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The economics of household waste management: a review†

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In recent years reducing the amount of waste generated by households has become an important policy issue in industrialised economies. It is no longer acceptable to discard waste without concern for environmental and natural resource issues. In an effort to reduce household waste various policy instruments such as kerbside charges, deposit-refund schemes, integrated sales tax exemptions and virgin material taxes, have been proposed and/or implemented. This article reviews the economics literature that has addressed household waste management. It is argued that a comprehensive modelling framework is necessary if the complex policy environment is to be accurately described.

1. Introduction

In recent years how society should deal with household waste has become a significant policy issue in industrialised economies. It is no longer acceptable to discard waste materials without concern for environmental and natural resource issues. The concern relates to resource scarcity for the production of consumption goods and the need to recover (recycle or reuse) waste materials, as well as the external effects, such as water pollution and site contamination at landfills, and littering resulting from the illegal disposal of the waste. The desire for improved household waste management is part of the greater emphasis placed upon environmental issues openly articulated at the 1992 Earth Summit. Take for example Agenda 21, which expresses the requirement for sustainable economic activity and the need for mankind to remain in harmony with the carrying capacity of the earth.

Like many nations, Australia has developed policies and initiatives to deal with household waste. In 1992 the Australian Commonwealth government declared a target of halving domestic waste going to landfill by the year 2000. Accompanying this target were related recycling targets for materials such as paper, glass and aluminium. These initiatives have the support of

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most Australians as the willingness of households to address the waste issue is strong. In a recent survey of Melbourne households (RRRC 1994) 76 per cent thought that it was very important for communities to cut down on the waste generated. Furthermore, 91 per cent of those interviewed undertook varying degrees of kerbside recycling.

Similar policy initiatives have been introduced in other parts of the world. For example, in the 1990 Environmental Protection Act, the UK government committed itself to reducing waste and promoting the amount of recycling. A target of 25 per cent of household waste to be recycled was set. In Germany the Packing Ordinance was passed in 1991. The law was introduced in recognition of the fact that about half of all municipal solid waste on a volume basis was packaging. The Ordinance is a cradle-to-grave approach to waste management with the emphasis for action (collecting, sorting and recycling packaging) falling on producers. Responsibility for packaging waste disposal requires industry to pay for waste management. In order to effect this scheme, industry formed the Duales System Deutsch (the Green Dot System). This is a non-profit organisation which is made up of fee-paying companies whose contributions go towards operating the organisation. This approach to waste management recognises the complex nature of the waste generation process and the need to involve related economic agents.[1] The European Union has agreed upon a community-wide packaging directive which sets minimum and maximum targets for recovery and recycling. In the United States and Canada individual states have introduced various initiatives, such as household kerbside charges (user-pays) schemes for household waste disposal as an incentive to reduce the amount of waste generated.[2] At the production stage, certain US states have introduced recycled content standards requiring producers to employ a given fraction of secondary materials (Palmer and Walls 1997). In addition to the above policies, many countries have also introduced eco/green labelling programmes as a means to inform consumers about the environmental impact of purchases (Menell 1995).

Government intervention in waste management is often rationalised because many of the costs (noxious smells, disease, environmental degradation) of waste generation and disposal are external in nature and decentralised markets cannot fully internalise them. An important decision in implementing and efficiently achieving any policy targets is thus the choice and combination of economic instruments and environmental regulation

[1] Brisson (1993), Rousso and Shah (1994) and Michaelis (1995) review the Green Dot scheme.

[2] For more on various countries' waste management policies, see Macdonald *et al.* 1996, chapter 10.

employed. Incorrectly chosen policy instruments can defeat the objectives of well-conceived policy.

A commonly articulated policy proposal as part of the solution to the underpricing is the introduction of kerbside charges for the collection and disposal of household waste. For example, the Industry Commission (1996)[3] as part of its investigation into packaging and labelling recommended that:

> In so far as is practical, waste disposal charges should be fed down to individual decision makers in the waste chain. In particular, post consumer waste collected at the kerbside should move more fully towards user pays systems with users billed according to use. (Recommendation 6, p. XXVII)

A similar recommendation has been made by Macdonald *et al.* (1996), in an evaluation of Australian efforts to reduce waste and encourage greater levels of recycling. This approach to reducing the stream of household waste has been considered at length in the literature. The reason for this is easy to understand. Typically, households make a flat-rate payment for waste disposal services through local government rates. This means that households face zero price at the margin for generating additional rubbish. An increase in the flat-rate payment only provides incentives via the income effect and as waste disposal costs are only a small portion of household income, the income elasticity of waste disposal is low (Wertz 1976). It is therefore argued that kerbside charges should reflect household marginal costs of waste disposal. But it is not entirely clear if this policy recommendation is economically justified. It can be demonstrated that kerbside charges need not be a first-best economic solution because they may lead the household to choose the option of illegal disposal to avoid the charges. Controlling illegal disposal involves additional costs, i.e. monitoring and enforcement. From a social cost perspective *The Economist* (1993) suggests that the costs incurred as a result of illegal dumping are significantly greater than the costs of efficiently operating a landfill. If additional costs of enforcing legal household waste disposal are too high, then it may be more efficient to target waste generation at the production stage through environmental taxes which can reduce waste at the source of production.[4] In general, user fees, environmental taxes and monitoring and enforcement are interdependent and it is not possible to choose an optimal combination of

[3] The Industry Commission (1991) made similar recommendations, albeit with several caveats in relation to implementation.

[4] Hatch and Mules (1993) provide an interesting Australian case study of the necessary size of a packaging tax to internalise external costs.

policy tools that can be applied to all types of household waste. Other concerns expressed about household waste management policy relate to, for example, recycling rates and targets. Baumol (1977) notes that there is an 'aura of unmixed virtue that surrounds recycling activities' (p. 83), and that there is no reason *a priori* to assume that the marginal net product of recycling is positive. It is therefore interesting to ask if we are undertaking too much household waste recycling in Australia and whether or not other methods of waste management, such as landfill, are to be preferred economically, despite being unpopular with environmental groups.

In this article we review the theoretical and empirical economics literature on household waste management. In particular we focus on kerbside charges/user fees, evaluating their feasibility as a policy option. It is found that the broad policy recommendations are only optimal under certain conditions and that policy needs to become specifically focused. The type of waste to be disposed of needs to be explicitly taken into account, considering the disposal incentives faced by the household. Throughout the article (specifically sections 2 and 5) policy issues are mostly focused on Australia. In section 2 we begin by briefly detailing the existing legislation relating to household waste management in Australia. Section 3 provides an informal analysis of the basic economic principles underlying household waste management and categorisation of policy instruments. In section 4 the theoretical and empirical literature on waste management is critically reviewed. Given the findings of the earlier sections, in section 5 we identify areas which require further research and policy development for Australia.

2. Household waste management in Australia

It is estimated that every Australian produces 1 metric tonne of post-consumer waste per year (Macdonald *et al.* 1996). In terms of individual households elsewhere in the world, DSD (1995) estimate that household waste per capita in kilograms is 701 in the United States, 400 in Japan and 348 in France. In Australia the largest components of this waste are paper and cardboard, non-fibre biodegradable waste and glass. Although there has been a large increase in the amount of waste being recycled, it still only accounts for a third of the total post-consumer waste stream. Of this third, paper, cardboard and glass account for almost all materials recycled.[5]

[5] A trend observed in the United States is that between 1960 and 1993 household waste doubled in weight. The plastic component increased forty-eight times, while aluminium only doubled. For landfill pressures, plastic's volume share in the waste stream is roughly double its weight (Hadjilambrinos 1996).

However, household waste is only a small fraction of total waste in the economy when primary industries (mining, agriculture, etc.) are included (Industry Commission 1996).

2.1 Federal waste management strategies and legislation

In Australia, environmental legislation and planning are carried out by individual State governments rather than the Commonwealth, with most policies and strategies developed by cooperation between the Commonwealth and the States. In terms of household waste management and recycling, local government plays the most significant role in waste management and litter control. In all States either local government or special waste management authorities have responsibility for the collection, treatment and disposal of solid waste. They also have charge over collection and on-sale of recyclable materials pursuant to State legislation. However, it is the State that establishes policy which directly impinges upon waste management functions. At present in Australia all States and Territory governments are establishing waste minimisation legislation which will directly affect local government waste management practice (Macdonald *et al.* 1996).

Commonwealth waste management policy is based upon two strategies introduced in 1992. The first is the National Waste Minimisation and Recycling Strategy (NWMRS). The NWMRS aimed to improve recycling collection and use, aiming for a 50 per cent reduction in total waste going to landfill by the year 2000 measured by weight per capita based on 1990 levels. Under the NWMRS, local government was set the target of adopting a waste management plan by 1993. By 1994 it was estimated (Maunsell Pty Ltd 1994) that only 24 per cent of local councils had satisfied the target. The second strategy is the National Kerbside Recycling Strategy, which complements the NWMRS. Examples of some of the recycling targets set under this strategy for 1995 are 45 per cent of glass, 65 per cent of aluminium cans, 20 per cent of liquid paperboard cartons, 40 per cent of newsprint and various levels for different types of plastic such as 50 per cent for HDPE. Progress towards the targets for glass, newsprint and aluminium cans has been good. For other materials, such as some of the plastic wastes, progress is poor (Macdonald *et al.* 1996). These targets do not have any legislative force, they have been negotiated with industry and are based on achievable targets. The Federal government has been criticised for failing to make sure that industry and local government attempt to progress towards many of the targets (DEST 1996).

Although there are Commonwealth strategies for household waste, there is no comprehensive Commonwealth legislation. There are only *ad hoc* pieces of legislation. For example, the Commonwealth taxation law (the Taxation

Laws Amendment (no. 5) Act, 1992), allows firms to claim a deduction for expenditure on equipment used to manage waste and control pollution. However, legislation at the Commonwealth level targeting waste management further down the waste chain is either absent or at best counterintuitive. For example, recycled products are not necessarily given tax concessions, while household compost bins are subject to local government subsidy but they are not exempt from sales tax.

2.2 State legislation

In terms of State Government we will focus on Victoria as most employ a similar waste management system.[6] In Victoria local government was given the responsibility for waste management under the 1958 Local Government Act. The Victorian Environmental Protection Authority was established in 1970 and it has responsibility for pollution licensing, and State policy on waste management. At present local government may charge rates for waste management services based upon the 1989 Local Government Act. If local government wished to implement a kerbside charges scheme it would need to pass a local by-law. In 1992 Victoria passed the Environmental Protection (Resource Recovery) Act which is based upon the NWMRS.

The Victorian approach of employing landfill levies on local government is in keeping with all other States and Territories. The 1992 Act included the provision for landfill charges. A landfill levy of A$2 per tonne is paid by local government and A$3 per tonne by other users. This revenue is used to fund waste reduction programmes and to promote waste minimisation measures. Finally, in all States the unlawful disposal of waste is viewed as criminal. Each State has Litter Laws. In Victoria this is covered by the Litter Act 1987 (amended in 1991). There are on the spot fines of A$600 and court-imposed fines of up to A$4000, and even in certain circumstances prison. These laws only apply to individuals.

2.3 Waste management in practice

Changes in household waste management practice have not been in keeping with the aims of the NWMRS. As an indication, a recent survey of Australian local councils found that landfill was the most economically viable waste management option — indeed 39 per cent of councils are

[6] The major exception to this is the South Australian container deposit-refund scheme. See references cited within for more details.

planning future landfills.[7] In addition, waste going to landfill in Victoria rose by 25 per cent between 1990 and 1993. There is also a movement away from kerbside recycling in Victoria and the introduction of drop-off centres because of increased operational costs of collection. Where recycling schemes have been introduced, their operational efficiency has been questioned. For example, in Banyule it has been estimated by Regional Recycling of Victoria, which sorts and sells recycled waste, that up to 50 per cent of material is only suitable for landfill disposal (*Heidelberger* 1997).

To date, no council in Australia has implemented kerbside charges. There have been trials such as 'Wising up to Weight' in the City of Melbourne between 1991 and 1994 (Monash University 1995). The trial attempted to simulate a kerbside charges system but it did not yield any significant results due to technical difficulties and changes in local government boundaries. Another example was the trial in Lismore, New South Wales where a A$2 payment was given to households who did not put a bin out for collection. There was a recorded reduction of one third in waste for disposal, but it was found that the costs of implementation exceeded benefits. In Wangaratta, Victoria, a ban was placed on the disposal of recyclable materials in waste for disposal. The local council introduced a penalty of A$400 if caught and although the law was heavily publicised, there were no recorded prosecutions. The scheme did result in an increase in recycling scheme participation, from 54 per cent up to 72 per cent. In addition, other local governments have introduced a variable rate charge for differing bin sizes.

With litter, packaging is the single largest contributing element, 90 per cent. Analygon (1994) estimated that for every dollar spent in reducing litter, over five dollars of benefits are generated. The emphasis on litter control has been through the use of public education campaigns highlighting the impact of litter on the environment. Evidence of the impact of legal disposal charges on littering has been noted by Waste Service New South Wales (1995) in that they observed, albeit anecdotally, increased illegal dumping after tipping fees were increased. In terms of local government enforcing existing legislation, the Keep South Australia Beautiful campaign group suggested that Adelaide was being too lenient in that the 26 councils in the metropolitan area imposed less than 100 littering notices per year. In Victoria it is estimated that 3000 notices are issued annually.

[7] For a comprehensive review of the environmental impact of landfills see El-Fadel *et al.* (1997).

2.4. Summary

At present there is a wide array of initiatives and strategies in place across Australia for household waste management. Despite all these initiatives there is no comprehensive incentive structure in place. The present situation is more a mosaic of incentive mechanisms over the economy both within and across tiers of government. From the review of policy it is interesting to note that despite the introduction of Commonwealth strategies on household waste management and recycling, local councils still show a preference for the increased use of landfills. This raises the issue of whether or not the targets set in the various strategies are optimal from an economic perspective. In the case of Australia, which is a land-abundant country, are the benefits from reduced landfill use greater than the costs incurred in collecting, separating, sorting and transforming the waste materials as a result of recycling? To be able to address this type of question, the basic economic principles underlying household waste management are informally discussed in the next section.

3. Economics of household waste management: basic principles

To understand the basic economic principles underlying household waste management, we consider it in two separate stages for convenience. In the first stage, we identify the optimal amount of waste to be generated. In the second stage, we focus on the optimal mix of available waste management technologies. It should be noted, however, that these two stages need to be considered simultaneously in all practical waste management problems.

In a decentralised market mechanism, an individual economic agent equates the private marginal benefits of their economic activity with marginal private cost (MPC) of such an activity. As household waste is the by-product of consumption, we may naturally think of waste being generated in some fixed proportion of consumption. This fixed proportion can change depending on the effort the household makes to reduce post-consumption waste. Thus, the amount of waste generated will be determined when the marginal utility of consumption (MU) is equal to the MPC of consumption, the latter being the sum of the price of a consumption good and the price of waste disposal services; more precisely the condition will be $MU/\lambda = MPC$ where λ is the marginal utility of income. The MPC fails to reflect full social cost of consumption, however. If both the consumption good and the waste disposal service are priced at marginal costs, then the difference between the MPC and marginal social cost (MSC) will be environmental externalities which, by definition, evade the decentralised

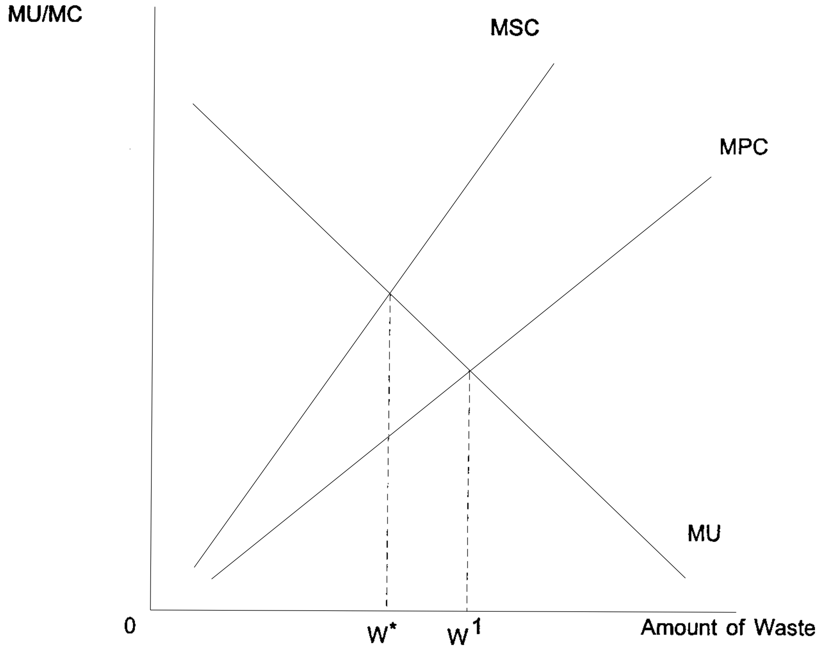

Figure 1 Discrepancy between MPC and MSC

market mechanism. Thus MSC generally exceeds MPC and using the standard externalities argument, the waste generated in a decentralised market is above its socially optimal level. This situation is illustrated in figure 1.

In figure 1, the socially optimal amount of waste is W^* where MU is equal to MSC while the decentralised market will lead to W_1. The question is then how to make decentralised decisions lead to a quantity of waste equal to W^*? Economic theory suggests various ways to solve this type of externality. The creation of a missing market for waste will have only limited applicability. Even if markets for recyclable waste are well established, not all wastes are recyclable. Moreover, defining property rights for the externalities, a necessary prerequisite for setting up a market, would not be easy. Because of this latter difficulty, the standard Coasian bargaining solution would not be practical in many cases. As an alternative, one can think of government intervention in both the product market and the market for waste disposal services. A tax on a consumption good can increase the MPC of consumption by increasing the price of the good. The effectiveness of such a tax depends of course on the nature of demand for and supply of the consumption good. The final increase in price of a consumption good as a result of the tax depends on the extent to which the tax is shifted to the consumer price. A tax on waste disposal services has a similar effect as it increases the price of waste disposal services. Thus, a suitably chosen Pigouvian tax on *either* the consumption good *or* the waste disposal service can shift MPC up to MSC in figure 1. Although the determination of an optimal tax requires careful measurement of environmental externalities, the argument for moving user-price closer to full social cost is obviously in line with this reasoning.

The discussion so far is valid if one assumes that the post-consumption waste is in a fixed-proportion relationship with consumption. However, the household can also undertake effort to reduce the amount of post-consumption waste for many household goods through, for example, reusing and composting. The household's waste reduction effort will be determined where the MPC of such effort (e.g. time, effort and money spent) is equal to its marginal private benefits (e.g. reduced waste disposal cost, and benefits from reuse or composting). In this case, the tax on consumption and waste disposal service may not have the desired effect. Consider the tax on waste disposal services. If the price of waste disposal services increases because of the tax, then the household may have an incentive to choose an alternative disposal method, say illegal dumping, to avoid higher disposal costs. Unless illegal dumping is monitored and offenders fined, the tax on waste disposal services will not shift MPC, in figure 1, up to the desired position. Moreover, it is likely that MSC will move even further up with illegal dumping, resulting in an even larger discrepancy between MPC and MSC. This argument rests on the assumption that illegal dumping leads to more severe environmental externalities, seemingly an innocuous assumption. On the other hand, a low price of waste disposal service may not give the household enough incentive to make the waste reduction effort. Any policy on direct waste disposal services thus has to solve two conflicting goals of preventing illegal waste disposal and encouraging waste reduction effort. Such conflict can be resolved to a certain extent through a tax on consumption goods as it will indirectly reduce the amount of waste generated. Note that such a tax has only an indirect effect on the household's waste reduction effort which may not lead to the optimal amount of waste, W^*, in general. We can thus conclude that, when the household can actively reduce the amount of post-consumption waste, there is in general a unique mix of a tax on the consumption good and a tax on waste disposal services that can achieve the second-best level of waste generated which may not be equal to W^*.

We now consider the second stage of the waste management process, the optimal mix of waste disposal technologies. The problem of waste disposal is more complicated than a standard externality example. This is because not only is there a decision about the optimal quantity of waste to dispose of (the first stage), there is also a decision to be made about the optimal mix of

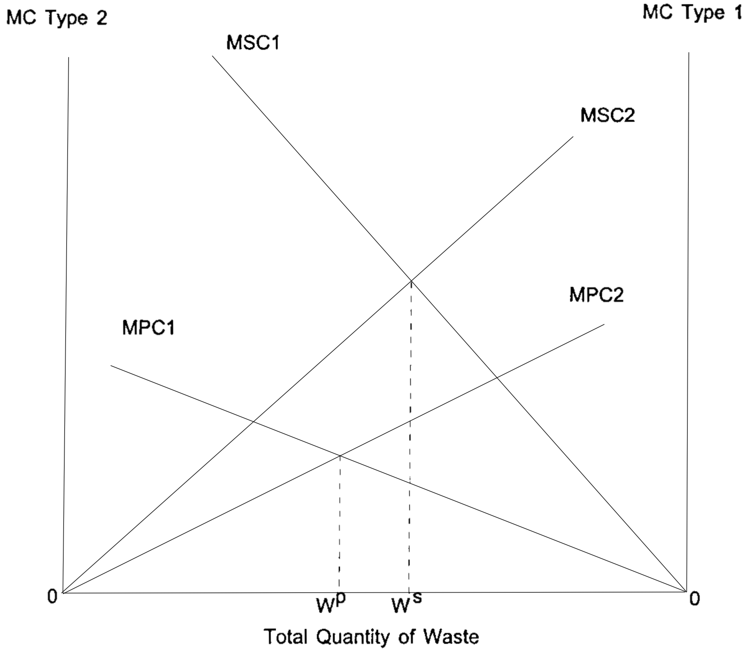

Figure 2 Optimal mix of waste disposal technologies

disposal options. This idea can be explained using figure 2. On the horizontal axis is the total quantity of waste to be disposed of, which is determined in the first stage. On the left-hand vertical axis is the MC associated with one type of waste disposal method and on the right hand we have the MC of another. For example, the two disposal technologies might be landfill and recycling. The optimal mix of waste technologies is derived by equating MSC of both. From figure 2 this yields a mix W^S. However, if we equate their MPC we derive a different mix, W^P. It is therefore necessary for each disposal option to be correctly priced. Otherwise the mix of waste disposal technologies will be non-optimal. The difference between the two technologies is the difference between their respective MSC and MPC. The bigger difference between MPC1 and MSC1 than for type 2 is a reflection that type 2 is the preferred technology from a social perspective. When the mix of technologies is based upon MSC1 = MSC2, then more of type 2 technology should be used as compared to technology mix associated with MPC1 = MPC2.[8]

[8] Another important issue is service delivery and whether it should be provided by local government directly or tendered out for private provision. See Bivand and Szymanski (1997).

As mentioned earlier, the two stages of waste management are not separable in general. The marginal costs and benefits described in figure 1 depend on the optimal mix of waste disposal methods considered in figure 2 and vice versa. Wiseman (1991) makes the same point by explaining that total costs are minimised when the MC of alternative waste disposal technologies are equated. Although the economic principle explained above is based on simple diagrams, it nonetheless demonstrates an important point: waste management needs to be analysed in a comprehensive framework where various policy instruments targeting consumption, waste disposal services and illegal waste disposal can be considered simultaneously, along with the choice of waste disposal technologies. For example, the introduction of a user-pays scheme in an attempt to move MPC closer to MSC in figure 1 may not achieve the intended outcome if the household chooses the option of illegal waste disposal. On the other hand, keeping the price of waste disposal services low will not give enough incentive to households to undertake waste reduction effort. The application of the principle thus invites the need to study the exact nature of waste generation and the extent to which the household waste reduction effort is significant. Furthermore, as waste services are frequently financed by flat rate taxes, the MC of waste generation for the household is zero, which in turn prevents the equalisation of MC of the various disposal technologies and hence the efficient disposal of waste.

3.1 Categorisation of policy instruments

To help our understanding of how a waste management policy can affect economic incentives, it is useful to divide incentives into three categories as suggested by Fenton and Hanley (1995). These are:

1. Purchase-relevant instruments;
2. Discard-relevant instruments;
3. Jointly-relevant instruments.

Purchase-relevant (upstream) instruments are those that affect the pricing of the product which in turn generates waste. In figure 1, the use of a purchase-relevant instrument can shift MPC by changing the price of a consumption good. These instruments affect the incentives of consumers by changing relative prices of consumption goods. Take, for example, a packaging or production tax. Producers using the packaging can reduce the tax payment on their product by reducing packaging material per unit of their product sold. On the other hand, consumers will demand less of a product whose price increases due to a packaging tax or purchase a product that uses less packaging material. Purchase-relevant instruments therefore affect the incentives of both consumers and producers.

Discard-relevant (downstream) instruments are targeted at the waste disposal stage. An example is a user fee for household waste collection and disposal. This type of instrument will shift the MPC in figure 1 by changing the price of waste disposal services. Also if the user fee is charged on the waste which goes to landfill, but not on the waste that is recycled, then it can encourage more recycling. Facing a higher user fee for waste disposal, consumers can reduce waste to be disposed of by exerting more effort in waste reduction and recycling although too high user fees can lead them to choose the option of illegal waste disposal. Monitoring of and the penalty for illegal waste disposal are thus discard-relevant instruments as well. Another channel whereby discard-relevant instruments can affect the amount of waste is by altering consumption behaviour. For example, consumers may change purchase decisions to buy goods that have a lower amount of intrinsic waste.

Jointly-relevant instruments affect both the purchase decision and the discard decision. An example is the deposit-refund on bottles which is employed in several countries throughout the world (Porter 1978; Bohm, 1981; Lee *et al.* 1988; Menell, 1990; Brisson, 1993): the consumer pays a levy on the purchased product and receives a refund when the bottle or container is returned after use. The objectives of this type of instrument are to reduce the amount of waste entering the waste stream and to reduce the need for the production of new bottles.

As Fenton and Hanley (1995) note, the dominant form of economic analysis has been the assessment of individual instruments instead of some complementary mix. An optimal waste management policy generally requires the simultaneous consideration of instruments. For bottles and containers, jointly-relevant instruments may be effective as they can change consumption without introducing the possibility of illegal waste disposal. For other waste materials, the use of jointly-relevant instruments is limited. Take, for example, food waste. As it is likely that the household's waste reduction effort can be significant for food waste, a discard-relevant instrument such as a user fee should be chosen properly to provide the incentive for waste reduction effort. However, the user fee in this case has its limits as it may lead to illegal waste disposal, which will have to be monitored at additional social cost. A purchase-relevant instrument can complement the use of these discard-relevant instruments by changing consumption, and thereby the amount of waste to be disposed of.

3.2 Summary

From the foregoing analysis it has been shown that to introduce an efficient household waste management policy requires policy-makers to be aware not only of the costs and benefits of implementing a given waste disposal technology but also its costs and benefits relative to other disposal technologies. The importance of these findings for Australia is that when setting policy targets, such as the 50 per cent reduction in waste going to landfill, unless all such costs and benefits have been considered, there is no guarantee that policy will be efficient. If this is not the case, then the various policy initiatives are likely to yield sub-optimal outcomes. The important lesson for Australian household waste management policy is to properly evaluate proposed policy options and not to let unwarranted social pressures dictate.

4. Review of literature

In this section we review existing theoretical and empirical literature on household waste management. Throughout this section, the focus will be on how the policy instruments described above are incorporated in the literature and how the suggested policies square with the basic economic principles developed in the previous section. It will be shown that most existing theoretical studies fail to address comprehensively the household waste management problem. Consequently, conclusions drawn from these studies are often applicable only to a limited class of household wastes. On the other hand, the focus of the empirical research is primarily on the estimation of household responses to discard-relevant instruments only, inviting further research which incorporates other policy instruments as well.

4.1 Theoretical literature

For convenience, we divide the literature into two strands. The first strand of literature pays attention primarily to the household's response to various discard-relevant policy instruments. These policy instruments are given *exogenously* so that the issue of optimal waste management policy is ignored. The second strand of the literature characterises optimal household waste management policies by determining policy instruments endogenously as the solution to some optimisation problem. This approach incorporates the household's response to policy instruments within the optimisation framework.

Exogenous policy instruments

In Wertz (1976), a theoretical model is developed to study the demand by households for solid waste disposal services. Wertz considers the impact of four specific variables on the generation of household waste: the price of waste removal; the frequency of collection; distance to the nearest refuse site; and household income. He shows that the utility from consumption needs to be traded off against the disutility associated with the waste generated from consumption. Not surprisingly, consumption is negatively related to the user fee for waste disposal and the disutility from the waste generated.

Following Wertz, Efaw and Lanen (1979) studied the household's demand for waste disposal services, but with a focus on how households would respond to container-based user fees. They assumed that the quantity of waste is a function of the number of containers, which in turn is a function of user fees and income. The important aspect of their paper relates to the observation that the changes in user fees can be moderated by the household if 'stomping' (volume reduction) of waste is carried out. Thus a container-based user fee system provides an incentive for the household to reduce the volume of the waste produced but not necessarily the weight, which was also noted by Fullerton and Kinnaman (1996).

Using a household production model, Jenkins (1993) studied the household's choice of effort in terms of waste disposal and recycling. This is an interesting and important choice faced by the household in that additional time and effort are needed for recycling as opposed to conventional disposal. The more time that is needed for recycling reduces the time available for leisure activities. She shows that the household will respond to positive waste disposal fees by reducing consumption of goods that yield larger amounts of post-consumption waste. However, for consumption goods producing recyclable waste, whether the household will respond to positive disposal fees by more recycling depends on the marginal value of leisure time foregone for recycling relative to the size of the disposal fees. Overall, Jenkins shows that the household will increase the time spent on recycling when faced by positive user fees.

Morris and Holthausen (1994) also adopted a household production model which they used to simulate policy alternatives. Like the previous papers, they show that positive waste disposal fees encourage the household to reduce waste, which is partly offset depending on the ease of recycling. This model was further extended by Jakus, Tiller and Park (1996) in an analysis of recycling by rural households. Jakus *et al.* note that the model developed by Morris and Holthausen implied that individuals would purchase a good for the additional benefits derived by increased recycling. The Jakus *et al.* model removes this behavioural anomaly. A fuller discussion of the policy implications stemming from these papers is left to the next section.

In sum, the papers reviewed in this section share several restrictive features. First, attention is paid only to discard-relevant instruments such as waste disposal fees. Second, the household's response to user fees is restricted to waste reduction or recycling only, omitting the option of illegal waste disposal. Perhaps most importantly, the policy variables that are studied are exogenously given which ignores the issue of optimal waste management. To analyse optimal waste management, a more comprehensive approach is necessary in which the household's response to policy instruments is embedded in an explicit policy objective. The optimisation of this objective over various policy instruments can then determine the optimal policy mix endogenously.[9] The next section reviews the literature which undertakes this task.

Endogenous policy instruments

Recently there has been a great deal of interest in which phase of the waste chain at which to introduce appropriate forms of policy instruments. Specifically, the question asked is whether incentives should be introduced through upstream purchase-relevant instruments or downstream discard-relevant instruments. All the papers reviewed in this section have a common feature in that several policy instruments are considered simultaneously. The difference between these papers is the generality with which policy implications drawn from a theoretical model have applicability. For example, Dobbs (1991), Dinan (1993), Palmer, Sigman and Walls (1997) and Palmer and Walls (1997), concur with the optimality of jointly relevant instruments such as a deposit-refund system. However, their optimal policy is only applicable to a limited class of wastes. Fullerton and Kinnaman (1995), and Choe and Fraser (1997, 1998) provide more general frameworks where purchase-relevant and discard-relevant instruments are considered together and in which the deposit-refund system emerges as optimal only in a special case.[10] An important difference, however, is that Fullerton and Kinnaman did not consider the household's effort to reduce waste and, consequently, their model is another special case of the model studied by Choe and Fraser.[11]

[9] Some papers focus on the choice of disposal technologies only, assuming that the waste generation process is exogenous. See Smith (1972), Keeler and Renkow (1994) and Highfill and McAsey (1997).

[10] It should be noted that the papers by Dinan (1993), Palmer, Sigman and Walls (1997), Huhtala (1997) and Palmer and Walls (1997) are not specifically focused on household waste.

[11] For details of papers dealing with waste management from a production perspective exclusively, see Miedema (1978), Copeland (1991), Jenkins (1993), Parry (1995) and Conrad (1997).

Dinan (1993) notes that there has been much interest shown in the introduction of a virgin materials tax (Miedema 1983) but that the arguments employed to justify this policy are misplaced. She develops a model that shows that a virgin materials tax, used in isolation, is an inefficient method of reducing waste but that the combination of a disposal tax and reuse subsidy is efficient (deposit-refund). This is a form of jointly-relevant policy instrument in that a tax is levied on producers of consumption goods and a subsidy is given to end-users of recycled materials.[12] Dinan states that the deposit-refund approach is theoretically consistent with household-based disposal fees, but that it has certain advantages in that incentives for illegal disposal are removed. Palmer *et al.* (1997), in a partial equilibrium analysis of waste generation and recycling, also argue that a deposit-refund policy, equivalent to the type described by Dinan, is the most cost effective when compared to other policy options (i.e. an advance disposal fee and a recycling subsidy). In a related paper, Palmer and Walls (1997) consider the effectiveness of employing recycled content standards as a means to reduce waste. They find that recycled content standards suffer from many of the same problems as a virgin materials tax. They demonstrate that it is possible to find an optimal recycled content standard if it is used in combination with additional taxes or subsidies. However, implementation of this policy portfolio is difficult to envisage. Again like Dinan, and Palmer *et al.*, Palmer and Walls (1997) advocate the use of a deposit-refund policy. A basic limitation of this proposal is that a deposit-refund policy can only be applied to certain materials in the waste stream for which recycling and reuse possibilities are significant. This point is borne out in that Palmer *et al.* only concentrate in their analysis on five common recyclable materials: paper, glass, plastic, aluminium and steel. For some waste materials there is no recycling or reuse possibility under current technological restrictions. In addition, the transaction costs of implementing a deposit-refund policy are high, although Palmer *et al.* using simulation analysis have shown positive net benefits from its implementation.

A paper that considers illegal disposal explicitly is Dobbs (1991). This paper models the choice between two alternative waste disposal options for the household — littering and legal waste disposal. By considering the two simultaneously Dobbs proposed that a refund or user subsidy should be given for household waste disposal. The idea that a simple Pigouvian tax can be employed to solve the existing externality is shown to be incorrect. Dobbs shows that welfare gains are achieved using a combination of instruments,

[12] In the United Kingdom a system of recycling credits has been detailed in the 1990 Environmental Protection Act Part II (Turner, Ozdemiroglu and Steele 1995).

a consumption tax and waste disposal refund, as opposed to each separately. Once this interdependence is recognised, the optimal user charge is 'quite likely to be negative' (p. 222). This instrument falls into the jointly-relevant category of Fenton and Hanley (1995). The major problem with drawing parallels with the deposit-refund system is that it is generally applied to materials such as glass. As with Dinan (1993), Palmer *et al.* (1997), and Palmer and Walls (1997), it is not a credible approach for materials that simply need to be disposed of and for which high transaction costs are incurred in implementation.

The approach to simultaneously considering economic instruments is extended by Fullerton and Kinnaman (1995). In this paper a comprehensive general equilibrium model of waste management is developed. Like Dinan, Fullerton and Kinnaman consider both production and consumption decisions. However, unlike Dinan, they stress the potential problems of illegal dumping by households in their model. With illicit dumping, a first-best solution can be achieved by waste-end taxes on household waste and on illicit dumping. However, they argue that a tax on illicit dumping is difficult, if not impossible, to enforce and monitor. They propose that a zero tax on illegal dumping is optimal, as long as the relative prices of all other policy instruments are calibrated to yield the optimal outcome. Thus, they demonstrate that it is possible to adjust the economic incentives on all other relevant activities, inducing the first-best solution simply using relative prices. The first-best solution removes the problem of dumping by manipulating other incentive prices. With the dumping tax set equal to zero, the relative price of legal and illegal disposal is found by subsidising legal disposal which acts as a cross-subsidy on consumption, and so a tax on consumption is also necessary.

This solution characterises the difference between upstream (purchase-relevant) and downstream (discard-relevant) instruments to waste management. Fullerton and Kinnaman use an upstream approach arguing that illicit dumping cannot be enforced. The consumption tax is set at a rate to reflect the externality from illicit dumping and the tax is returned as a subsidy on correct disposal. This in effect amounts to another deposit-refund type system. Like Dinan they rule out the use of a virgin materials tax except to correct for any externalities generated as a result of production. They explain that downstream taxes, including a tax on illegal dumping or burning, can be used for the first-best solution in their model. However, an upstream tax is easier to implement because of the informational difficulties associated with hidden actions — illegal behaviour. As Fullerton and Kinnaman note, however, the preferred solution is being determined as a result of illegal actions by households: 'The tail wags the dog, since illicit dumping is corrected by multiple taxes on all inputs other than recycling, combined with a subsidy on all output' (p. 87).

Although their model is one of the most comprehensive to date, Fullerton and Kinnaman incorporate several strong assumptions. First, they allow for the existence of recycling of some waste that can be reused in the basic production process and they assume that this input can displace virgin material. An important issue overlooked is the costs incurred in the recycling process and the feasibility of substituting (First and Second Laws of Thermodynamics) the recycled material for virgin inputs. The costs incurred in recycling are not small (Baumol 1977; Fairlie 1992; Wills 1997) and need to be included in a model of this type.[13] Second, the evidence that exists about the feasibility of selling recycled materials is unclear. For example, in the case of lead (Sigman 1995) and steel (O'Neill 1983) it has been found that a market in secondary materials is feasible. This results from the fact that the vast bulk of the waste is easy to collect (from a factory, for example), is not contaminated by other waste materials and is easy to transform into resaleable form. This can be contrasted with household recycled waste which requires significant cost to collect, separate, treat and to recover a usable resource. This corresponds with the findings of the recycling programme in Banyule, Victoria. Also in the case of plastics (Hadjilambrinos 1996), and paper (Nestor 1992, and Hanley and Stark 1994), recycling is frequently unattractive from an economic perspective, although in terms of social cost-benefit analysis there may be a case for government support (Hanley and Stark 1994). It should be noted that in the case of paper, recycling produces substantially more CO_2 than in the production of virgin paper, because virgin paper requires trees to be grown which in turn sequester CO_2 from the atmosphere (Industry Commission 1996).

Although the models reviewed so far have incorporated a set of economic instruments to improve household waste management, they pay insufficient detail to the issue of effort expended by economic agents in relation to waste generation. Choe and Fraser (1997, 1998) have addressed theoretically the issue of waste reduction effort in a comprehensive modelling framework. They simultaneously model the interaction of a producer, consumer and an environmental agency. Effort expended by the producer and consumer in relation to waste is modelled in terms of the type of waste generated. The reason why this approach is taken is that for many waste materials, such as packaging, the only decision faced by the household is legal or illegal disposal. The choice of whether to recycle or reuse is not available. Similarly, for many products a firm cannot reduce the amount of waste intrinsic in

[13] An alternative approach to analysing resource use and waste management is a materials-product chain analysis. Example are provided by Opschoor (1994) and Kandelaars and van den Bergh (1997).

the final product under the available production technology. Recognising this significantly alters the optimal policy response in relation to kerbside charges. In addition, unlike in Fullerton and Kinnaman, particular attention is paid to the problem of illegal dumping by households and the necessary monitoring and compliance incentives that need to be employed. The trade-off between the costs of monitoring, size of the fine and the costs incurred by households elsewhere in the waste chain is analysed. The most significant result provided is that in relation to household waste: kerbside charges need not be an optimal policy response. Under certain circumstances when the waste can be recycled or reused, a kerbside charge may be optimal. For many waste materials, however, it can be desirable to collect household waste free while levying a tax on the waste component of the consumption good. An optimal policy therefore depends on the type of waste and the extent to which producers and consumers can take action to reduce waste. The policy recommendation of Fullerton and Kinnaman therefore turns out to be a specific result of Choe and Fraser.

4.2 Empirical literature

In the previous section we considered the theoretical literature on household waste management. Those papers that focus specifically on household responses to kerbside charges form the basis of empirical research into various aspects of household waste management. It is the empirical results provided by the literature on kerbside charges to which we now turn.[14] The focus of the empirical literature has been the responsiveness of households to the introduction of kerbside charges and recycling schemes. Most empirical research has concentrated on the calculation of elasticities (own price, income, cross-price) to assess the response to policy. Table 1 summarises the elasticities generated in the literature to date.

As can be seen in table 1, the initial interest in household waste management spawned several studies in the 1970s. Since then the focus of empirical research has become sharper and the data sets constructed to estimate elasticities better suited to answering specific policy questions. The form of econometric method employed has become more sophisticated (Probit, Generalised Least Squares), and the use of specifically generated survey data has increased (Fullerton and Kinnaman 1996). We will discuss the implications of the findings reported in table 1, in publication date order, for those studies of more importance.

[14] There is also a large empirical literature which considers deposit-refund schemes and various aspects of recycling. See reference cited within.

Table 1 Elasticity estimates for household waste disposal services

Authors	Data and estimation method	Dependent variable	Price elasticities	Income elasticities	Cross-price elasticities
Fullerton and Kinnaman (1996)	Individual Household Survey (1992), Charlottesville, VI – Arc Elasticity Calculations		– 0.076 (by weight) – 0.058 (by weight / seasonally adjusted) – 0.226 (by volume)		0.073 (waste disposal with recycling)
Strathman *et al.* (1995)	Seasonally Adjusted Monthly Data (1984-1991), Portland, Oregon – OLS	Tonnes of solid waste per 1,000 of the population	– 0.47		
Reschovsky and Stone (1994)	Household Survey (1990), Tompkins County, NY – Probit	Dichotomous, recycling or not		0.23 (by volume) 0.24 (by weight)	
Morris and Holthausen (1994)	Simulation – Household Production Model		– 0.51 to – 0.6		0.51–0.59(waste disposal / recycling) 0.97–1.49(waste disposal / reduction)
Jenkins (1993)	Panel Data, Cross-Section of 9 Communities (For more details see table 5.1, pp. 64 Jenkins, 1993) – GLS	Quantity of residential waste discarded per capita per day	– 0.12	0.41	
Hong *et al.* (1993)	Population and Household Survey (1990), Metropolitan Service District, Portland OR – (i) Ordered Probit (ii) OLS / 2SLS	(i) Recycling Effort (ii) Quantity of household waste	(ii) 'low' (inelastic) but statistically insignificant	(ii) 0.049	
Skumatz (1990)	Annual Average Revenue Data (1971–1987), Seattle – OLS	Average annual residential pounds discarded per capita	– 0.14		
Efaw and Lanen (1979)	Monthly Data (4 years mid 1970s), Three US cities – OLS / 2SLS	Weight of refuse	'highly inelastic' but insignificant		0.2–0.4 (waste disposal / recycling)
Richardson and Havlicek (1978)	Survey and Census Data (1972 / 1970), Indianapolis, Indiana – OLS	Quantity of *k*th component in pounds per household per week		0.242 (also have by waste material)	
Wertz (1976)	(i) Cross-Sectional Data (1970), 16 suburbs Detriot- OLS (Income Elasticity), (ii) San Francisco (1970) – Arc Price Elasticity	(i) Annual pounds refuse collected per capita	(ii) – 0.15	(i) 0.279 and 0.272	
McFarland (1972)	Cross-Section Data (1967 / 68), 13 US cities in California – OLS	Annual per capita quantity of household waste	– 0.455	0.178	

Wertz (1976) is among the important early studies shown in table 1.[15] From his theoretical model Wertz set out to test two hypotheses:

1. That the quantity of household waste varies negatively with a user fee; and
2. That the quantity of household waste varies positively with income.

Wertz employed cross-sectional data for 1970 for ten suburbs of Detroit. Using OLS he derived an income elasticity of waste of 0.279. The estimation was repeated for six further suburbs in Detroit yielding a value of 0.272. To test his hypothesis on demand for household waste services Wertz collected only two data points for 1970, the quantity of waste discarded per capita in San Francisco (699 pounds) and the quantity of waste discarded per capita in all urban areas of the United States (937 pounds). San Francisco was selected because its residents had to pay for waste disposal by volume. From these an own price arc elasticity of demand for household waste services of -0.15 was found. An important shortcoming of this analysis is that the empirical analysis does not distinguish between waste generated and waste discarded. The empirical work considered the latter, whereas the theoretical model considered the former. This is an important distinction when faced with the prospect of recycling. Despite the limitations, the findings of Wertz have proven to be a starting point for many subsequent studies.

Efaw and Lanen (1979) developed a theoretical representation of the household which was subsequently employed as the basis of their empirical work. In this paper the data used were collected from three cities (Sacramento (CA), Grand Rapids (MI), Tacoma (WA)). In each of the cities households faced different types of user fees. They collected monthly data for the 1970s for periods up to four years for each of the three cities. For each city linear equations incorporating explanatory variables specific to each city were included. They found that demand for household waste services did vary positively with income, but that the response of households to user fees was statistically insignificant. These results need to be treated cautiously. First of all they faced general data problems which meant that in some cases crude proxies needed to be employed in the estimation. Second, and probably more importantly, each of the demand equations estimated was an oversimplification in that expected explanatory variables were not included. For example, Jenkins (1993), notes that, ‘A blatant oversight is not considering the prices of goods that produce solid waste’ (p. 21). Another

[15] The discussion of the earlier studies considered here, McFarland (1972), Wertz (1976), Efaw and Lanen (1979) and Skumatz (1990), is based heavily on chapter 2 of Jenkins (1993).

variable not included in the analysis is the market price of recycled goods and products. Jenkins further criticises this paper because the authors fail to maximise the usefulness of their data. She argues that they should have pooled the data to yield more efficient estimates.

Of the earlier studies McFarland (1972) used revenue per tonne as a crude proxy for a user fee. As Jenkins (1993) notes, this is crude approximation as average revenue will not reflect the diversity of user fees amongst a community. McFarland collected his data in 1967 and 1968 in thirteen cities across California. The cities were selected because each employed user fees and information on the quantity of household waste was available. The user fee variable employed by McFarland is a service-level fee which does not impose a positive cost on households for waste collection. Therefore, the interpretation given by McFarland to his results can be considered to be liberal. A similar proxy for price was employed by Skumatz (1990). In this paper annual data (1971–87) was used in a way similar to McFarland, although according to Jenkins the results derived are more consistent for the latter period of the data set.

Jenkins (1993) used panel data (a cross-section of communities) to estimate demand equation for household waste services. There were five communities with user fees and four without to be used for comparison. The difficulty in constructing this data set is described by Jenkins and this can be seen as a comment on the haphazard and limited information on this subject at an appropriate level for carrying out serious empirical economic analysis. The equations estimated by Jenkins relate to the theoretical model discussed above. Using GLS Jenkins found that user fees were negatively related to household waste production. As can be seen in table 1, the elasticities results obtained by Jenkins are within the range of values reported by other studies.

A very different approach to estimating household elasticities is the paper of Morris and Holthausen (1994). In this paper a household production model is used to simulate responses to different waste technology adoption using calibration techniques. Morris and Holthausen estimate that waste disposal service has an own price elasticity of between -0.51 and -0.6, which is higher than the existing econometric literature would suggest. They also find that the cross-price elasticity with respect to recycling is negative and less than 1. However, the cross-price elasticity for waste reduction is near to 1 or higher. Although these results would seem to be slightly at variance with other studies this paper does provide an interesting alternative approach to elasticity estimation.

A more conventional empirical study of household waste management is that of Reschovsky and Stone (1994). They employ an econometric model to analyse actual consumer responses to quantity-based pricing using survey data obtained from Tompkins County in the Finger Lakes Region of upstate New York. The survey was carried out in September 1990. Reschovsky and Stone randomly selected 3 040 households with 1 422 returning. This county was selected because it implemented a per-unit pricing system in March 1990. It did so on the assumption that households need incentives to motivate participation. The dependent variables in the analysis are dichotomous indicating recycling or not for a particular material type. Independent variables include income, education and age. Given the form of the dependent variable Reschovsky and Stone estimated their equations using a Probit. As can be seen in table 1, the results derived are similar to Wertz (1976).

Reschovsky and Stone allow household waste to be disposed of as rubbish or recycled. They note that there may be an altruistic motivation for recycling. However, mandatory recycling imposes an additional cost on the household and frequently it leads to too much recycling. In relation to illegal dumping Reschovsky and Stone's survey of the evidence suggests that much of the illegal dumping that happens uses alternative disposal sources such as roadside dumpsters. They were unable to ascertain a clear indication of illegal dumping or burning though. They did, however, find that households are sensitive to MPCs of waste reduction but that they are less sensitive to the costs of waste disposal. This implies the need to have recycling efforts along with quantity-based fees (volume or weight). Fees alone will be unpopular and ineffective.

Hong, Adams and Love (1993) also consider the responsiveness of households to price incentives to undertake recycling. They note that the existing literature (Richardson and Havelicek 1974 and 1978; Saleh and Havelicek 1975) identifies household income and size as important explanatory factors in the generation of household waste, as well as the disposal services provided (Wertz 1976). The empirical results generated by Hong *et al.* suggest that the income elasticity for garbage collection is less than 1. Furthermore, they found that a user-fee for household waste disposal does affect recycling behaviour positively, reinforcing the findings of Reschovsky and Stone (1994). It should be noted that the kerbside recycling service employs zero marginal cost pricing. Interestingly, the responsiveness of households to the user fee with respect to demand for waste services appears to be insensitive to price increases. However, although there is no simple association between income and aggregate household waste, there is a relationship for separate parts of the waste stream. Hong *et al.* also observe that a household is less likely to actually undertake recycling if a high degree of effort is required and that a bigger household is more likely to undertake recycling and to demand more frequent waste collection. However, previous work (see Jenkins 1993) has found that average household size is positively related to waste production but it is not statistically significant. On a related matter Kemper and Quigley (1976) found that the number of collection visits per year is not significantly related to the annual quantity of waste discarded.

A different approach to estimating household demand for waste disposal services is provided by Strathman, Rrufulo and Mildner (1995). In this paper a distinction is drawn between the demand for waste collection services and the demand for landfill disposal. An important observation relating back to the analysis related to figure 2 is made as a result of this distinction here. Although there may exist efficiency gains from introducing marginal cost pricing for waste collection, these will not be realised if the landfill disposal price is below its opportunity cost. Strathman *et al.* also explain that depending upon the focus of the research, different information is required to estimate the demand for household waste disposal and the demand for landfill. They note that the two demand estimates will be related and they use this in their estimation of household demand for waste disposal services. Assuming that landfill charges are passed on in full to waste collection charges, they assume that the demand elasticities have the following relationship

$$\varepsilon_c = \frac{\varepsilon_d}{\rho_d}$$

where ε_c and ε_d are the demand elasticities for collection and final disposal and ρ_d is share of final cost of disposal in price. Strathman *et al.* estimate a demand equation using OLS, which is seasonally adjusted and corrected for serial correlation. Their data are drawn from Portland and are monthly over the period 1984 to 1991. The results of their estimation for landfill yield an own price elasticity of demand of -0.11. Based upon this estimate they find that the household own price elasticity of demand using the specified relationship is -0.45. This is somewhat larger than estimates derived in comparable studies. They suggest that this is because of data difficulties, and because in the residential area used in the study the ability of households to illegally dump is high because it is a relatively large region. The findings of Strathman *et al.* (1995) have been questioned by Nestor and Podolsky (1996), who argue that the elasticity estimates derived are overestimates because no allowance is made for substitute waste disposal options. Although in general the criticisms are rejected by Strathman, Rufulo and Mildner (1996), they do acknowledge that price-induced increases in illegal dumping do inflate the estimate for the price elasticity of demand for collection services.

Another econometric study of the household waste problem is provided by Fullerton and Kinnaman (1996). They study the behaviour of households after the introduction of a unit pricing system for household waste disposal, focusing on the incentives for illegal behaviour. Fullerton and Kinnaman use individual household data collected from a specially designed survey which specifically measured garbage (volume and weight)[16] rather than the weekly number of bins contracted to assess 'Seattle Stomp'. Also the weight of recycled materials was measured, not just the frequency of collection. The data were collected in Charlottesville, Virginia, which had just introduced a unit pricing programme. Fullerton and Kinnaman surveyed 400 households, offering a US$5 incentive to complete the survey questionnaire. In total, 97 households agreed to participate. To avoid seasonal effects Fullerton and Kinnaman controlled the collection and adjusted their final results accordingly. The own price elasticity estimates derived are smaller than those found in previous research.

To measure illegal dumping Fullerton and Kinnaman (1996) use two proxy measures. They suggest that illegal dumping can account for between 28 and 43 per cent of the reduction in garbage after the introduction of user pay schemes, confirming the findings of Reschovsky and Stone (1994). Although Fullerton and Kinnaman consider illegal dumping extensively, a problem with this study — and others attempting to assess illegal dumping — is that it does not provide a direct measure of illegal behaviour. Fullerton and Kinnaman do not know what action 'dumping' actually describes, although they provide some indirect verbal evidence about observed increases in littering. However, it is unclear if this evidence can be relied upon as responses might be perceived as opposed to real in the light of the introduction of policy (prior belief is that there will be an increase in illegal dumping and littering). Strathman *et al.* (1995) note that in the Portland area local officials when questioned did not know if illegal dumping had increased because there appeared to be no jurisdiction monitoring and enforcing sanctions for illegal dumping.

Many of the above empirical findings on household responsiveness to waste charges have been used as the basis of cost-benefit analysis (CBA), evaluating the overall social implications of implementation. The use of CBA to evaluate many of the waste management policy options has been high.[17] In terms of user charges existing results yield conflicting evidence. Jenkins (1993) suggests that there exists a very large welfare gain from the

[16] Social costs of household waste disposal depend more on volume after compacting at the landfill and are better proxied by weight at the kerb than by volume. Monash University (1995) argued for weight over volume in that a more precise measure of waste is provided. In addition, Monash note that volume-based systems discriminate against low density waste.

[17] For example, in relation to the introduction of a deposit-refund system for beverage containers, see Naughton *et al.* (1990), Alter (1993), Pearce and Turner (1993), and Brisson (1993).

introduction of user charges. Morris and Holthausen (1994) derive a Hicksian compensation measure which yields a value of US$117 a year to make the representative household no better off than it was before introducing unit pricing, curbside recycling, and a once-a-week collection. This means that annual income would have to be reduced by US$117 which therefore implies a substantial welfare improvement from the implementation of such a scheme. However, Fullerton and Kinnaman (1996) find that the benefits of unit pricing do not cover the administrative cost of implementation, implying that the introduction of user fees is not an economically sensible policy.

Finally, a new strand of empirical work has been emerging in the applied literature. In rural areas the availability of public or private kerbside waste services is limited and frequently non-existent.[18] There is a greater reliance on drop-off centres for waste and recycled goods. Several papers have recently considered the motivation of households to participate in waste recycling in this context. In the rural setting the decision to recycle cannot simply be interpreted as means to avoid additional costs of waste disposal. Instead it is suggested that recycling is undertaken for the 'warm glow' benefits. These papers (Jakus *et al.* 1996; Jakus *et al.* 1997; Tiller *et al.* 1997; Lake *et al.* 1996) have estimated the willingness to pay using several non-market estimation techniques (Contingent Valuation and Travel Cost) for recycling. A comparison of the findings of these papers provided by Tiller *et al.* shows that the costs of providing the recycling service are significantly outweighed by the benefits accruing to participants.

4.3 Summary

In the preceding sections a comprehensive review of the literature on household waste management has been provided. Emerging from the literature are several common themes of importance. First, to fully understand the complex nature of the waste production and disposal chain, policy needs to be evaluated within a comprehensive modelling framework. Otherwise it is likely that recommendations can be made which are based on imperfect and inaccurate representations of the actual operational environment. Fullerton and Kinnaman (1995) and Choe and Fraser (1997, 1998) are attempts in this direction. However, it should be noted that although the theoretical literature provides useful insights into the design of policy instruments and the interdependence of the various participants, in too many cases the policy recommendations that are offered are of little practical relevance. This point

[18] Halstead and Park (1996) provide an overview of issues relating to waste management in rural areas.

is borne out by the frequent policy recommendation to introduce deposit-refund schemes which in practice are subject to high transaction costs resulting from administrative complexity.

Second, the empirical literature has shown the need to carefully develop the necessary data to answer the questions of most interest. As the use of specific surveys has increased, more refined estimates of household behaviour have been produced. Interestingly, as table 1 shows, the range of price and income elasticities estimates derived is wide; the available empirical results therefore need to be treated cautiously. Thus, although the calculations of Wertz (1976) are crude, they appear to provide a good ball park estimate for the price elasticity for waste disposal serivces. Another point that needs to be emphasised is that the number of studies is small. In terms of Australia, there are at present no studies reporting price or income elasticities for the introduction of kerbside charges. Also many of the empirical papers caution against extrapolating from specific case studies (for example Lake *et al.* 1996) and that there is a need for much more applied work in this area. Thus in the context of introducing kerbside charges in Australia it is not at all clear at what level a charge should be set. However, if kerbside charges are to be introduced, it would appear to be sensible that efforts are made to establish sensible elasticity estimates. Given the developments in the empirical literature, the work of Fullerton and Kinnaman (1996) provides a useful methodology to follow. Finally, if there is to be a move towards the introduction of kerbside charges for household waste disposal services in Australia, then upstream taxes, monitoring and compliance costs for legal disposal and appropriate recycling and reuse need to be considered simultaneously.

5. Discussion and conclusion

The review of the theoretical and empirical literature almost seems to raise more questions about the issue of household waste management than are resolved. It is clear that the models, both empirical and theoretical, have become more sophisticated through time. However, the complexity has brought problems of interpretation, especially with the theoretical models. In this section some of the major issues raised by this review are discussed in terms of policy proposals and the implementation of legislation.

In the theoretical models considered here there has been a general trend towards the use of taxes in terms of consumption decisions. This has in turn yielded strong support for the introduction of deposit-refund type schemes. Although this mechanism is shown to be efficient, the practical application of this type of incentive structure is limited with numerous potential problems. For example, the South Australian container deposit-refund scheme for bottles has a negative impact on water quality. In addition the transaction costs of implementation may render the net impact of introducing a deposit-refund scheme negative.

Associated with the deposit-refund recommendation, Fullerton and Kinnaman (1995) argue that upstream taxes are appropriate. However, Choe and Fraser (1997, 1998) have derived a similar result only as a special case conditional upon the type of waste material produced by the household. For specific types of wastes such as food or many types of plastic, the deposit-refund system is both inappropriate and ineffective in giving the household incentives to undertake the effort to reduce waste. This introduces the need for user fees. On the other hand, user fees lead to the possibility of illegal waste disposal. It is thus to be borne in mind that different waste materials need to be managed by different policy instruments. In particular, neither the deposit-refund system nor the kerbside charges should be viewed as a policy with universal applicability.

The focus of analysis on the disposal decisions has in many ways obscured the important issue, that of the consumption decision. This brings the whole discussion around to the simple but powerful mantra so frequently articulated by environmentalists — reduce, reuse, recycle and dispose. There is no reason to argue against this approach to waste management. What must be guarded against, however, is an unquestioning application without taking into account the very real economic costs associated with the implementation of any policy. The particular point is well illustrated in the case of Australia in relation to recycling and landfill. Any choice of policy option will depend on the relative costs, both private and social, and if landfill is the cheaper option then there is no reason for pursuing the present recycling targets. The policy option is therefore an empirical question of costing out the mix of specific environmental objectives.

From the empirical literature the reaction of households to kerbside charges are as expected. Of the various elasticities calculated, the own price elasticity of demand for household waste services is generally very low. This is not altogether surprising as waste disposal services could be considered to be essential and resulting price effects will be small. However, these elasticities might reflect the low level of present charges. Importantly from a policy implementation perspective, they imply that unless kerbside charges are significantly higher than the present fixed payments, minimal household response can be expected. Whether the political will exists to introduce the necessary kerbside charges implied by the elasticity estimates remains to be seen. Indirectly related to the introduction of kerbside charges, the measurement of illegal dumping remains problematic, both in terms of assessing its likelihood and monitoring its impact.

From a practical point of view there are some interesting issues raised by the analysis for Australia. First, it is not sufficiently clear if the introduction of variable kerbside charges for household waste management will yield the policy results that have been claimed by some organisations (e.g. Industry Commission). To date, policy is *ad hoc* and there appears to be little chance that a more comprehensive approach will be introduced. Thus whilst policy in Australia remains an *ad hoc* affair, the likelihood that household waste management policy will approach a second-best optimum is debatable.

Second, within the literature considered here it is implicitly assumed that households live within a single jurisdiction, but in Australia, for example, this is far from the case. Policy initiatives and strategies are formulated at the Federal level, State governments then introduce legislation which in turn enables local government to formulate and implement policy. This type of government structure poses difficulties in terms of efficient implementation and the potential introduction of differences in environmental regulation to alter the prevailing economic incentives in favour of a particular area, region or state. Furthermore, although local council can via a by-law introduce kerbside charges for household waste disposal, it is clear from recent legal decisions that the relationship between the States and Federal government in relation to tax raising is unclear. Once this type of jurisdictional complexity is added into the modelling structure, it is no longer clear if the optimal economic properties presently derived will hold.

Third, and a related issue, there is a need to be aware of the difference in service provision for urban and rural households. In most of the literature considered here, the focus has been on urban household waste management. It is only very recently in the economics literature that specific attention has been paid to rural waste management. The dispersion of the rural population in Australia is self-evident and any comprehensive household waste management analysis needs to take this into consideration.

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