscedasticity corrected	No	No	No	No	Yes
Model Performance:					
Unrestricted log-likelihood	−575.3685	−725.4448	−818.5878	−695.3053	−566.2492
Restricted log-likelihood	−618.9042	−763.3578	−903.0128	−764.2360	−791.0328
Chi square statistic	87.0716***	75.8260***	168.8500***	137.8613***	449.5671***

Note: N = 853 across materials. Standard errors are reported in parentheses. *,**,***Indicate significance at the 90%, 95%, 99% level of confidence, respectively. [a]In NOK 100,000.

curbside recycling program, which was detected and corrected for in two out of five materials, namely, glass bottles and plastic bottles. In addition to the variables discussed in Section 3, we include regional indicator variables to control for policy influences operating beyond the community level, as each Norwegian municipality belongs to one of 20 different counties. These variables are jointly significant in the estimations, with qualitative findings for the variables of primary interest, that is, the policy variables, robust with respect to inclusion of these regional variables. JMPP find metropolitan statistical area indicators, variables serving a similar purpose, to be jointly significant in their estimations.

5.1. SIGNIFICANCE OF USER FEES ON WASTE DISPOSAL AND CRP ATTRIBUTES

The results in Table IV suggest that user fees on waste disposal have a positive effect on recycling intensities, except in the case of glass, with the voluntary fee indicator entering significantly in the estimations for paper, plastics, and food waste and the mandatory fee indicator being significant in the metals and food waste equations. This finding differs from that of JMPP, who find no effect of disposal fee on recycling intensities.[16] The broader literature is mixed on this point, as there are several studies which find an effect of disposal fees on recycling effort (e.g., Hong et al. 1993; Callan and Thomas 1997; Hong 1999; Ferrara and Missios 2005), and several which do not (e.g., Reschovsky and Stone 1994; Fullerton and Kinnaman 1996; Kinnaman and Fullerton 2000). Within both sets of studies there are some that control for presence of disposal fee and some that use information on the marginal disposal price, which means that the mixed findings cannot be attributed to this difference.[17] For example, the household-level, material-specific study by Ferrara and Missios (2005) finds that the marginal price of waste disposal significantly increases recycling intensity.[18] Interestingly, Ferrara and Missios (2005) investigate recycling behavior in Canada, while Hong (1999) studies recycling by Korean households. Hence, perhaps an emerging insight is that user fees work in several societies, including in Norway, whereas their effectiveness in the United States is yet to be fully established.

When it comes to CRP effectiveness, our estimation results indicate that presence of a curbside collection program increases paper, plastics, and food waste recycling intensities. But, we fail to establish that this program type is effective in the cases of glass and metals. For drop-off recycling availability, we note that all parameter estimates, except that for food waste, are positive, as expected. Given relatively large standard errors, however, the effect of drop-off recycling is statistically significant only in the glass recycling estimation. Overall, we find fewer significant effects of presence of curbside and drop-off recycling options than do JMPP, who conclude that both types of program have positive effects on recycling intensities across all materials.

On the surface, this comparison suggests that convenience, or reducing the effort it takes to recycle, is less important to Norwegian households than to U.S. households. However, drawing such a conclusion requires several qualifications. With regards to curbside recycling, very few households in the Norwegian sample had collection of glass, metals, and plastics, making the estimations sensitive to both measurement error and unobservable factors operating on the behaviors of these people. Moreover, exclusion of deposit-refund materials and the fact that aluminum beverage containers are less common in Norway (relative to glass and plastic bottles) mean that the glass and metal material categories are comprised of small amounts of items, which are relative time-intensive to recycle. Subsequently, if convenience matters, it is to be expected that a curbside recycling option for these materials is relatively ineffective. Finally, when it comes to drop-off recycling, the fact that the Norwegian sample was more rural could matter. By not controlling for other attributes of this program type, such as transportation distances to drop-off locations, the parameters estimates for availability are likely to be noisy.[19]

Among other CRP attributes, we find that increasing the number of materials collected in curbside recycling programs has a positive effect on paper, metals, and plastics recycling intensities.[20] This finding is important for policy as the average household had curbside collection of less than two materials. In comparison, JMPP find this variable to be significant only in the case of newspaper recycling. At the same time though, their control variable for how long the program had been in existence was significant in their newspaper, plastic bottles, and yard waste models. Combined, these findings lend support to the hypothesis that experience with CRPs, which in turn may be related to household recycling efficiency, affects behavior. With regards to whether recycling is voluntary or mandatory, both our analysis and JMPP find that making recycling mandatory is generally ineffective, though our results suggests that it may have an impact on food waste recycling intensities. This differ from the finding by Ferrara and Missios (2005) that Canadian households increase recycling effort for most materials when participation in mandatory.

5.2. SIGNIFICANCE OF SOCIOECONOMIC AND DEMOGRAPHIC FACTORS

We find relatively few socioeconomic and demographic influences on recycling intensities. Age has a significant positive effect in the paper, glass, and metals estimations and the population density variable enters significantly with negative parameter in the metals, plastics, and food waste estimations.

In comparison, JMPP find the age variable to be significant and positive in explaining recycling of newspaper, aluminum, plastic bottles, and yard waste, whereas their population density variable is a significantly negative only in the yard waste estimation. These authors also find several other variables, such as income and education, to be significant in the regressions for various materials.[21]

Table V. Marginal effects of significant policy variables

Variable name	Paper	Glass	Metals	Plastics	Food waste
Disposal fee 1 (voluntary)					
Change in Probability of Recycling...					
None	−0.0382			−0.1419	−0.1499
Some or most	−0.0925			+0.0421	+0.0197
All	+0.1307			+0.0998	+0.1302
Disposal fee 2 (mandatory)					
Change in probability of recycling...					
None			−0.1721		−0.3754
Some or most			+0.0116		+0.0494
All			+0.1605		+0.3260
Drop-off recycling program					
Chance in probability of recycling...					
None		−0.0538			
Some or most		−0.0442			
All		+0.0980			
Curbside recycling program					
Change in probability of recycling...					
None	−0.0377			−0.5500	−0.5048
Some or most	−0.0913			+0.1631	+0.0665
All	+0.1290			+0.3869	+0.4383
Number of curbside materials					
Change in probability of recycling...					
None	−0.0211		−0.0857	−0.1106	
Some or most	−0.0511		+0.0058	+0.0328	
All	+0.0722		+0.0799	+0.0778	
Perceived mandatory recycling and curbside					
Change in probability of recycling...					
None					−0.0805
Some or most					+0.0106
All					+0.0699

Note: For discrete variables, the estimated model in Table IV is used to calculate the response category (*none*, *some/most*, and *all*) probabilities for the variable of interest first set at 1 and then at 0, with all other variables held at their mean values. The differences are the marginal effects. For the count variable, number of curbside materials, the marginal effects are calculated for a 1 unit change, with all other variables held at their means.

5.3. QUANTITATIVE EFFECTS OF SIGNIFICANT POLICY VARIABLES

Due to the non-linear nature of the ordered logit model, the estimated parameters discussed above can only be interpreted qualitatively. In order to assess the potential magnitude of policy effects, so-called marginal effects are calculated and given in Table V.[22]

Using paper as an example, instituting a ("voluntary") disposal fee scheme, decreases the probability that none of this material is recycled by 3.82% and increases the probability of recycling all paper by 13.07%. Similarly, presence of a curbside recycling option for paper is predicted to change these probabilities by −3.77% and +12.9%, respectively. Finally, adding one additional material to the curbside recycling program decreases the probability of zero paper recycling by 2.11% and increases the probability of full paper recycling by 7.22%.

Looking at the marginal effects across materials, a couple of potentially important insights emerge. First, since the marginal effects can be interpreted as changes in "market shares" for the three recycling intensity categories, the predicted quantitative effects of policies are generally non-trivial. Making a curbside recycling option available and instituting user fees on waste disposal fees appear particularly effective, whereas drop-off recycling, increasing the number of materials collected in the curbside program, and making participation mandatory have more modest effects. Second, the effectiveness of various policy options varies greatly from material to material (with households perhaps particularly responsive to food waste recycling policies). These findings echo results in JMPP as well as Ferrara and Missios (2005).[23]

5.4. POLICY OPTIONS FOR FOOD WASTE DIVERSION

Our analysis provides a unique look at the effects of food waste policies. This material has received little attention in the literature, despite the fact that it constitutes a significant portion of the total waste stream, 28% of all household waste in Norway and 11% of all municipal solid waste in the United States, with most of it ending up in landfills (Statistics Norway 2001; USEPA 2001). Many communities in both countries encourage and facilitate home composting of food waste, an activity that requires significant effort on the part of households, with mixed results; see Sterner and Bartelling (1999) for an analysis of the determinants of composting.

An alternative to promoting home composting is to provide households with curbside collection of separated food waste, with the material subsequently taken to a central composting facility (potentially along with yard waste). While this option is now widespread in Norway (as indicated by Table III), it is relatively uncommon in the United States, but increasingly being considered by policy planners in U.S. municipalities. Our estimation results suggest that both availability of curbside recycling and user fees on waste disposal have positive effect on food waste recycling, though future research is needed to establish the effectiveness of these policies elsewhere.

5.5. ECONOMETRIC ROBUSTNESS ISSUES

This paper has sought to make a cross-country comparison, using a particular U.S. study as the primary point of reference. For this reason, we have presented results for an econometric specification that tracks the reference study as closely as possible. Here, we would like to briefly summarize econometric specification issues that were addressed either explicitly in exploratory work or implicitly in other analyses of the same data (Halvorsen and Kipperberg 2003; Kipperberg 2005).

First, with regards to the error term in equation (1), we assumed iid logistic distribution, leading to the ordered logit model. An alternative would be to assume normality, which leads to the ordered probit model. These models typically yield identical results, up to a scale factor, which was true in our case as well.[24]

Second, with regards to potential nonlinearities in the deterministic part of equation (1), we tried several extensions of the specification reported here. A common prediction of simple theoretical models of household waste management is that providing a recycling option cannot induce recycling behavior as long as disposal is free and recycling is costly in terms of requiring additional effort. This suggests that availability of curbside recycling will be more (or only) effective in the presence of user fees. This hypothesis was tested for and rejected with curbside availability/disposal fee presence interaction terms. Nonlinearities in income and age were also explored (with logarithmic transformations or adding squared terms), without yielding further insights.

Third, robustness issues, related to selection of covariates, are omitted variable bias and the extent to which certain included variables, or specific observations, unduly influence the results. With regards to the first issue, additional explanatory variables were explored in Halvorsen and Kipperberg (2003), without qualitatively changing the results reported here.[25] Also, as already noted, unavailability of data on some CRP attributes, may have resulted in parameter estimates with higher standard errors than would otherwise have been the case. With regards to the issue of influential variables (potentially due to outliers), most of the variables are discrete, making this less of an issue. Trying various specifications with different sets of socioeconomic and demographic variables did not qualitatively alter the policy findings. However, preliminary data explorations uncovered sensitivity to observations from one specific municipality, likely due to measurement error in its policy data, which was dealt with by removing these households from final analysis.

Finally, ordered response models are subject to the parallel regression assumption, in our case, implicitly imposing the restriction that $\partial \Pr(y \leq 1)/\partial X = \partial \Pr(y \leq 2)/\partial X$. This assumption is frequently violated, suggesting that practitioners might want to consider other econometric specifications, as noted by Long (1997, pp. 145–147). Informal tests for some of our material equations were indeed indicative of such violation. However, as qualitative results (in terms of the direction of the effects and significance of the policy variables) were upheld in these tests, we opt to proceed with the ordered response model specification, which makes conceptually sense given the type of data we have.[26]

6. Concluding Remarks

This paper has (1) added to limited disaggregated empirical research on recycling behavior by investigating the determinants of material-specific recycling for households in Norway, (2) provided a cross-country comparison of behavior using a specific U.S. study, Jenkins et al. (2003), as primary point of reference, and (3) offered a unique look at food waste recycling. Overall, we corroborate the finding from other studies that waste diversion policies influence recycling behavior, with the effectiveness of different policies varying across materials, both qualitatively and quantitatively. We also provide evidence that policy influences vary across social environments.

In terms of Norwegian households, user fees on waste disposal may have a double pay-off. User fees constitute a source of financing for the overall waste management system and increase recycling participation (which, in turn, reduces the marginal operating cost of CRPs, making them more cost-effective). An additional effect, not studied here, is that user fees can reduce the amount of waste generated and disposed of, thereby correcting for over-production caused by zero or flat disposal fees. While disposal fees are found to positively influence recycling intensities in Norway, their effectiveness as a recycling incentive in the United States is yet to be fully established. Different CRP types and attributes are found to have different impacts on recycling behavior, depending on the material type in question. Policy makers who consider CRP options should also evaluate how important specific materials are in the overall waste stream, which other policies are already in place or could be pursued (e.g. deposit-refund systems), and most importantly, the relative costs of these options across materials. In the case of Norway, we find that availability of curbside recycling has a positive effect on paper, plastics, and food waste recycling intensities and that availability of drop-recycling affects glass recycling, with some evidence that these options may influence recycling of other materials as well. In comparison, JMPP found that both curbside and drop-off options have positive impacts across materials in the United States, with curbside programs having stronger quantitative effects. For Norway, we have noted that both glass and plastics constitute small material-categories, when deposit-refund items are excluded from consideration, raising question about whether it makes sense to have curbside recycling options for these materials. Policies targeted towards food waste recycling are found to be particularly effective, a result that has important policy relevance, given the compositional importance of this material in the overall waste stream.

In closing, let us suggest two additional aspects of recycling that deserve more research attention. First, it is a fact that many households in both Norway and the United States recycle without direct economic incentives to do so, suggesting that other motivations (perhaps a desire to contribute to public goods or conform to social norms) may be at play (Hornik et al. 1995; Kipperberg 2005). Comparing the relative importance of such factors across societies, and the extent to which they are complements or substitutes for policies, both in the context of household recycling as well as for other types of *pro-environmental behaviors*, is an interesting direction for future work. Second, many empirical studies of household recycling behavior are conceptually based on the standard single (money)-constraint model of utility maximization, failing to take proper account of the fact that recycling is an activity that requires non-trivial time efforts. Indeed, in the absence of money incentives to reduce waste disposal, time costs might be a dominant economic consideration, suggesting that empirical research should explicitly incorporate time prices, associated opportunity costs of time, and household time constraints (Hong et al. 1993; Jakus et al. 1996; Halvorsen and Kipperberg 2003). While beyond the specific scope of this paper, we believe that including non-pecuniary motivations and the time aspect would improve conceptual realism, statistical predictive power, and policy relevance of future empirical work on household recycling behavior.

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Notes

1. See Kinnaman and Fullerton (1999) for a good overview of policy issues and the existing literature.

2. Ferrara and Missios (2005) present another recent micro-level, material-specific study, carried out in Ontario, Canada. Whenever possible, we also compare results with this study as well as other relevant empirical work.
3. Deposit-refund systems, another policy instrument widely in use in Norway but less common in the United States, place a tax on certain products with recyclable material content at the time of purchase, which is given back if the recyclable component is returned. Recycling under deposit refund systems is not coverer in this paper.
4. *Variable can subscription*, the most prevalent U.S. user fee scheme, permits households to choose between different sizes of trash can and pay corresponding monthly or bi-monthly fees for collection. A second type of fee scheme is what JMPP refer to as *bag/tag/sticker programs*, wherein households pay a per-bag fee. For Norway, Statistics Norway distinguishes between ‘‘mandatory’’ and ‘‘voluntary’’ fee schemes. The former classification encompasses both variable can subscription and bag/tag/sticker programs, but may also include other forms of waste fee gradation. The magnitude of the fees and the choices that households make are not tracked. The latter classification may include systems that permit neighbors to share waste collection service and various forms of mixed payment schemes (e.g., payment of a fixed fee up to an allowable amount of waste disposal, with an incremental charge thereafter).
5. Curbside recycling programs in Norway typically offer collection of only one or two materials, whereas in the United States more materials are covered. Drop-off recycling, anothe common type of CRP, provides households with a central location to which various recyclable materials can be taken. This program type requires greater household effort and is therefore expected to be less effective than curbside recycling. Drop-off recycling is more common in sparsely populated areas where operating curbside recycling programs is seen as too costly.
6. Material recovery, sometimes referred to as waste diversion and typically reported in percentage terms, is a measure of how much of the total waste generated in the economy is diverted away from landfills and other traditional waste handling methods and thereby made available for secondary uses.
7. These country-level material recovery rates cannot be compared directly since incineration with energy recovery, more common in Norway, is included in the Norwegian rate but not the U.S. rate. Norway is also likely to have had higher baseline (pre-policy implementation) recycling levels, due to more extensive usage of deposit-refund systems.
8. The USEPA reports total municipal solid waste of 4.6 pounds per capita per day, of which an estimated 55–65% is post-consumer waste originating in the residential household sector.
9. There is also growing concern about the impact of ‘‘new’’ waste materials, such as obsolete consumer electronics, an important topic for further research that is beyond the scope of this paper.
10. See Bruvoll et al. (2002) for a more detailed discussion of the survey data.
11. Respondents with item non-response for key variables or without curbside recycling for at least one material were dropped following JMPP, resulting in a usable sample of 853 households, slightly smaller than in the reference study.
12. By using stated (self-reported) behavior data, neither of the studies can address the extent to which self-reported behavior differs from actual behavior, and whether this leads to biased results.
13. We assume that the participants’ reported recycling intensities are representative of their household-level recycling.
14. As an approximation, it is reasonable to think of recycling 0–10% as close to recycling *none*, 11–95% as similar to recycling *some* to *most*, and 96–100% as about the same as

recycling *all* of any given material type. This folding makes sense due to the clustering of responses in the lowest and highest response categories. Moreover, the econometric estimations presented later are qualitatively robust with respect to this folding.

15. Ordered response models are standard in the literature and are described in various econometric textbooks (e.g., Maddala 1983; Greene 2002; Train 2003; Long 1997). Hence, we only give a brief outline of the econometric model here.
16. JMPP give three explanations for this result: (1) small, potentially trivial, marginal prices for the households subject to waste disposal fees; (2) the discontinuous nature of most fee schemes; (3) disposal fees only constitute an indirect incentives to recycle, whereas its primary incentive is to reduce the amount of waste produced in the first place.
17. These indicator variables may also capture other unobserved factors if communities that implement disposal fee schemes are also vigorous in pursuing recycling participation through other means. We believe this potential *proxy error* is minimized by inclusion of variables that describe the CRPs and regional control variables.
18. A strong point of this study is that it controlled for specific features of the disposal fee scheme, namely, whether it permitted disposal of some amounts for free and whether there was upper limit on how much could be thrown away.
19. Halvorsen and Kipperberg (2003) constitute an important robustness check for results presented here and reinforce this caveat. When the monetized time cost of recycling (with both material-specific time prices and household-specific values of time derived from first-stage estimations) is included, along with various attitudinal measures, policy parameters are estimated more efficiently: the curbside glass recycling indicator is significant, as are the drop-off indicators for glass, plastics, and metals.
20. Additional curbside materials may increase household familiarity with the program or be associated with economies of scale, thought the opposite effect, namely that more materials lower participation propensity by introducing uncertainty about which items are permitted, could be argued as well. Our estimation results suggest that the overall effect is positive.
21. One explanation for why we find fewer socioeconomic and demographic effects could be the fact that the Norwegian population is much more homogeneous in these aspects than the U.S. population. A different exploration of the same survey data in Halvorsen and Kipperberg (2003) finds that recycling intensity is related to other personal/household characteristics, to wit, attitudes towards public good contributions and the environment.
22. Other approaches to obtaining quantitative measures, such as by computing (fully) standardized parameters, are discussed in detail in Long (1997).
23. These general observations are robust with respect to model specification (see discussion below), though point estimates for the quantitative effects vary some from specification to specification.
24. According to Long (1997, p. 120) the choice between these two models "is largely one of convenience". He also notes that other distributions "have been considered, but are not often used", p. 119.
25. We do not report these results in this paper because these aspects of behavior were not included in JMPP.
26. As we consider multiple equations, it is unclear what fix would be appropriate if we formally found violations in some, but not all cases. Moreover, using different econometric models would make the comparison with JMPP (and other relevant studies using ordered response models, such as Ferrara and Missios (2005), less straightforward and potentially invalid. We have also already noted that the results are robust with respect to the folding of four recycling intensity categories into three. Finally, an analysis in

Kipperberg (2005), which simply conceptualizes the recycling decision as a binary choice, finds CRPs and disposal fees to be generally effective in influencing behavior.

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