ACIL ALLEN CONSULTING

DEPARTMENT OF ENVIRONMENT REGULATION AND WASTE AUTHORITY OF WESTERN AUSTRALIA

MARCH 2014

ECONOMIC DRIVERS OF WASTE

ACIL ALLEN CONSULTING PTY LTD
ABN 68 102 652 148

LEVEL FIFTEEN
127 CREEK STREET
BRISBANE QLD 4000
AUSTRALIA
T+61 7 3009 8700
F+61 7 3009 8799

LEVEL TWO
33 AINSLIE PLACE
CANBERRA ACT 2600
AUSTRALIA
T+61 2 6103 8200
F+61 2 6103 8233

LEVEL NINE
60 COLLINS STREET
MELBOURNE VIC 3000
AUSTRALIA
T+61 3 8650 6000
F+61 3 9654 6363

LEVEL ONE
50 PITT STREET
SYDNEY NSW 2000
AUSTRALIA
T+61 2 8272 5100
F+61 2 9247 2455

SUITE C2 CENTA BUILDING
118 RAILWAY STREET
WEST PERTH WA 6005
AUSTRALIA
T+61 8 9449 9600
F+61 8 9322 3955

ACILALLEN.COM.AU

SUGGESTED CITATION FOR THIS REPORT

JAMES FOGARTY

CONTENTS

1 Landfill in Western Australia 1

1.1 Inert material to landfill 1

1.1.1 Levy impacts explored 3

1.1.2 Information asymmetry 4

1.2 Putrescible material to landfill 4

1.2.1 Other commercial considerations 5

1.2.2 Municipal waste flows 8

1.2.3 Impact on the relative price of different technologies 11

1.2.4 Diversion options for putrescible waste 13

1.3 Summary 13

2 Landfill price responsiveness 15

2.1 Measures of price responsiveness 15

2.2 Estimating the price elasticity 18

2.2.1 Partial adjustment model 19

2.2.2 Distributed lag model 19

2.2.3 The Autoregressive Distributed Lag model 20

2.2.4 Implications for interpretation of the literature 20

2.3 Landfill elasticity literature review 20

2.3.1 Overview of general issues 20

2.3.2 Key study summary 22

2.4 Implications for Western Australia 28

3 Externality costs 33

3.1 Carbon dioxide emissions 33

3.1.1 Discount rate assumptions 33

3.1.2 Quality of landfill site 33

3.1.3 Marginal damage cost literature 33

3.1.4 Social cost literature 34

3.1.5 Market price approach 35

3.1.6 Alternative to a market based approach 36

3.1.7 Greenhouse gas emissions externality 36

3.1.8 Existing literature and the issue of energy displacement 37

3.2 Disamenity costs 38

3.2.1 Existing information 38

3.2.2 Benefit transfer approach 39

3.3 Other externality costs 41
3.3.1 Leachate 41
3.3.2 Other gas emissions 41
3.3.3 Transport 42
3.4 Other relevant linkages 43
3.5 Summary 43

4 Case studies 47

4.1 Netherlands 47
4.1.1 Historical overview 47
4.1.2 Landfill tax details 48
4.1.3 Other legislation: landfill ban 49
4.1.4 Construction and Demolition Waste 49
4.1.5 Economic evaluation of the system 50
4.1.6 General summary and implications for Western Australia 51
4.2 Canada 51
4.2.1 Construction and Demolition in Ontario 52
4.2.2 Municipal waste in Markham City, Ontario 53
4.2.3 When landfill runs out: Toronto 53
4.2.4 General summary and implications for Western Australia 53
4.3 United Kingdom 54
4.3.1 Landfill tax 54
4.3.2 Primary Aggregates Tax 55
4.3.3 Municipal sector actions 56
4.3.4 Commercial and Industrial waste 57
4.3.5 General summary and implications for Western Australia 57
4.4 Italy 57
4.4.1 Municipal waste diversion 57
4.4.2 Capital investment and tendering 58
4.4.3 General summary and implications for Western Australia 59
4.5 Japan 59
4.5.1 Municipal waste: Pay as you throw systems 60
4.5.2 Construction and demolition waste 61
4.5.3 General summary and implications for Western Australia 62
4.6 South Australia 62
4.6.1 Kerbside recycling incentive payments 64

4.6.2 Other observations 65
4.6.3 General summary and implications for Western Australia 66
4.7 Victoria 66
4.7.1 Landfill level and recycling in Victoria 68
4.7.2 Municipal waste 71
4.7.3 General summary and implications for Western Australia 71
4.8 New South Wales 71
4.8.1 NSW Summary 71
4.8.2 General summary and implications for Western Australia 73

List of figures

Figure 1 Inert waste flows to landfill in metro WA: 1999-2012 per capita 2
Figure 2 Stylised representation of the market for inert material 3
Figure 3 Putrescible waste to landfill in metro WA: 1999-2012 per capita 4
Figure 4 PET price: Jan 2007 – Jun 2013 6
Figure 5 HDPE price: Jan 2007 – Jun 2013 6
Figure 6 Mixed paper export price: Jan 2007 – Jun 2013 7
Figure 7 Old cardboard export price: Jan 2007 – Jun 2013 7
Figure 8 MSW waste flows shares in metropolitan WA 8
Figure 9 Diversion rates associated with different technologies 9
Figure 10 Impact of the landfill levy on different technologies 12
Figure 11 Hypothetical linear demand curve for landfill 17
Figure 12 Landfill own-price elasticity density plot 27
Figure 13 Non-linear response curve: total price and diversion rate 30
Figure 14 Marginal effect of price changes in total cost of landfill 30
Figure 15 Welfare implications of externalities and taxes 45
Figure 16 Changes in the municipal waste shares in the UK 56
Figure 17 Landfill levy in South Australia 63
Figure 18 Waste diversion in South Australia 64
Figure 19 Victorian recovery rate and metropolitan landfill levy (2001-11) 68
Figure 20 Volume of C&D waste recovered (2000-11) 70
Figure 21 Sydney metropolitan area levy rate through time 72

List of tables

Table 1 Own-price elasticity for landfill literature summary 24
Table 2 Social cost of carbon (in $US1995 per metric tonne of carbon) 35
Table 3 CO2 cost for different waste streams: various benchmark (2013 dollars) 37
Table 4 Preferred CO2 externality estimates per tonne of waste (2013 dollars) 37

Table 5	Updated Productivity Commission impacts: CO2 externality	38
Table 6	Estimates of the disamenity cost of landfill in WA	40
Table 7	Landfill externality costs for Western Australia (current dollars)	44
Table 8	Construction and Demolition waste flows in the Netherlands (1990-2000)	49
Table 9	Total social cost of waste disposal: Netherlands (2000)	50
Table 10	Comparative waste performance: Canadian regions (per capita kg)	52
Table 11	UK landfill tax rates	55
Table 12	Variation in waste management performance: Italian cities	59
Table 13	Waste diversion in South Australia: by sector	63
Table 14	Organics recycling in South Australia through time	65
Table 15	Landfill levy rates: Victoria	68
Table 16	Recycling in Victoria by waste stream (2010-11)	69
Table 17	Detailed construction and demolition recovery volumes (2000-11)	70

Executive summary

Levy impact in Western Australia

As Western Australia has had a landfill levy in place for some time, and because there has been variation in the levy rate, baseline information on the effect of the levy on volumes sent to landfill is available. The levy is not the only factor that has been at work over the past 15 years, but the observed changes in volumes sent to landfill during the levy period represent a solid starting point for understanding the effect of the levy on material sent to landfill.

For inert waste, if landfill volumes for the period when the levy was $1 per cubic metre are compared to the current period, where the levy is $12 per cubic metre, the conclusion drawn is that there has been a statistically significant decrease in the annual volumes sent to landfill. Specifically, the change in the levy rate has been correlated with a reduction in annual inert material sent to landfill of 670,000 m^3. In terms of inert material it is necessary to also note that during this period there has been a significant investment in developing end markets for recycled products such as road base.

Increases in the landfill levy work to increase the total price of disposal at landfills. Effectively, such a situation increases the range of locations where it is more cost effective for material to be sent to a recycler, and decreases the range of locations where it is more cost effective to send material to landfill.

In summary, this means that the inert material market structure is such that following marginal changes in the cost of sending material to landfill there is a corresponding marginal response in the inert material diversion rate.

A final point regarding inert waste is that the levy impacts recycling operations differently. For an operation only accepting source separated material, the residual fraction that must be disposed of will be very small. For operations that collect waste that is not source separated, there will be a residual fraction of material that needs to be disposed of. The residual fraction may be between 10 percent and 20 percent, and is likely to be classified as putrescible waste. That there is a differential landfill rate between putrescible material and inert material therefore means recyclers providing a mixed stream collection service are affected by increases in putrescible landfilling costs.

Controlling for population growth, for putrescible waste, if landfill volumes for the period prior to September 2006, when the levy rate was $3 per tonne, are compared to the period after January 2010, when the levy rate was $28 per tonne, a statistically significant fall in landfill volumes can be detected. The estimated difference in material sent to landfill between the two periods is 116,000 tonnes. The reduction in landfill volumes appears to be correlated with alternative processing and market development arising from the implementation of Alternative Waste Treatment facilities.

The current market structure for putrescible waste is such that, in the short to medium term there is unlikely to be a response to relatively small changes in the landfill levy. This is due to the lack of cost effective alternatives. This situation will continue at least into the short to medium term. A possible exception to this would be if there was a change in the kerbside collection infrastructure such that existing AWT infrastructure was able to service an increased population base by moving from mixed streams to source separated streams.

Another important factor for some waste streams, especially source separated material, is the price in destination markets for items such as recycled plastics, paper, and cardboard. Price volatility in end markets is substantial, and from a business perspective, may be seen as a more important factor than the levy.

Landfill price elasticity

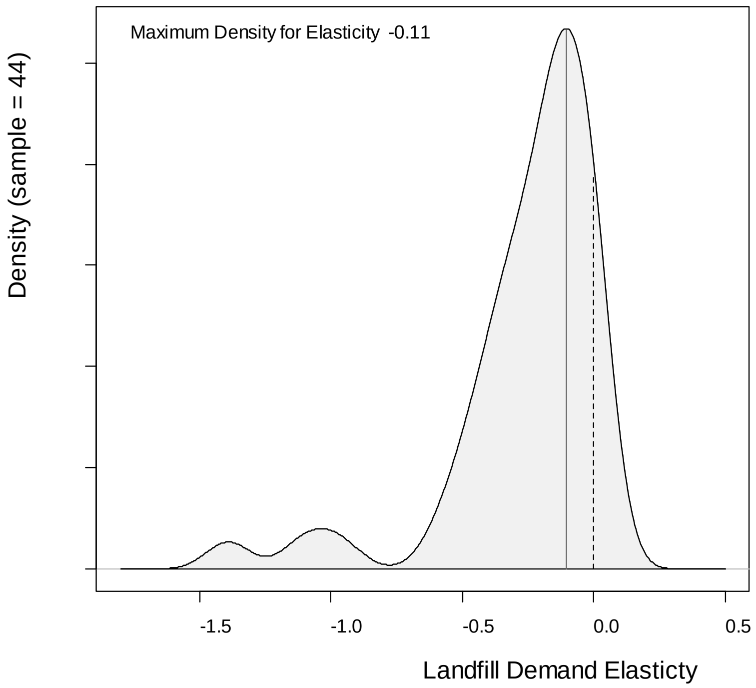
Formally, the own-price elasticity of demand for landfill is defined as the percentage change in the quantity of landfill demanded that flows from a one percent change in the price of landfill. Thus, if the own-price elasticity of demand for landfill is -0.1, this means that if the price of landfill were to increase by ten percent, the quantity demanded would decrease by one percent. The way the own-price elasticity metric can be understood in practical terms is as follows. If the own-price elasticity is less than minus one the good can be describes as relatively responsive to price changes: changes in price have a relatively large impact on quantity. If the own-price elasticity is greater than minus one the good can be described as relatively unresponsive to price changes: increases in price have a relatively small impact on quantity.

A structured literature search identified 26 studies where an own-price elasticity estimate was published, or could be calculated from the published data. These published estimates have been summarised in Figure ES 1 through the use of a density plot.

The demand for landfill literature can be summarised by the following stylised facts:

- The demand for landfill is relatively unresponsive to price changes. In practice this means that relatively large changes in gate fees are associated with relatively modest changes in the volume of material sent to landfill, especially in the short-run
- Although generally still relatively unresponsive to price changes, long-run estimates are generally more price responsive than short-run estimates. This suggests that it takes landfill operators, households, and businesses time to adjust when landfill prices change. More specifically, this says that a change in the landfill levy in 2010 has an impact in 2010, but also has an impact in 2011 and subsequent years. Failure to consider the impact of a price change over an appropriately long timeframe can result in an underestimation of the long-run price responsiveness
- The exception to landfill volumes being relatively unresponsive is observed when the increase in the total gate fee is substantial enough to make an alternative technology, such as energy from waste, clearly a cheaper technology than landfilling. When the change in gate fees is such that it does cause a new technology to become the least cost disposal technology, and when the timeframe under consideration is suitably long, landfill volumes are seen to be responsive to the price change
- When the demand for biodegradable waste is estimated separately from total waste, the biodegradable waste elasticity is more price responsive than the general waste elasticity. This in turn implies that there are more substitute options to sending biodegradable waste to landfill than there are for some other waste streams
- Although the effect in terms of diversion is still relatively low, when the price signal is clear, as it is with pay-by-weight systems, consumer responsiveness is greater
- The one published Australian specific study estimated that a 50 percent price increase would result in a one percent decrease in landfill volumes. Although it should be noted that based on the evidence in the study it is not possible to reject the proposition that price changes have no impact on landfill volumes.

Figure ES 1 **Landfill own-price elasticity density plot**



Overall, the likely response will be influenced by the scale of the price change and the time period under consideration. A small change in the dollar value of the levy will generally have little impact. For a large change in the levy, if the increase results in the total cost of disposal using an alternative technology becoming cheaper than landfill, then the long run quantity response can be substantial. The critical issue behind the long run quantity response is whether or not the price change makes the total disposal cost of an alternative technology to landfilling cheaper than landfilling. If a technology threshold is not crossed, then the long run effect will not be dramatically different to the short run effect.

Externality costs

Where there are unpriced externalities the volume of material sent to landfill is greater than the socially optimal quantity and total social welfare is not optimised. In such circumstances imposition of a tax can be used to increase total social welfare. Note, however, that if the tax rate is set above the externality correcting tax rate total social welfare is lowered in a manner similar to that of the situation of no tax.

All major studies of the externality costs associated with landfill note the existence of significant uncertainty in the calculations. The findings presented here are no different.

Estimates of the landfill externality by waste stream are shown in Table ES 1. The main externality impact of landfill is the impact of greenhouse gas emissions. At the moment these impacts are addressed through Commonwealth charges. It is however the stated intention of the current Commonwealth government to remove these charges. As such, in Table ES 1 total externality cost details are presented by core component. In the calculations the assumed externality cost of a tonne of CO_2 is \$23. Using a number of different approaches the implied externality cost of CO_2 consistently comes out at a value of around \$23 per tonne.

In summary, for well run and regulated landfills the full externality correcting tax rates are around \$10 per tonne for putrescible waste and around \$4 per tonne for inert waste.

Table ES 1 **Landfill externality costs for Western Australia (current dollars)**

Details	MSW	C&I	C&D
	$ per T of waste	$ per T of waste	$ per T of waste
Greenhouse gas emissions			
No gas capture technology	27.60	25.30	4.60
Best practice	6.90	6.30	1.20
Best practice Inc. energy displaced	2.20	2.00	0.40
Other air emissions			
No gas capture technology	0.67	0.67	0.67
Best practice	1.07	1.07	1.07
Other externalities			
Disamenity	2.66	2.66	2.66
Leachate	0.00	0.00	0.00
Transport	NA	NA	NA
Total landfill externality			
No gas capture technology	30.93	28.63	7.93
Best practice	10.63	10.06	4.88
Best practice Inc. energy displaced	5.93	5.73	4.10

The project terms of reference refer only to the externality costs of landfill, and in the published literature the research focus to date has largely been on understanding the externality costs associated with landfill. There are however a number of other issues that are relevant to a comprehensive treatment of the role played by landfill. Specifically, there are negative externality costs associated with virgin material extraction and positive externalities associated with recycling. The limited international evidence that does exist on these issues suggest that combined these effects may be greater in magnitude than the externality costs associated with landfill. To the extent that these effects are not captured through other specific measures, these externality costs and benefits are relevant to discussions of the externality correcting landfill levy.

Observations from case studies

- The waste hierarchy does not consider cost effectiveness and the use of high landfill taxes or landfill bans to achieve high diversion rates is associated with substantial welfare losses. Any situation where the landfill levy is set at a rate above the total social cost results in a lowering of total community welfare. Understanding the full impact of setting landfill tax charges above the full social cost is complicated because of the many other tax and spend interactions within an economy. Detailed general equilibrium modelling for the Netherlands that considers the full range of interactions in the economy has, however, shown that setting taxes above the level that addresses social costs results in lower consumption, and hence lower welfare. The welfare losses are also shown to increase non-linearly with further in the landfill tax
- Where there is a mis-match between policy decisions such as landfill bans and the practical realities involved in building new waste management infrastructure, undesirable outcomes can be realised. There have been examples in Europe where a landfill ban has been introduced, but sufficient time has not been allowed to build alternative infrastructure. Such scenarios then result in exporting waste to other countries, or waste being stored in an inappropriate manner
- In general, a high level of source separation is associated with higher diversion rates. The mechanisms for encouraging source separation are varied and not all approaches

rely on price as the main driver. For example, requiring compliance with a recycling plan for building and demolition work can result in very high recycling rates

- For putrescible waste, externality correcting tax rates tend to be associated with only very small changes in diversion rates. The first reason for this is that externality correcting tax rates generally reflect a modest percentage change in the total gate fee. The second reason is that externality correcting taxes are generally not sufficient to make an alternative technology to landfilling cheaper than sending waste to landfill. If, after the price change, landfilling remains the cheapest option for disposing of waste, the change in volumes landfilled will be small
- The introduction of an externality correcting aggregates tax can have a material impact on the uptake of recycled aggregate. In jurisdictions where the calculations have been made, the eternality correcting tax on virgin aggregates has been found to be quite low. However, even a small increase in the cost of virgin aggregate can represent a large change in the relative price of virgin aggregate compared to recycled aggregate. The significant change in the relative price of two goods that are highly substitutable in turn drives substantial increases in the use of recycled aggregate
- Cap and trade systems have been used to lower the total volume of material sent to landfill. More generally, cap and trade systems allow the central agency to set a maximum total amount of material that can be sent to landfill, and then allow individual market participants to work out the best way to meet the targets. Such systems have the advantage of harnessing the ingenuity of the entire community, but have the disadvantage of requiring a significant investment in institutional architecture
- In jurisdictions that have been able to implement some sort of waste stream separation at the residential level, such as a three-bin system, the amount of organic material diverted from landfill is notably higher than in jurisdictions that have collection systems that involve less source separation at the residential level. Diversion rates for such systems may be lower than some of the longer term aspirational diversion rate targets set for Western Australia, but the approach appears to be relatively low cost approach compared to other technology processing options.

1 Landfill in Western Australia

As Western Australia has experience with a landfill levy that has varied through time, the information on the way volumes have changed as the levy has changed provides valuable information. The observed changes provide a direct measure of the correlation between the landfill levy and the quantity of material disposed of at a landfill site. This measure of correlation can then be augmented by a discussion of the other factors that influence the market.

1.1 Inert material to landfill

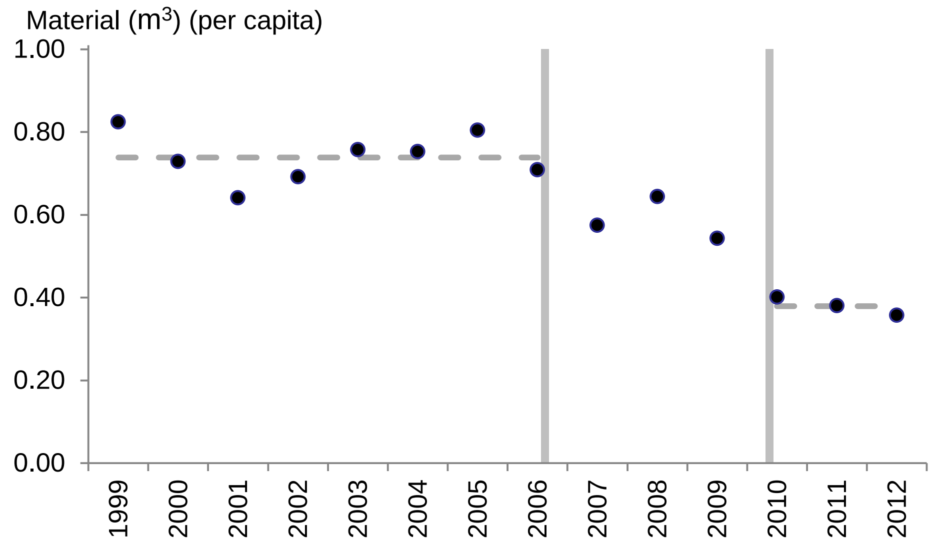
Figure 1 plots estimates of the volume of inert material sent to landfill in the metropolitan area through time, on a per capita basis. In the figure the solid black dots represent estimates of actual per capita volumes through time. Note that the individual dots represent estimates, and so there is some uncertainty surrounding these values. Additionally, we know that inert waste flows are driven by the level of economic activity and the price of virgin aggregate as well as changes in population, so individual year per capita estimates can still move around substantially.

The plot space is then divided into three periods. Period one is the period prior to September 2006 when the inert landfill levy was set at $1 per cubic metre, which for practical purposes is functionally equivalent to not imposing a levy. The middle period might be thought of as a transition period that saw the inert landfill levy increase to $3 per cubic metre. The final period in the plot space is the post-January 2010 period where the inert landfill levy was increased to $12 per cubic metre. In the figure the most instructive comparison to make is between the pre-September 2006 period and the post-January 2010 period. The mean value for these two periods is illustrated with the dashed grey line.

While noting that there are factors not considered in the comparison, Figure 1 provides a clear indication that relative to no levy, or a very small levy, an inert landfill levy of $12 per cubic metre is correlated with a meaningful fall in the per capita volume of inert material sent to landfill. If we convert back to a total implied population constant volume measure, the difference in the mean annual values across the two sample periods is 670,000 m^3, and the difference is statistically significant (p-value<.001).[1]

[1] The small sample size in the second period suggests a cautious approach to statistical testing. For example, with a small unbalanced sample the power of the test is low. Specifically, an unbalanced total sample size of 9 (6,3) the power of the test to detect a moderate effect is only 0.13. As an alternative to a classic parametric test one could rely on a non-parametric Mann-Whitney rank test. For such a test, with α=0.05 the critical U-value is 2 and the calculated U-statistic is 27, hence the null of no difference is rejected.

Figure 1 **Inert waste flows to landfill in metro WA: 1999-2012 per capita**



Source: Raw waste volume data Department or Environmental Regulation (WA).

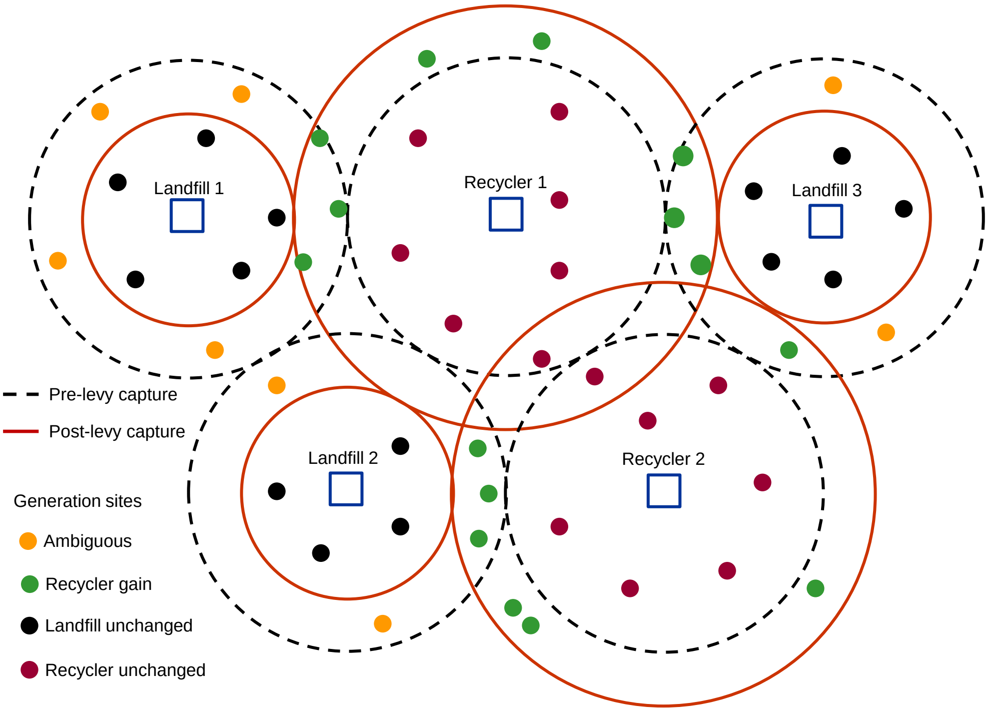
As discussed in detail in the landfill elasticity chapter, the extent of the quantity response to price changes is driven by the extent of substitute options available. That there has been a quantity response to the increase in the landfill levy suggests that for inert waste there are genuine substitute options to landfilling the material.

Across metropolitan Perth there are numerous landfill and inert material recycling sites that compete for material. In terms of disposal options, it is reasonable to suggest that many of those responsible for the generation of substantial volumes of inert waste have an agnostic view on disposal. That in turn suggests that price is a critical factor in the decision regarding where to send material. Additionally, it suggests the nature of the competition for material is both between landfill and recycling sites, and within alternative landfill and recycling sites.

Inert material recyclers operate a number of different business models, with some operators offering a skip bin service, and other facilities accepting source separated material. However, regardless of the nature of the recycling operation, two critical components in the total cost faced by users of these services are the gate fees charged and the transport cost of sending material to the facility. Increases in the landfill levy work to increase the total price of disposal at landfills. Effectively, such a situation increases the range of locations where it is more cost effective for material to be sent to a recycler, and decreases the range of locations where it is more cost effective to send material to landfill.

Stylistically, the situation is illustrated in Figure 2. In the figure the blue squares represent the various inert landfill and recycling sites located throughout the metropolitan area, and the dots represent generation sites. The black dashed line around each site represents the radius where, for that site, the total gate fee plus transport cost is lower than any alternative disposal option. When there is an increase in the landfill levy the radius around each landfill site where the landfill is clearly the lowest cost option shrinks. Conversely, the radius around each recycling site where the recycler is clearly the most cost effective disposal option increases. In the figure the impact of the increase in the levy is illustrated via the red circles. As can be seen in the stylistic representation, an increase in the levy increases the number of generation sites where it is clearly more cost effective to send material to a recycler. The volumes sent to recyclers therefore increase, and the volumes sent to landfill fall.

Figure 2 **Stylised representation of the market for inert material**



Source: ACIL Allen

In broad terms this kind of market structure suggests a relatively smooth growth path for volumes diverted as the price of disposing of material at landfill increases.

1.1.1 Levy impacts explored

The stylistic representation shown above implicitly assumes a 100 percent pass through rate of the levy increase to landfill customers. Pass through rates are a function of the interaction of supply and demand, where the supply curve is, in turn, a function of cost structure. In some instances there have also been compliance issues and disputes regarding whether the levy is payable.

More generally, as different landfill operators have different cost structures the optimal pass through rate will vary by operation. The business cost structure at a large landfill operation will typically be focused on maintaining volumes, and so the pass through rate may be 50 percent of the levy increase or less. Small operations are likely to have less flexibility, and or have lower fixed costs and higher marginal costs and so the pass through rate might be higher. Overall, as the volume of material sent to landfills falls it could be expected that the operations with the highest marginal costs would be the first sites to become uneconomic operations.

The levy will also impact recycling operations differently. For an operation only accepting source separated material, the residual fraction that must be disposed of will be very small. For operations that collect waste that is not source separated, there will be a residual fraction of material that needs to be disposed of. The residual fraction may be between 10 percent and 20 percent, and will be waste classified as putrescible. That there is a differential landfill rate between putrescible material and inert material therefore impacts recyclers providing a mixed stream collection service.

A final differential impact relates to how activities that take place on-site are treated. Currently, on-site activities involve significant volumes of material, so how they are treated has a material impact on the diversion rate calculation, and hence the assessment of the effectiveness of the levy.

1.1.2 Information asymmetry

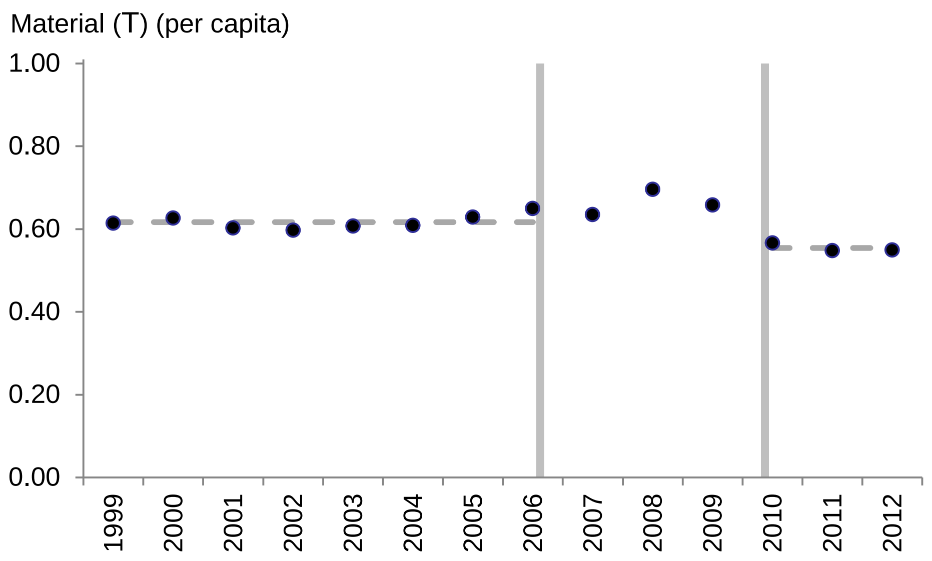
An exploratory analysis of the cost involved in disposing of demolition material at a representative site in the metropolitan area revealed that under a wide range of circumstances a recycler is the least cost option. Yet recyclers are not always the preferred receival site for the C&D material. That the least cost option is not always followed suggests there is some other factor driving decision making in this market segment. Possible issues could be a lack of information about disposal options, or some kind of inertia in the C&D market where there is a reluctance to take up new approaches. In such circumstances it may be more appropriate to focus on non-price based methods to drive change.

1.2 Putrescible material to landfill

Figure 3 has been derived in a manner similar to Figure 1; except that in Figure 3 the metric is the volume of putrescible waste sent to landfill measured in tonnes rather than cubic metres. The solid black dots represent estimates of the per capita volume of putrescible material sent to landfill; the solid grey lines divide the plot space into a pre-September 2006 period, a post-January 2010 period; and the mean per capita estimated volume of material sent to landfill during the period pre-September 2006 and the period post-January 2010 are indicated using grey, dashed lines.

During the pre-September 2006 period the putrescible landfill levy rate was $3 per tonne, and in the post-January 2010 period the putrescible landfill levy rate has been $28 per tonne. In the middle period the levy was initially $6 per tonne but had increased to $8 per tonne at the end of the period. In contrast to the inert material figure, for putrescible waste we see that an increase in the levy from $3 per tonne to $28 per tonne is correlated with a small decrease in the per capita volume of putrescible waste sent to landfill.

Figure 3 **Putrescible waste to landfill in metro WA: 1999-2012 per capita**



Source: Raw waste volume data Department or Environmental Regulation (WA).

As with the inert material flow, there may be a number of other factors impacting the volume of putrescible material sent to landfill in addition to the levy, and here such factors are not controlled for in the analysis. However, the implied population constant difference in the mean annual volume of material sent to landfill across the two periods is 116,000 tonnes, and based on a parametric t-test the difference is statistically significant (p-value < 0.001). Using the non-parametric Mann-Whitney test the difference is also statistically significant.

More importantly, the reduction in the volume of material sent to landfill, although substantially smaller than the observed reduction in inert material landfilled, is still a meaningful reduction.

In contrast to the inert material waste flow, for putrescible waste there is no matching marginal transition from landfill to alternatives following each small change in the price of landfill. Rather, changes only take place following the addition of large capital intensive facilities, such as alternative waste treatment plants. As such it can be noted that the observed reduction in material sent landfill appears to be correlated with the additional Alternative Waste Treatment facility capacity added at this time.

1.2.1 Other commercial considerations

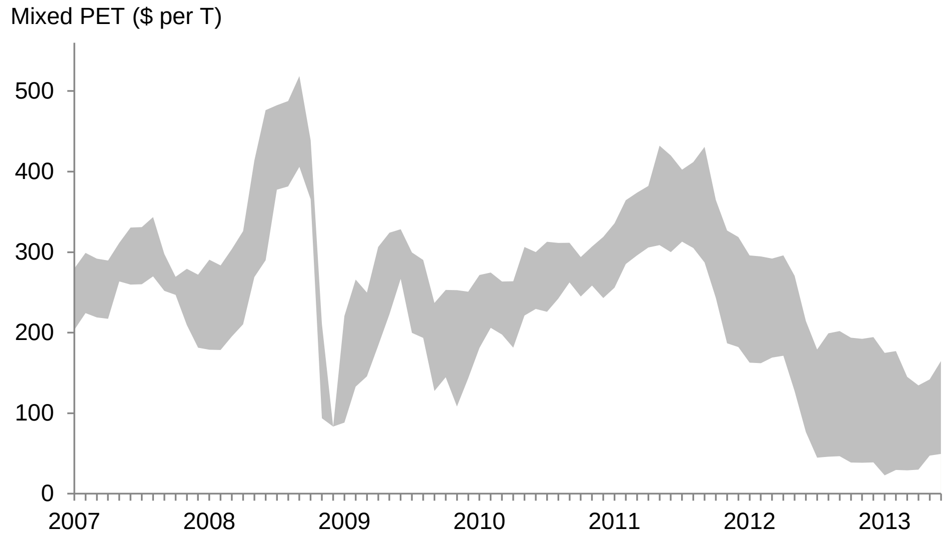
As with the inert material discussion, a key element in diverting waste flows from landfill is the availability of cost effective alternatives. If we think of the recycling function of municipal councils and the commercial waste service providers, the overall cost effectiveness of some diversion activity is related to the price received for bulk recycled material such at PET and HDPE plastics, and mixed cardboard and paper. Core market considerations are (i) the price level and (ii) the extent of price fluctuations. For example, if the price fluctuations in markets for recycled products are several times the dollar value of the levy this would indicated that the levy is of low importance relative to external market factors.

In the market for bulk recycled material there is not a single market clearing price, but rather a number of different independent transactions. In such a market individual transactions represent commercially sensitive information. Transactions for individual market participants in Australia also only take place sporadically, which in turn makes it difficult to develop a consistent time series of transaction.

The law of one price holds in markets for bulk commodity products including bulk recycled material, and there is also evidence that individual product market exhibit efficient pricing (Karikallio et al. 2011; Chen et al. 2012). This in turn means that if a price series can be derived for any of the relevant markets for products such as HDPE and mixed cardboard, that series gives a fair reflection of global prices. With this understanding that the markets for these products are co-integrated the one publicly available price series for these markets, which is based on details recorded in the UK provides a good indication of the extent of price variation for different recycled products. For each price series it is still the case that there is there is not a single unique spot price for products, but rather a range of reported sale prices. To reflect this, in each of the plots below the range of reported sale values for the month is shown. All values have been converted to the Australian dollars using the exchange rate that applied during that month

Figure 4 and Figure 5 plot the prices paid for, respectively, PET and HDPE. The core message contained in the figures is that the price for recycled plastics is quite volatile. This in turn means that although the landfill levy provides an incentive to divert material from landfill, in terms of the overall business model for those in the business of recycling, a more important factor than the levy can be what is happening to the price of bulk recycled goods.

Figure 4 **PET price: Jan 2007 – Jun 2013**



Note: Prices are in nominal terms and have been converted using the actual market exchange rate at the relevant time period.

Source: www.letsrecycle.com [accessed 28 October 2013]; www.rba.gov.au [accessed 1 December 2013]

In both Figure 4 and Figure 5 the sharp collapse in prices in late 2008 is associated with the economic shock following the collapse of the investment bank Lehman Brothers. Both the PET and HDPE markets recovered quite quickly following this shock, but while the HDPE price has subsequently remained quite high, the PET price at first recovered and has then fallen substantially.

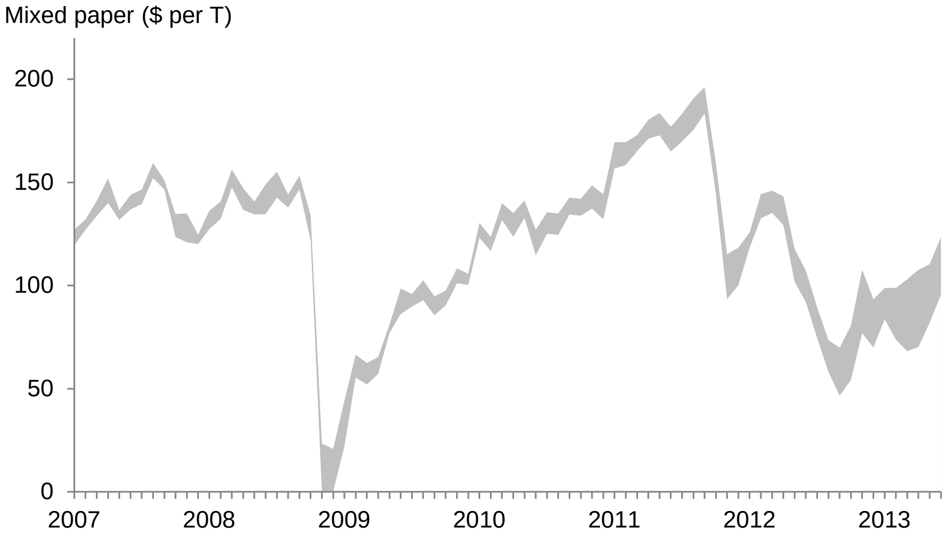
Figure 5 **HDPE price: Jan 2007 – Jun 2013**

Note: Prices are in nominal terms and have been converted using the actual market exchange rate at the relevant time period.

Source: www.letsrecycle.com [accessed 28 October 2013]; www.rba.gov.au [accessed 1 December 2013]

In the bulk recycled paper and cardboard markets (Figure 6 and Figure 7) trades generally take place in a relatively tight range. These markets were similarly affected by the Lehman Brothers induced financial crisis but the markets subsequently recovered.

Figure 6 **Mixed paper export price: Jan 2007 – Jun 2013**



Note: Prices are in nominal terms and have been converted using the actual market exchange rate at the relevant time period.

Source: www.letsrecycle.com [accessed 28 October 2013]; www.rba.gov.au [accessed 1 December 2013]

Figure 7 **Old cardboard export price: Jan 2007 – Jun 2013**

Note: Prices are in nominal terms and have been converted using the actual market exchange rate at the relevant time period.

Source: www.letsrecycle.com [accessed 28 October 2013]; www.rba.gov.au [accessed 1 December 2013]

China is Australia's main export market for recyclables and earlier this year Chinese customs officials began to enforce quality standards for material imported into the country. Given China's role as a destination country for recycled material from across the world this was a change of global significance. The UK newspaper *The Guardian* reported that as part of the enforcement of low contamination rates for material allowed into China 7,600 tonnes of recycled material was rejected (Earley 2013).

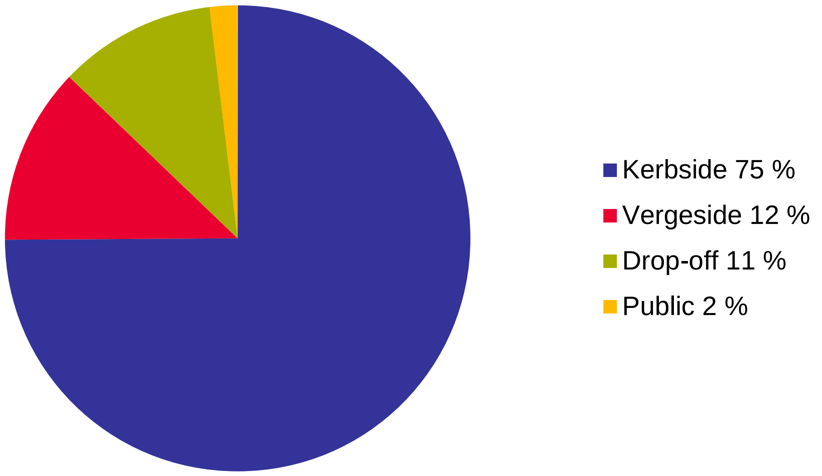
The landfill levy works to make it more attractive to divert material from landfill, but overall decisions about trying to capture recyclable material from the waste stream are also influenced by what is occurring in the global market for recycled products. For Australian operators the impact of price changes for recycled products, or even the closure of destination markets can be more important than changes to the landfill levy.

An alternative way of expressing this concept is that the levy can change the mean return to recycling, but it does not change the volatility of returns. Depending on the risk profile of operators, lowering return volatility can be a more effective mechanism than increasing mean returns.

1.2.2 Municipal waste flows

In terms of municipal waste flows, the main waste flow is the kerbside system waste flow (Figure 8). This in turn means that to change diversion rates it is the changes that take place in the kerbside system that are important.

Figure 8 **MSW waste flows shares in metropolitan WA**



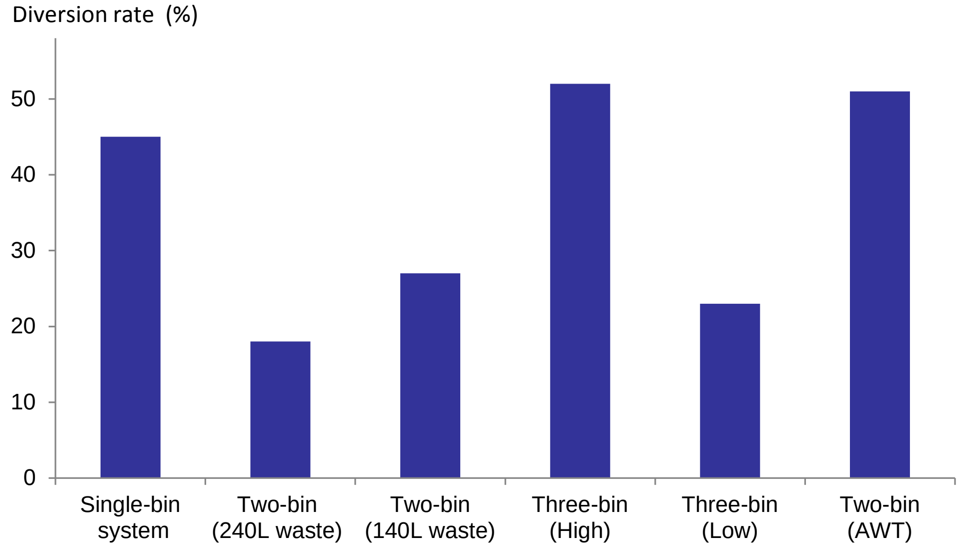
Source: Department of Environmental Regulation summary of publicly reported data

In terms of different technologies, or the substitutes to the traditional two bin-system with the residual bin sent straight to landfill, there are a number of different options. Each of the options has a different recovery rate, and each system had a different cost. If the decision framework relies on economic incentives alone, then the technology system with the least cost should be the one adopted.

Application of a least cost decision making rule

At the present time the least cost disposal system is a two-bin system, where the residual waste is sent to a landfill site. The recovery rate observed for two-bin systems varies depending on the size of the residual waste bin. Where the residual waste bin is approximately half the size of the recycling bin the diversion rates observed in metropolitan WA appears to be around 27 percent. Where both the recycling bin and the residual waste bin are both 240L bins, the diversion rate observed in metropolitan Perth appears to be around 18 percent. That the imposition of a smaller residual waste bin can lower the amount of waste sent to landfill has been noted in other Australian jurisdictions (Pickin 2008).

Figure 9 **Diversion rates associated with different technologies**



Source: Department of Environmental Regulation (WA) summary of publicly available data.

A review of gate fee information suggests that net of the levy impact, the difference in gate fees at a well run and regulated landfill and an AWT are at least $100 per tonne of waste. Given observed residual disposal rates with existing AWT technology of around 50 percent, this suggests that for an AWT based system to be least cost, a levy rate of around $200 per tonne would be required. If a least cost to resident decision making rule is used, under current conditions it makes sense to treat any investment in an AWT as a sunk cost, and switch to a two-bin system where the waste bin is sent straight to landfill. In practice councils that have taken such an approach have been able to deliver substantial cost savings to residents.

Box 1 The concept of sunk costs

Economists assume rational decision making processes and as such argue that sunk costs should not be considered when making a decision. The idea of sunk costs is best illustrated through illustrative examples.

Sunk costs for the individual

At the individual level, say you purchase a non-refundable ticket to a seminar about investing in property. Subsequent to purchasing the ticket a friend informs you that they have been to the same seminar and the speaker is dreadful, has nothing useful to say, and that everything you need to know about property investment is outlined in the latest Paul Clitheroe investment book which he just happens to have a spare copy of and so gives you for free. Having paid for the ticket you are faced with two choices:

1. Go to the seminar, waste your time, and be bored.

2. Use the time to instead engage in some other activity you enjoy.

In both cases the ticket has been paid for, but in option one you are bored, and in option two you get to do something enjoyable. Many people would feel compelled to attend the seminar even knowing that it would be boring because to not attend would be wasting money. This feeling is known as loss aversion.

Economics argues that the cost of the seminar ticket is a sunk cost that should be ignored. The real choice the individual faces is between attending the seminar and being bored or undertaking an activity they enjoy. The opportunity cost of attending the seminar is doing something enjoyable. As participating in an enjoyable activity is to be preferred to attending a seminar that will provide no new information, economics argues option two is the correct choice.

Sunk costs for a firm

Consider a firm that has spent $50M on the initial construction phase for a Waste to Energy power station. The power station is not yet operational and requires a further $10M to be spent before it becomes operational. The plant will have an estimated production life of 35 years and gate fee charges have been fixed. Now, let there be a change in policy from government such that there is no longer a Green Power specific feed in tariff. The loss of the Green Power feed in tariff makes it clear that Waste to Energy power station will be a marginal dispatch supplier of energy into the grid and that future energy generation will be gas based.

The company, having spent $50M developing the plant, has a dilemma. Having already spent money, and with the plant 80 percent complete, it would seem wasteful to not complete the plant. Yet the firm knows that the future will be a gas based energy generation dominated system where there is little role for energy sources that previously would have accessed the Green Power specific feed in tariff. Economics argues that the $50M is a sunk cost and so the real options faced by the firm are as follows:

1. Having spent $50M, spend another $10M completing the Waste to Energy power station and end up with an operational power plant that produces energy no-one wants and runs at a loss.

2. Having spent $50M, use the $10M for investment in a different product (not necessarily energy) that that can be sold profitably.

Economists argue that the correct decision to take is option two. The opportunity cost of option one is the better use of $10M.

Firms are run by people and as many people find the idea of simply walking away from a substantial investment wasteful, loss aversion is also an issue at the firm level.

Source: ACIL Allen

Three-bin systems

To the extent that the landfill levy works to increase the price differential between sending material directly to landfill and composting material, the levy works to increase the probability that three-bin kerbside systems will become the least cost option. However, the economics of such systems depend not just on the gate fee difference between landfill and compostable material, but also the proportion of contamination free green material that can be diverted from landfill.

To illustrate the issue consider the following example. Let there be two local governments that each service 28,000 households. Both LGAs operate a separate co-mingled dry recyclables bin, and the non-dry recyclables waste generated in each local council is 25,000 tonnes per year. However, the proportion of clean green waste in each LGA is substantially different. Assume that in LGA 1 the proportion of clean green waste that can be collected is 15 percent of the 25,000 tonnes, and that in LGA 2 the proportion is 45 percent. Further, assume the contamination rate in LGA 1 is ten percent, and the contamination rate in LGA 2 is one percent. Finally, assume that in both LGAs the pick-up charge for an additional bin collection is $1.50 per pick-up, and that in both LGAs residents demand a fortnightly service for the extra bin.

In this example, for both LGAs, the implied increase in annual operating cost due to the extra collection each fortnight is $780,000. Operational savings then flow from two sources: (i) the savings on levy payments for any material diverted from landfill; and (ii) the savings due to the lower cost of processing clean green material to form compost rather than depositing the material directly in landfill, which can be substantial. In practice we can combine these two sources of savings and just think of the total per tonne gate fee price differential between processing clean green waste and sending material directly to landfill.

Under the assumptions given above, and considering operating costs only, in LGA 1, for a three-bin system to be preferable on economic grounds, the difference in the per tonne gate fee between a tonne of green waste diverted from landfill and processed into organic matter and a tonne of material sent directly to landfill needs to be around $200 per tonne.

The situation for LGA 2 is however quite different. In LGA 2, for the average operating cost of a three-bin system to be lower than the average operating cost of a two-bin system, where the residual waste bin is send straight to landfill, the total gate fee price differential between a tonne of material processed to produce organic matter and landfill needs to be only around $60.

The values used in the above example are illustrative only, and there are a number of possible different configurations for a three bin system that can be considered. Given processing costs for dealing with a tonne of green waste are lower than processing costs for a tonne of general waste, the above example shows that: (i) relative to existing options for processing municipal waste in Western Australia, a three-bin system can be a relatively cost effective mechanism for increasing diversion rates; and (ii) the relative attractiveness of such a system will vary across local government areas.

Municipal system price responsiveness

The drivers of change in the municipal system are varied. Price is an important consideration, but it is not the only consideration. Community expectations for processing play a part, as do historical collaborations across LGAs. This in turn means that it is not possible to come to a clear market based characterisation of the way the municipal sector responds to price changes in the same way as other sectors. To be clear, unlike for other waste flows it is not possible to derive a general structural function that can explain the observed pattern of waste services provided in the Municipal sector.

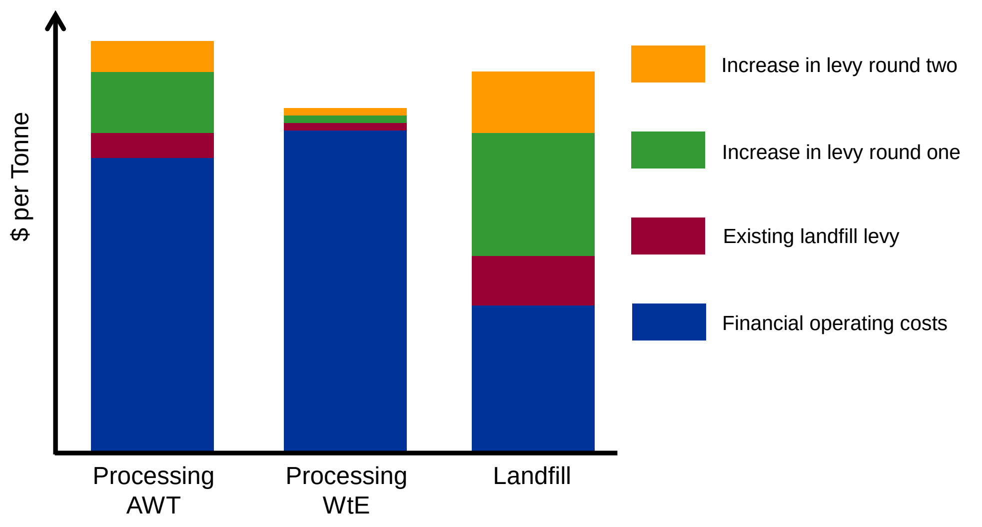
1.2.3 Impact on the relative price of different technologies

The residual fraction associated with different treatment technologies varies. As the levy is payable on the residual fraction, the levy changes the relative price of each technology option. Figure 10 provides a stylistic representation of three different waste technologies that might be considered relevant to Western Australia. The first representative technology is one of the standard Alternative Waste Treatment technologies that already exist in

Western Australia where the residual fraction sent to landfill is around 50 percent. The second technology is a medium sized Waste to Energy plant where the residual fraction that must be sent to landfill might be as low as 10 percent. The third technology is a well run and regulated landfill. The starting assumption is that pure financial costs are lowest for the landfill operation. Here, for illustration purposes, the pure financial cost of operating the AWT facility is assumed to be lower than for the WtE plant. This may or may not be correct, but the assumption aids exposition. In the figure the pure financial costs are represented by the blue section of the cost stack. The implication of the current landfill levy is then shown in red. The levy applies to all material sent to landfill, so for the landfill the full levy amount is added to the per tonne processing cost. For the AWT, as the residual sent to landfill is 50 percent, operating costs per tonne increase by 50 percent of the levy amount. For the WtE plant, as the residual fraction sent to landfill is only ten percent, the levy has little impact on per tonne operating costs. Although the figure is illustrative only, the representation of costs is consistent with the current environment: with the levy set at the current level landfilling remains the most cost effective disposal option.

The effect of future increases in the levy on each technology is then illustrated. The hypothetical levy increase represented by the green block represents an increase in the levy that is exactly equal to the per tonne difference between landfilling and the cheapest alternative waste processing option, which in this example is the AWT plant. However, because of the residual fraction sent to landfill, following the levy increase direct landfilling remains the most cost effective option. The final hypothetical levy increase considered is shown in orange, and following this increase, due to the low residual fraction sent to landfill, WtE becomes the most cost effective disposal option.

Figure 10 **Impact of the landfill levy on different technologies**



Source: ACIL Allen

The above example considers the variable impacts of the levy due to variation in the residual fraction sent to landfill after processing. Similar issues also arise regarding the need for cover material on-site and other civil works at some sites, where specific requirements can vary substantially across sites, and also vary substantially from the default allowance. The detail in Western Australian Advisory Council on Waste Management (1997) indicates that it was originally envisaged that the administration of the levy would be flexible enough so that the specific volumes required for cover material and on-site construction would not be subject to the levy. In practice, however, a balance has to be struck between efficiency and administrative complexity so that such total flexibility is not always possible. Within the landfill industry there are, however, examples where a combination of systems are used for calculating a specific charge. Typically these approaches involve a default simple/ automatic procedure that can be applied, but also allow operators, at their own expense to present detailed site specific calculations.

1.2.4 Diversion options for putrescible waste

For inert material the change in the levy works to increase the relative attractiveness of recycling, and decrease the attractiveness of direct landfilling. Volumes sent to landfill therefore fall. For MSW and other putrescible waste the situation is somewhat different. These differences mean that changes in the levy are likely to have less impact on waste volumes diverted.

First, for waste to be diverted there has to be an alternative destination that has the capacity to receive and process the material. Currently there is not substantial free capacity at licenced alternatives to landfill and adding capacity takes substantial time. Although, if there were changes to the waste collection system, in terms of the introduction of a three bin collection system, then it would be possible to achieve substantial additional diversion without the introduction of new processing plants.

Second, the alternatives to landfilling need to be cost competitive to be a compelling alternative. The current alternatives to landfilling are not cost competitive in most situations, and the difference in costs across technologies relative to landfilling are substantial (Hyder 2013).

1.3 Summary

The increase in the landfill levy on inert material to $12 per cubic metre has been associated with a substantial (and statistically significant) decrease in the flow of inert material to landfill.

There are substitute options for inert material and so for marginal changes in the inert material levy increase there is a marginal response in terms of diversion. A differential levy for inert and putrescible material impacts the different types of inert recyclers differently, as different approaches have differing amounts of residual waste.

In the inert material market it is not clear that when a recycler is the least cost option for demolition waste that a recycler is the receival site for material. This in turn suggest that non-price based interventions may be required in this market.

The increase in the putrescible landfill levy to $28 per tonne has been associated with a modest, though statistically significant decrease in the volume of material sent to landfill.

Unlike the inert material market for much of the material in the putrescible market there are not substitute options at the margin. This means that for much of the material there is very little change in disposal to landfill following marginal changes in the cost of landfilling material.

For putrescible material where there are alternative markets, price volatility in these markets is significantly greater than the levy. This in turn means that the dominant driver of change in these markets relates to external market developments and not changes in the levy.

At the moment the alternatives to direct landfilling of putrescible waste are generally significantly more expensive than the direct landfilling of the waste. If decision making regarding waste management was made on pure cost effectiveness criteria, sunk costs would be ignored, there would be a reversion to two-bin systems, and there would be a substantial increase in landfilling activity.

Because of the high residual component associated with technologies currently in use, very substantial changes in the levy would be required to make alternatives to landfilling cost competitive. Additionally, due to differences in the residual component associated with different technologies the size of the change in the levy that would make current technologies cost competitive would possibly make technologies currently not in use in Western Australia cost competitive (for example waste to energy).

References

Chen, R., Wang, W., & Li, W. (2012, August). Efficiency Test in Chinese Plastic Futures Market. In *Business Intelligence and Financial Engineering (BIFE), 2012 Fifth International Conference on* (pp. 404-407). IEEE

Karikallio, H., Mäki-Fränti, P., & Suhonen, N. (2011). Competition in the global pulp and paper industries–An evaluation based on three approaches. *Journal of Forest Economics*, *17*(1), 91-104.

Earley K, 2013, Could China's 'green fence' prompt a global recycling innovation?' The Guardian, 27 August 2013, available www.theguardian.com [accessed 30 October 2013]

Pickin, J. (2008). Unit pricing of household garbage in Melbourne: improving welfare, reducing garbage, or neither?. *Waste Management & Research*, *26*(6), 508-514.

Western Australian Advisory Council on Waste Management (1997). The Introduction of an urban landfill disposal levy in Western Australia: A discussion paper for distribution to interested parties.

2 Landfill price responsiveness

This chapter discusses the quantity response of the volume of material sent to landfill when the price of landfill changes. First the way price responsiveness is measured is discussed, as are the methods for obtaining estimates of price responsiveness. Some of this material is technical, but it lays the foundation for subsequent discussion, especially in relation to material in the case study chapters. The findings of a comprehensive literature search are then presented. The evidence from the literature search indicates that landfill volumes generally exhibit low price responsiveness. The chapter concludes with summary comments that relate the content and findings of the chapter to Western Australia.

2.1 Measures of price responsiveness

The change in quantity following a change in price can be reported in a number of different ways. In the context of measuring the change in the quantity of material landfilled following a change in the price of landfill (due to a change in the landfill levy or some other factor) one approach is to report the per capita kilogram (or tonne) reduction in material landfilled for every one dollar increase in price. Within an individual country interpreting such information is straightforward, but the value of this approach is limited unless the relative importance of a one dollar change is made clear. For example, if gate fees exclusive of the landfill levy are $5 per tonne, a $1 per tonne change in price due to the introduction of a landfill levy represents a 20 percent increase in price, which is substantial. If, however, gate fees exclusive of the landfill levy are $100 per tonne, then a $1 change in price due to the introduction of a landfill levy represents a one percent increase in price, which is not a significant price change.

For cross-country comparisons where gate fees vary widely it is therefore difficult to obtain a clear picture of landfill price responsiveness from measures of the change in per capita volumes sent to landfill per dollar change in the landfill levy. As the appropriate exchange rate to use for converting values to a common currency is not universally agreed, cross-country comparisons are further complicated by the use of different exchange rates.

Due to such issues it has become common in the economics literature to report price responsiveness using the elasticity metric, which is a unit free measure. In addition to being a unit free measure, price elasticities, which in general terms measure the change in quantity when price changes, can be derived from the generalised demand equations of consumer theory (Gravelle and Rees 1992). Consumer theory is well developed, and as such, consumer theory can be used to gain an *a priori* understanding of the factors that impact the landfill own-price elasticity.

Formally, the own-price elasticity of demand for landfill is defined as the percentage change in the quantity of landfill demanded that flows from a one percent change in the price of landfill. Thus, if the own-price elasticity of demand for landfill is minus 0.1, this means that if the price of landfill were to increase by ten percent, the quantity demanded would decrease by one percent. If Q_L is used to denote the volume of material sent to landfill, and P_L is used to denote the price of material sent to landfill, then the landfill own-price elasticity formula at a specific point can be given as:

$$\eta_{Q_L,P_L} = \frac{\partial Q_L / Q_L}{\partial P_L / P_L} = \frac{\partial Q_L}{Q_L} \times \frac{P_L}{\partial P_L} = \frac{\partial log Q_L}{\partial log P_L} = \frac{\text{percentage change in } Q_L}{\text{percentage change in } P_L}. \quad (1)$$

It is because the expression is the ratio of two percentage changes that the problem of the relative importance of the price change is avoided. In terms of interpretation, if the own-price elasticity of landfill is less than minus one, the demand for landfill is said to be price elastic. A value of less than minus one would mean that the landfill volume is relatively sensitive to price changes. If the own-price elasticity of landfill is greater than minus one, the demand for landfill is said to be price inelastic, meaning that the landfill volume is not sensitive to price changes. By the law of demand there is a non-positive relationship between price and quantity and so the own-price elasticity values must be non-positive, i.e. lie between zero and minus infinity.

The cross-price elasticity of a good measures the percentage change in the quantity of a good -- say recycled waste -- demanded as a result of a one percent change in the price of a different but related good, say the price of landfill. The key difference between an own-price elasticity measure and a cross-price elasticity measure is that for the own-price elasticity measure the price and quantity relate to the same good (eg the price of landfill and the quantity of landfill) whereas for the cross-price elasticity the price and quantity relate to different things (eg the price of landfill and the quantity of recycled waste).

If the cross-price elasticity of demand for landfill and recycled waste is 0.05, it implies that if the price of landfill were to increase by ten percent, the quantity of waste recycled would increase by half of one percent. Where the cross-price elasticity is positive, the goods are referred to as substitutes. Where the cross-price elasticity is negative, the goods are referred to as complements. The formal result for the cross-price elasticity of demand between landfill and recycled waste, where Q_R denotes the volume of material recycled, and P_L is the price of landfill is given as:

$$\eta_{Q_R,P_L} = \frac{\partial Q_R / Q_R}{\partial P_L / P_L} = \frac{\partial Q_R}{Q_R} \times \frac{P_L}{\partial P_L} = \frac{\partial log Q_R}{\partial log P_L} = \frac{\text{percentage change in } Q_R}{\text{percentage change in } P_L}. \quad (2)$$

The fundamental economic theorem of demand homogeneity requires that the (Hicksian) own-price elasticity of a good, plus all the relevant cross-price elasticities, must sum to zero. Using the above notation this theoretical requirement can be expressed as:

$$\sum_j \eta_{Q_i,P_j} = 0, \qquad j = (1, \ldots, n). \quad (3)$$

Demand homogeneity therefore tells us that own-price elasticity of landfill (or any other good) is determined by: (i) the number of substitutes, and (ii) the extent to which products are substitutable.

This insight about the determinates of the own-price elasticity is important, and has the following implications:

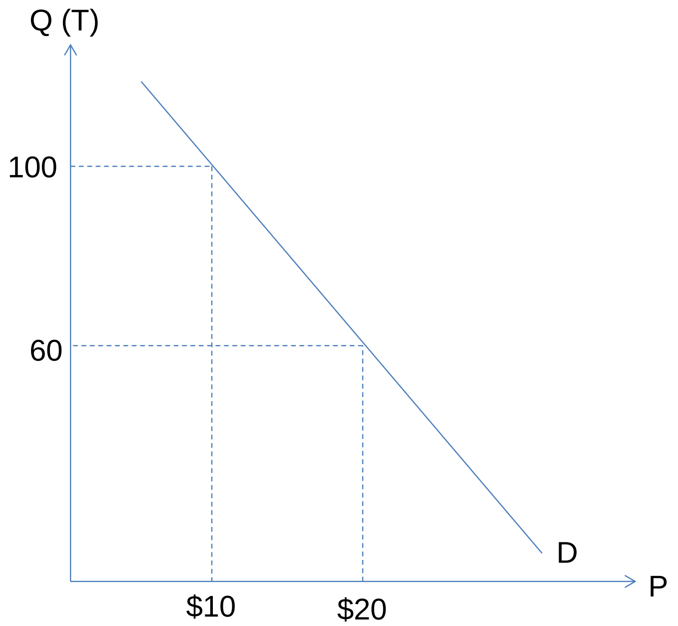
- if the alternatives to sending waste to landfill are limited, price changes through raising the landfill levy will not drive changes in the volume of material sent to landfill
- if alternatives to sending waste to landfill are introduced, the price elasticity will change. That means that non-price related policies that increase the number of substitutes to landfilling waste will impact price responsiveness

- if, for some waste streams, or sub-waste streams such as biodegradable municipal waste, there are more alternative options to landfilling, then the responsiveness of these waste flows to price changes will be greater
- as the scale of the price increase considered increases, the alternatives to landfilling that are economically viable will increase. As such, price responsiveness could be very low for modest price changes, but become higher as each new substitute technology crosses the threshold of economic viability. This means that the price elasticity may not be constant
- it can take time to understand the alternative options available following a price change, and it can take time to build the infrastructure required for substitute options to landfilling. The immediate, or short-run own-price elasticity is therefore likely to be more inelastic (less price responsive) than the long-run own-price elasticity.

Although elasticity measures are unit free measures, depending on the specific way that a researcher decides to estimate the price-quantity relationship for landfill there can still be some issues with interpretation. As can be seen from equation (1), the percentage change in the quantity of waste sent to landfill can be expressed as the change in the natural logarithm of the quantity of material sent to landfill; or as the change in the volume of material sent to landfill, measured in the original scale, divided by the level of material sent to landfill. It is from this second expression for percentage changes that a potential issue arises.

To illustrate the issue consider the following linear demand curve, which shows a hypothetical relationship between the quantity of material sent to landfill and the price of landfill, where we consider a change in the landfill price of $10 per tonne.

Figure 11   **Hypothetical linear demand curve for landfill**



Source: ACIL Allen

For the hypothetical linear demand curve shown in Figure 11, if the starting gate price for landfill waste was $10 per tonne, and the gate fee rose to $20 per tonne due to a landfill tax, then the quantity of waste sent to landfill would fall by 40 tonnes. With a linear demand curve, and a non-infinitesimal change, if we calculate the own-price elasticity using the original price and quantity as the divisor in the percentage change formula we get:

$$\eta_{Q_L,P_L} = \frac{\frac{\Delta Q_L}{Q_L}}{\frac{\Delta P_L}{P_L}} = \frac{\frac{-40}{100}}{\frac{10}{10}} = -0.4. \tag{4}$$

Yet, if we use the post price change quantity and price as the divisor in the percentage change formula we get:

$$\eta_{Q_L,P_L} = \frac{\frac{\Delta Q_L}{Q_L}}{\frac{\Delta P_L}{P_L}} = \frac{\frac{-40}{60}}{\frac{10}{20}} = -1.33. \tag{5}$$

The important point to note about the above example is that with one set of calculations landfill demand would be defined as exhibiting high price responsiveness, and with the other set of equally valid calculations landfill demand would be defined exhibiting low price responsiveness. Based on one set of calculations the conclusion drawn would be that changes in the price of landfill are an effective means of achieving changes in the volume of material sent to landfill; based on the other set of calculations we would conclude that price changes are an ineffective means of achieving changes in landfill volumes. So we can reach different conclusions in terms of the effectiveness of different policies based on the same observed data.

In such circumstances it is recommended that the arc elasticity of demand measure proposed by Allen and Lerner (1934) be used. The arc elasticity of demand can be calculated as:

$$\eta_{Q_L,P_L} = \frac{\frac{\Delta Q_L}{(Q_{L_1} + Q_{L_2})/2}}{\frac{\Delta P_L}{(P_{L_1} + P_{L_2})/2}} = \frac{\frac{-40}{(100+60)/2}}{\frac{10}{(10+20)/2}} = \frac{\frac{-40}{80}}{\frac{10}{15}} - 0.75. \tag{6}$$

As such, for the given example the demand for landfill would be classified exhibiting low price responsiveness.

Although there exists a standard approach to calculating the own-price elasticity when large changes in price are considered, it remains true that with a linear demand curve that has a constant slope, the own-price elasticity varies along the demand curve. Hence the point along the demand curve where the evaluation takes place impacts the reported elasticity value.

2.2 Estimating the price elasticity

In the peer reviewed academic literature a number of different approaches have been used to estimate the landfill elasticity. Some approaches are relatively simple, using average price and quantity information before and after the introduction of a landfill tax or other charging system. Other approaches compare average waste per capita in local government locations that have a direct and transparent waste price charging system to local government locations that do not have a direct waste price charging system. There are also a number of studies that use time series data or a combination of time series and cross-section data to estimate the landfill own-price elasticity.

The most flexible models used in the literature to estimate the own-price elasticity of landfill appear to be models that are described as Autoregressive Distributed Lag (ADL) models. These models require time series data and can estimate both short-run and long-run own-price elasticity values. Originally developed in the 1960s, interest in ADL type models was revived in the 1990s as techniques emerged to estimate long run relationships with non-stationary time series data.

The ADL model can be difficult to interpret, so it is worth explaining the approach as a combination of two underlying sub-models that have a more natural interpretation. Here the sub-models are explained in words, but a formal explanation is also provided in the appendix.

2.2.1 Partial adjustment model

The first sub-model to consider is the partial adjustment model, and in the context of estimating the landfill own-price elasticity the model can be understood as follows.

Consider a firm generating significant C&D waste. When the price of landfill changes it is unlikely the firm can adjust instantaneously. For example, if, in response to a much higher landfill charge the firm wanted to decrease the amount of waste it sent to landfill substantially, it would not be able to make a complete adjustment quickly. It takes time to implement new practices on-site that will allow for waste separation and diversion. It takes time to adjust material ordering practices to minimise the level of on-site waste. It takes time to find an alternative facility to accept the waste for recycling. People have to be trained in new site practices, etc.

The aggregate effect of these practical limitations to business operations is that in response to a price increase for landfill in the current time period there will be only a partial adjustment of waste volumes towards the intended target level in the current time period. With this characterisation of the market there is then a very real problem in assessing the impact of price based policies such as changes in the landfill levy. What is required for policy development and evaluation is a measure of the long-run volume response; what is observed every period is the short-run volume response. As explained in the appendix, the partial adjustment framework provides a mechanism that allows us to retrieve both the long-run and short-run effects from actual data observations.

2.2.2 Distributed lag model

The second sub-model to consider is the distributed lag model. The distributed lag model says that there can be a delay in responses to a change in operating environment today. In the context of landfill price responsiveness, the model says what happen today, in terms of price changes, matters, but because we have a slow moving process, what happened in the past also matters for what happens today. The motivation is similar to that outlined for the partial adjustment model. If the landfill levy is increased in 2010, that increase is important for what happens in 2010, but the increase is also important for what happens in 2011, and subsequent years.

To see why this is the case, consider an example where the landfill levy increases by $20 per tonne in 2010. In the immediate period the increase in the levy might encourage some people to look at composting waste etc., so there is a landfill quantity response in 2010 from a levy increase in 2010. The increase may however also encourage someone to start building a new material recovery facility in 2010, where the facility only becomes operational in 2011. The price change in 2010 therefore has a modest immediate period effect on the quantity of material sent to landfill, and a second, additional effect on the quantity sent to landfill in 2011. As with the partial adjustment model, the distributed lag model allows both long-run and short-run effects to be derived from the observable data. The specific process for achieving this is described in the appendix.

2.2.3 The Autoregressive Distributed Lag model

With an understanding of the relevant sub-models, and the motivation for the way these models are estimated, it become possible to consider the ADL model. There are a number of forms of ADL model but the form used in the landfill demand literature is known and an ADL(1,1) model. The model is very flexible and essentially incorporates both the partial adjustment model and the distributed lag model within the one overarching framework. In effect this means that should there be some form of adjustment process that means there are both long run and short run effects the process will be detected. The specific way that the partial adjustment model and the distributed lag model are combined into the ADL(1,1) is described in the appendix.

2.2.4 Implications for interpretation of the literature

The core implication from the discussion of estimation frameworks presented is the need to be aware of the difference between the short-run and long-run impact of a price change. Failure to consider the impact of a price change over an appropriately long timeframe can result in an underestimation of the true price responsiveness.

2.3 Landfill elasticity literature review

2.3.1 Overview of general issues

There are some existing survey reviews of the landfill own-price elasticity literature. OECD (2004) suggests that landfill own-price elasticity estimates cluster around -0.2, and Bartelings et al. (2005) suggests that landfill own-price elasticity estimates generally fall between -0.1 and -0.5. Both existing surveys therefore suggest that the demand for landfill is price elastic. However, in OECD (2004) the exact papers reviewed are not made clear; and in the case of Bartelings et al. (2005), although a number of papers are reviewed, explicit own-price elasticty values are reported only for five studies. Additionally, the most recent study reviwed in Bartelings et al. (2005) was published in 2000. This suggests that despite widespread interest in the landfill own-price elastity, there is no up-to-date systematic review of studies that has estimated an average own-price elasticty effect size.

For the current review, relevant studies were identified using several steps. First, the Google Scholar search engine was used to identify relevant papers. For this review, as papers on waste have been published in both Science journals and Economics journals, the Google Scholar search engine, which indexes both science and social science journals, is an appropriate search engine to use for paper identification. Second, the references in each paper identified as relevant were used to identify other relevant studies. Third, specific key word searches were conducted across all articles in journals such as *Waste Management* and *Journal of Environmental Management* that were identified as journals publishing regular articles on waste management.

The focus of the review was largely peer reviewed literature, books from publishers with a track record of publishing academic work, and the publications of respected NGO organisations such as the Organisation for Economic Co-operation and Development (OECD), and the National Bureau of Economic Research (NBER).

The way the landfill elasticity is measured varies across studies. Some studies consider total waste, some consider only biodegradable waste, some consider only commercial and industrial waste, etc. Additionally, there is variation across studies in how direct the price mechanism considered is. Some studies consider the effect of changes in the landfill tax, and do not consider how direct the price signal is to the generator of waste. Other studies derive estimates of the own-price elasticity following the introduction of block pricing schemes; a system where it could be argued the price signal is muted. Finally, some studies use the introduction of direct pay-by-weight systems to estimate the own-price elasticity.

Even across studies that consider the same policy change, and the same waste stream, the quality of the methodology used to estimate the own-price elasticity of demand is variable. Additionally, the size of the data set used to estimate responses is highly variable. For example, Sakai, et al. (2008) use a cross-section sample of 39 observations to estimate landfill price responsiveness in Japan; whereas Linderhof et al. (2001) use observations on 3,459 individual households in the Netherlands over the period 1993-96 and rely on over 120,000 data points for estimation of the landfill elasticity.

As a general principle it can be argued that as estimate precision is related to sample size, the own-price elasticity estimates from studies with larger sample sizes should be given more weight than estimates from studies that have smaller sample sizes (Hedges and Olkin 1985). Although, if publication bias is not an issue, a simple average of estimates, combined with appropriate qualitative commentary, will provide useful insights regarding the underlying effect size.

Further issues arise in summarising the literature as some authors have incorrectly interpreted their results. For example, Nestor and Podolsky (1996) show that the Strathman, Rufolo and Mildner (1995) estimate of the own-price elasticity of landfill of -0.46 for the US is an incorrect interpretation of the data. Specifically, Nestor and Podolsky show that the correct interpretation of the data used in the study implies an own-price elasticity of landfill of -0.11 rather than -0.46. As shown in Kinnaman (2008), another example of erroneous interpretations of waste data is Dijkgraaf and Gradus (2008).

In a number of studies, for example, Dijkgraaf and Gradus (2004), it is possible that there is a conflation of policy effects that leads to an overestimate of the price responsiveness. To understand this issue consider the following example. A local government introduces, at the same time, a pay-as-you-throw municipal waste system, a new recycling collection service, and also makes available subsidies for residents to purchase home composting bins. All three initiatives will work to reduce waste to landfill. However, unless the model estimated explicitly controls for the introduction of the recycling service and the composting initiative, the effects of these programs in reducing waste will be incorrectly attributed to the pay-as-you-throw waste system. Formally the issue is known as the omitted variables problem, and it results in estimates of the landfill elasticity being biased and inconsistent. The conflation of effects is a real issue. For example, Miranda et al. (1994) reviews the impact of the introduction of pricing for municipal waste in 21 US cities and finds that in the one location that did not concurrently implement a recycling program, unit based pricing had no effect on the flow of waste to landfill.

As, when a landfill levy is introduced, there is often a policy of recycling the revenue raised to support diversion activities, the introduction of a landfill levy can be a trigger for the establishment of recycling and other diversion programs (Ventosa, Martínez and Sora 2012). Conflation of effects is also an issue in cases where an elasticity estimate has been calculated from available summary data only. Studies where conflation is an issue are a problem in terms of identifying the landfill elasticity effect, but have a possible interpretation as the total effect considering price changes and the average effect of other typical policy responses. In this sense, such measures are still useful to consider.

A final technical issue that until very recently has not been addressed in the models estimated is endogeneity. Endogeneity in the price variable could be an issue for many reasons, but two plausible reasons are: (i) communities that have greater environmental awareness are more likely to introduce a unit based pricing scheme, and (ii) the introduction of a block pricing scheme where price and quantity are jointly determined. The evidence that is available suggests endogenity is an issue for studies of the way landfill volumes respond to price changes. However, the impact of the failure to address the issue may not have introduced any systematic bias into the landfill own-price elasticity estimate literature. This is because while Kinnaman and Fullerton (2000) find that failure to address the endogeneity issue results in estimates of the own-price elasticity of landfill that are too close to zero, Allers and Hoeben (2010) find the opposite effect.

Although the list of study limitations seems substantial, if study limitations are treated as random errors in approach, there may not be any systematic bias in the average of the published own-price elasticity estimates. In a study where block blocking is used rather than a pure direct pay-by-volume system, we may get an underestimate of the true landfill elasticity. Where conflation with recycling and other initiatives is an issue we may get an overestimate of the true landfill elasticity. For failure to treat endogenity the evidence suggests sometimes will get an overstatement of the true landfill elasticity and sometimes we will get an understatement of the true landfill elasticity. As long as the errors resulting from methodological shortcomings in studies are approximately random, the mean effect from a direct average of the published estimates will be a reasonable estimate of the true average effect.

In this instance, with the limitations of different studies noted, a direct summary of the literature provides clear insights. This is because despite any issues with the estimation approach of studies, the published estimates cluster around a highly inelastic average effect. Where the own-price elasticity is close to zero, price based policies such as changes to the landfill levy are not effective drivers of change.

2.3.2 Key study summary

A summary of the studies that reported an own-price elasticity estimate for landfill is provided in Table 1. As the critical value for considering elasticity estimates is minus one, the elasticity estimate summary table reports estimates to two decimal places only.

Reviewing the summary information presented in Table 1 suggests the following stylised facts:

1. The demand for landfill changes little when there is a change in price. This in turn suggests that marginal increases in landfill gate prices through increases in landfill taxes will have little impact on waste flows to landfill – at least until a threshold is reached
2. Long-run estimates are generally more price responsive than short-run estimates. This suggests that it takes landfill operators, households, and businesses time to adjust when landfill price changes. Where the price change is a marginal price change the long run response is, however, still generally modest. Long run responses are therefore still generally classified as consistent with low price responsiveness
3. When the demand for biodegradable waste is estimated separately from total waste, the biodegradable waste elasticity is more price responsive than the general waste elasticity. This in turn implies that there are more substitute options to sending biodegradable waste to landfill than there are for other waste streams
4. Most studies have been concerned with municipal waste flows
5. Despite the EU directive on sending waste to landfill, the US is the most studied market
6. Where the price signal is clear, such as with pay-by-weight systems, the own-price elasticity estimate is further from zero

7. For the one Australian study it was estimated that a 50 percent price increase would result in a one percent decrease in landfill volumes. Although the study was also not able to reject the null hypothesis that price had no impact on the volume of material sent to landfill.

Table 1 **Own-price elasticity for landfill literature summary**

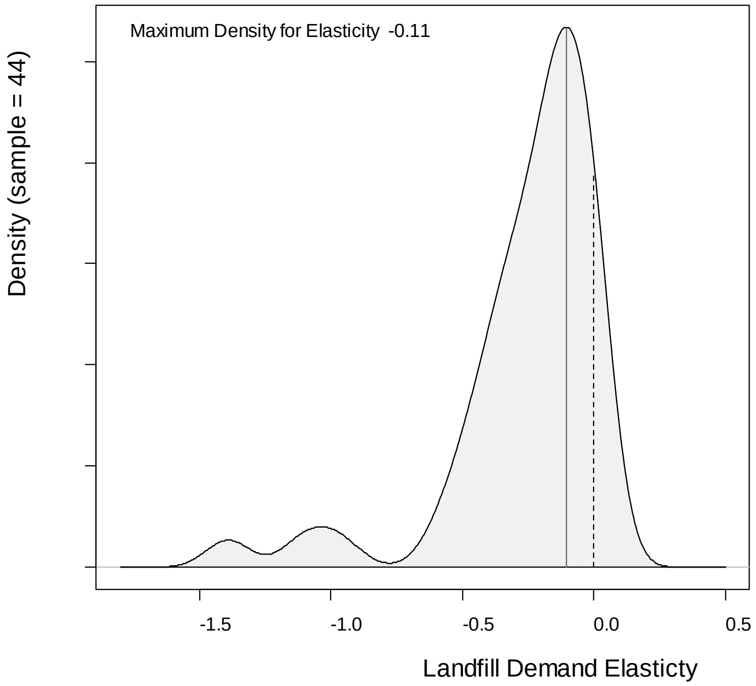
No.	Paper	Country	Waste	Period	Elasticity	Obs.	Pricing	Study type	Comments
1	Allers & Hoeben (2010)	Netherlands	Municipal	1998-2006	-0.11	3,605	Bag/Tag	Panel fixed effects difference in differences	Endogeneity issue treated. Effects can be consider LR effects. OLS overstates responsiveness. B = Biodegradable only, and is more responsive.
		Netherlands	Municipal	1998-2006	-0.37 (B)	3,605	Bag/Tag		
2	ACIL Allen (2013)	UK	Municipal	2005-12	-0.97 (B)	2	Landfill tax	Estimate inferred from LATS data and statement on tax effectiveness	Fails to subtract recycling effect, but deals with a change from £18 to £56, that favours incineration. LR effect. B = Biodegradable only.
	Bassi and Watkins (2012)	Denmark	Municipal	1995-2009	0.00	15	Landfill tax	Extended ADL model	Some effects are positive but no SR or LR effects are statistically significant. Issue is lack of degrees of freedom in each country equation. Model is in levels, but as effects are not different from zero, so is the elasticity.
		Finland	Municipal	1995-2009	0.00	15	Landfill tax		
		Netherlands	Municipal	1995-2009	0.00	15	Landfill tax		
		Sweden	Municipal	1995-2009	0.00	15	Landfill tax		
		UK	Municipal	1995-2009	0.00	15	Landfill tax		
3	Callan and Thomas (2006)	US	Municipal	2000	-0.58	351	Bag/Tag	351 cities 3SLS estimates	Response includes a pure price response of -0.195 and an induced recycling response of -0.387.
4	Dijkgraaf & Gradus (2004)	Netherlands	Municipal	1998-00	0.00	1,323	Block	Panel model with 538 individual LGAs. Fixed effects estimation.	As the pricing system becomes more transparent to the user, the elasticity estimate is further from zero, but still inelastic. Bag/Tag and weight system are not statistically different from each other.
		Netherlands	Municipal	1998-00	-0.16	1,323	Frequency		
		Netherlands	Municipal	1998-00	-0.36	1,323	Bag/Tag		
		Netherlands	Municipal	1998-00	-0.40	1,323	Weight		
5	Fullerton & Kinnaman (1996)	US	Municipal	1992	-0.06	300	Bag/Tag	Before and after effect for 75 individual households	Emphasised that illegal dumping is a serious issue. Short run effect.
6	Gellynck & Verhelst (2007)	Belgium	Municipal	2003	-0.14	295	Bag/Tag	295 LGAs cross section	Some of the interpretation in the paper of coefficients is difficult to follow.
7	Hong (1999)	Finland	Municipal	1995	-0.15	3,017	Bag/Tag	3,017 individual households	3SLS estimation. Relies on variation in city pricing to estimate effect.

No.	Paper	Country	Waste	Period	Elasticity	Obs.	Pricing	Study type	Comments
8	Hong & Adams (1999)	US	Municipal	1992-93	-0.01	8,388	Block	944 individual households through time	2SLS estimation. Although implied elasticity is close to zero, price coefficient in the regression model is statistically significant at 90% level.
9	Hong, Adams & Love (1993)	US	Municipal	1990	-0.03	2,298	Block	2,298 individual households	2SLS estimation. Elasticity estimate is not statistically different than zero. Difficult to calculate the elasticity value from available information in the paper.
10	Jenkins (1991)	US	Municipal	1990	-0.12	NA	NA	Community pooled time series model	Unpublished PhD cited in Kinnaman & Fullerton (2000) and Strathman, Rufolo & Mildner (1995).
			Commercial	1990	-0.29	NA	NA		
11	Kinnaman & Fullerton (2000)	US	Municipal	1993	-0.28	658	Bag/Tag	Community cross-section	Endogeneity issue treated. OLS overstates responsiveness. LR effects. As price signal become clearer responsiveness increases.
		US	Municipal	1993	-0.01	658	Block	Community cross-section	
12	Linderhof et al. (2001)	Netherlands	Municipal	1993-96	-.26 SR	124,100	Weight	3,459 individual households Panel FE	Little evidence of illegal dumping. Relates to the Oostzaan local government area, which, as the first municipality in the Netherlands to implement weight-based pricing may be an especially environmental aware LGA (ie more responsive).
		Netherlands	Municipal	1993-96	-.34 LR	124,100	Weight		
		Netherlands	Municipal	1993-96	-1.10 SR (B)	124,100	Weight		
		Netherlands	Municipal	1993-96	-1.39 LR (B)	124,100	Weight		
13	McFarland (1972)	US	Municipal	1967-68	-0.46	NA	NA	13 cities in California	Cited in Gellynck & Verhelst (2007).
14	Miranda et al. (1994)	US	Municipal	1990-92	-0.20	21	Bag/Tag + Block	Average effect	Taken from the average effect in Table 2 of the study.
15	Morris & Byrd (1990)	US	Municipal	NA	-0.26	NA	NA	NA	US EPA report cited in Kinnaman & Fullerton (2000).
16	Morris & Byrd (1990)	US	Municipal	NA	-0.22	NA	NA	NA	US EPA report cited in Kinnaman & Fullerton (2000).
17	Morris and Holthausen (1994)	US	Municipal	1988	-0.51	NA	Bag/Tag	Simulation study	Reported value uses the middle elasticity estimate as the model has linear demand equation.
18	Pickin (2008)	Australia	Municipal	2000-05	-0.02	12	NA	28 individual LGAs	Not statistically different from zero. Bin size regulation more effective at reducing waste

No.	Paper	Country	Waste	Period	Elasticity	Obs.	Pricing	Study type	Comments
19	Podolsky & Spiegel (1998)	US	Municipal	1992	-0.37	149	Bag/Tag	Community cross-section	Linear demand equation so the elasticity estimate is evaluated at the sample mean.
20	Sakai, Ikematsu, Hirai & Yoshida (2008)	Japan	Municipal	2003	-0.28	39	Bag/Tag	Log-linear model	Elasticity value given is evaluated at sample means.
21	Skumatz (1990)	US	Municipal	NA	-0.14	NA	NA	NA	Cited in Strathman, Rufolo & Mildner (1995).
22	Skumatz & Breckinridge (1990)	US	Municipal	NA	-0.14	NA	NA	NA	US EPA publication cited in Kinnaman & Fullerton (2000).
23	Strathman, Rufolo & Mildner (1995)	US	All waste	1984-91	-0.11	95	Tipping fee	OLS time series across 95 months	High R^2 suggests spurious time series regression problem. Nestor & Podolsky (1996) show -.46 incorrect and that the -.11 value is correct.
					-0.46	95	Inferred		
24	Usui (2003)	Japan	Municipal	NA	-0.08	NA	NA	NA	Cited in Sakai, et al. (2008) original paper is in Japanese.
25	Van Houtven & Morris (1999)	US	Municipal	1991-94	-0.15	624	Bag/Tag	Aggregated across 16 waste collection routes. Panel data estimation	Elasticity estimates derived by using exp(B)-1 from the log-linear regression model as percentage changes. Slight difference due to pricing method.
		US	Municipal	1991-94	-0.12	624	Block		
		US	Municipal	1993-94	-0.26	796	Bag/Tag	Before and after for 398 individual households	Elasticity based on the percentage change values reported in the text. More direct pricing gives greater price responsiveness.
		US	Municipal	1993-94	-0.10	796	Block	Before and after for 398 individual households	
26	Wertz (1976)	US	Municipal	1970	-0.15	NA	Block	Community cross-section	Compares San Francisco that has charging to non-charging areas.

Table 1 provides a succinct summary of the key features of each study, and by reading down the elasticity estimate column in Table 1 it can be seen that the own-price elasticity estimates are generally close to zero. Figure 11 provides a density plot of the own-price elasticity values identified in the literature. The figure shows a very strong clustering of estimates near zero, with approximately 90 percent of estimates greater than -0.5. This shows that the demand for landfill has low price responsiveness. The point of maximum density is at the point -0.11. This in turn suggests that the best estimate of what would happen following a ten percent increase in landfill process is that the volume of material sent to landfill would fall by around one percent.

Figure 12 **Landfill own-price elasticity density plot**



Note: Unlike with a histogram or frequency polygon, the y-axis values in a density plot have no direct interpretation. The y-axis labels have therefore been deliberately suppressed. Also note that similar to a frequency polygon, the density plot joins the zero point at a point further along the axis than the last observed value, hence in the figure the plot meets the x-axis at a point where the x-axis value is positive. There are a number of reasons for preferring the data representation provided by a density plot over a histogram, including, in this case, smoothing that likely captures a publication bias effect where values close to zero show up as not statistically significant and hence are not published.

Source: Data on Table 1

The plot also suggests it is worth looking further at the three elasticity estimates in the far left-hand tail of the distribution. These are interesting cases as they are the cases where increasing the landfill levy is shown to be an effective means of reducing the flow of waste to landfill.

The first thing to note about the three landfill elasticity estimates in the far left-hand tail of the distribution is that they all relate to municipal biodegradable waste only. The second thing to note is that two of the three estimates are long-run elasticity estimates. Next, it is worth noting that two of the estimates relate to the first local government area to introduce direct municipal waste pricing in the Netherlands. As the estimates relate to a local government area that was an early adopter of waste pricing, these estimates may be thought of as relating to an especially environmentally sensitive community. This in turn suggests there is an endogeneity problem with these estimates. Given the estimates are so different from the other estimates it is reasonable to conclude that in this case the endogeneity bias has resulted in an over estimation of price responsiveness.

The third estimate in the far left-hand tail of the distribution relates to the UK, and has been calculated using the percentage change in biodegradable waste sent to landfill over the period 2005-12 and the percentage change in landfill gate prices due to the increase in the landfill levy from £18 to £56. Several things about this estimate are notable. First, the estimate picks up the effect of the other council actions to increase recycling and composting and so is a total elasticity rather than a pure price elasticity. Although, to the extent that increases in landfill taxes are generally associated with other activity the change is still relevant for policy purposes as it reflects a likely total outcome.

Second, landfill gate fees in the UK have been relatively constant at around £21 per tonne, so the landfill tax change represented a dramatic change in total landfill prices. In practice the increases in the landfill levy were of such a scale they made alternative technologies preferable to landfilling, and the timeframe considered was long enough to allow plants to be built.

Third, the government had outlined a clear path of future increases in the levy to £80 per tonne, so councils could have been acting in advance of future known price increases in landfill. If this is the case, then the percentage change in price that councils are responding to is understated, and the price responsiveness overstated.

Fourth, the UK was bound by EU related commitments and some material may have been reclassified to assist with meeting EU targets (DEFRA 2011). Although there may have been some reclassification of waste, new disposal technologies were introduced during this period and it is these new technologies that were responsible for the majority of the observed changes.

Finally, the estimate of the externality correcting landfill tax for the UK is a small fraction of the landfill tax. As such, the change was achieved through an unambiguous reduction in community welfare.

2.4 Implications for Western Australia

The following worked example provides some illustrative insights of potential responses in Western Australia based on the insights from the summary of landfill price responsiveness. For this illustrative example consider a representative landfill gate fee of \$100 per tonne for municipal waste excluding any levy charges (and ignoring other taxes and CO_2 emissions charges for the moment). Then, consider the impact of introducing a per tonne levy that changed the relative price between direct landfilling and alternatives by \$50 per tonne.[2] With an immediate period elasticity of -0.1 we then have:

$$\eta_{Q_L,P_L} = -.1 = \frac{\dfrac{\Delta Q_L}{(Q_{L_1} + Q_{L_2})/2}}{\dfrac{\Delta P_L}{(P_{L_1} + P_{L_2})/2}} = \frac{\dfrac{\Delta Q_L}{(Q_{L_1} + Q_{L_2})/2}}{\dfrac{50}{(150 + 100)/2}} = \frac{Q_L \text{ percent}}{40 \text{ per cent}}, \tag{7}$$

[2] The actual impact of any given landfill levy depends on the proportion of residual waste landfilled. For example, if, following processing there is a 50 percent residual that is landfilled the change in the relative price between the processing options and the direct landfilling option following the addition of a \$50 per tonne levy is only \$25.

so that the percentage reduction in municipal waste to landfill over the short-run is then likely to be around 4 percent. Given the change in the price in this example is substantial the potential long-run total effect can then also be considered.

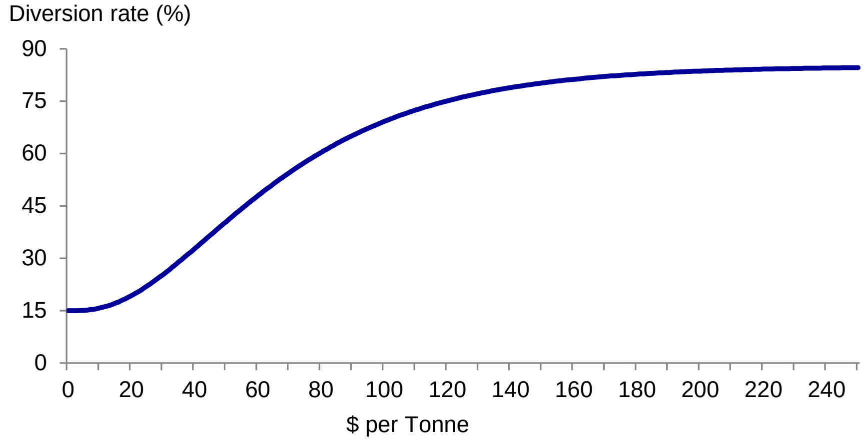
Over the long run, where there are substantial price changes the number of substitute options that become economically viable increases. Considering the long-run total impact involves issues of the time it takes to install new infrastructure (up to eight years), the issue of new alternative technologies becoming viable substitutes, and the impact of other programs that might be initiated at the local level. Based on the indicative values for different waste technologies in Hyder (2013), the price change described in this example of $50 per tonne is of the order of magnitude that would make alternative technologies viable. Assuming that this information is correct, and allowing an appropriately long time frame, say at least five years, and also allowing for the recycling of levy funding to recycling initiatives, the long-run observed quantity reduction might then be around 40 percent.

The critical issue with the long run elasticity estimate is that the change must make the total disposal cost of an alternative technology to landfilling cheaper than landfilling. If a technology threshold is not crossed, then the change will be approximately equal to the short run impact. A necessary corollary of this is that for current market conditions the long run effect is likely to be similar to the short run effect.

More generally the situation described reflects a non-constant price response. There are a number of ways to illustrate this relationship. Here the approach taken is to focus on the relationship between the gate price for landfill and the diversion rate. In Figure 13 the vertical axis measures the landfill diversion rate, and the horizontal axis measures the total landfill gate fee. Formally, the waste diversion curve can be written as: $W_t = W_\infty \times [1 - e^{-K(t-t_0)}]^3$, where W_t is the diversion rate at time *t*, W_∞ is the theoretical maximum diversion rate, *K* is the curvature parameter, *t* is time, and t_0 is an adjustment we use to take account of the fact that diversion behaviour when price is near zero be slightly differently to our the general diversion response equation.[3] There are two important features to the diversion equation. The first is that when the gate fee is zero, landfill diversion is not zero. This is because for some products there is a strong profit motive to divert these materials to recycling markets rather than to landfill. The second important feature is the sigmoid shape of the curve. This shape means that the marginal response of landfill to a given change in price is non-constant. This particular feature is illustrated in Figure 14.

[3] Although a logistic curve is a simpler expression than the Bertalanffy equation and also traces a sigmoid shape, such a curve implies symmetric marginal effects, which in the case of the landfill diversion rate is unrealistic.

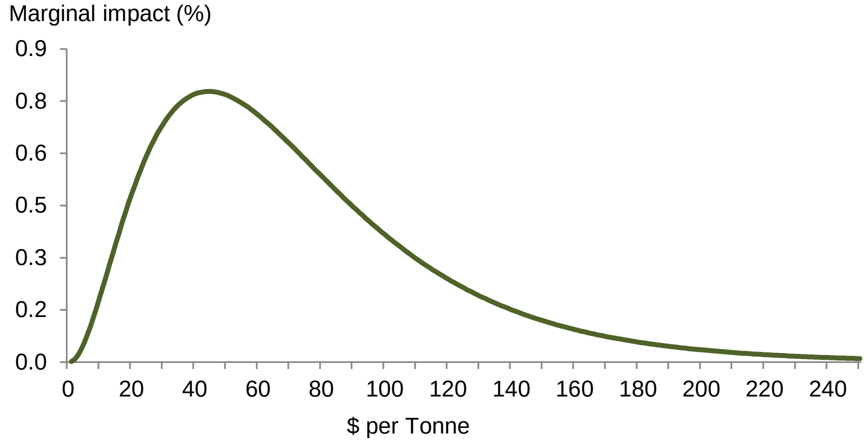
Figure 13 **Non-linear response curve: total price and diversion rate**



Source: ACIL Allen

In Figure 14 the vertical axis plots the marginal effect on diversion rates of a price change on the total diversion rate for different total gate prices. The plot shows that at low total gate prices marginal changes in total price have little additional effect. Then, over some range, as the changes in landfill gate price increasingly make alternative technologies preferable, the marginal effect is quite strong. Finally, as the overall diversion rate approaches the theoretical maximum diversion rate the marginal effect of changes again diminishes. Although such relationships exist, at some point further increases in the landfill levy have very little impact.

Figure 14 **Marginal effect of price changes in total cost of landfill**



Source: ACIL Allen

References

Allen, R. G. D., & Lerner, A. P. (1934). The concept of arc elasticity of demand. *The Review of Economic Studies*, *1*(3), 226-230.

Allers, M. A., & Hoeben, C. (2010). Effects of unit-based garbage pricing: a differences-in-differences approach. *Environmental and Resource Economics*, *45*(3), 405-428.

Bassi, S. and Watkins, E. (2012). Looking for evidence of landfill tax effectiveness in the European Union. Chapter 16 in eds L. Kreiser, A. Yábar, P. Herrera, J.E. Milne, and H. Ashiabor *Green Taxation and Environmental Sustainability*. Cheltenham: Edward Elgar.

Callan, S. J., & Thomas, J. M. (2006). Analyzing demand for disposal and recycling services: a systems approach. *Eastern Economic Journal*, *32*(2), 221-240.

Dijkgraaf, E., & Gradus, R. H. J. M. (2004). Cost savings in unit-based pricing of household waste: the case of The Netherlands. *Resource and energy economics*, *26*(4), 353-371.

Dijkgraaf, E., & Gradus, R. H. J. M. (2008). Per-unit garbage charges. *Journal of Economic Perspectives*, *22*(2), 243-244.

Fullerton, D., & Kinnaman, T. C. (1996). Household responses to pricing garbage by the bag. *The American Economic Review*, *86*(4), 971-984.

Gellynck, X., & Verhelst, P. (2007). Assessing instruments for mixed household solid waste collection services in the Flemish region of Belgium. *Resources, conservation and recycling*, *49*(4), 372-387.

Gravelle, H., & Rees, R. (1992). *Microeconomics*. 2nd, London: Longman.

Hedges, L. V., & Olkin, I. (1985). *Statistical Method for Meta-Analysis*. Academic Press.

Hong, S. (1999). The effects of unit pricing system upon household solid waste management: The Korean experience. *Journal of Environmental Management*, *57*(1), 1-10.

Hong, S., & Adams, R. M. (1999). Household responses to price incentives for recycling: some further evidence. *Land Economics*, 75(4), 505-514.

Hong, S., Adams, R. M., & Love, H. A. (1993). An economic analysis of household recycling of solid wastes: The case of Portland, Oregon. *Journal of environmental economics and management*, *25*(2), 136-146.

Hyder (2013) Strategic Waste Infrastructure Planning Project (SWIPP). Prepared for the Department of Environment and Conservation, Western Australia.

Jenkins, R. R. (1991). Municipal demand for solid-waste-disposal services: The impact of user fees. Maryland Univ., College Park, MD (United States).

Kinnaman, T. (2008). Response from Thomas Kinnaman. *Journal of Economic Perspectives*, *22*(2), 244-245.

Kinnaman, T. C., & Fullerton, D. (2000). Garbage and recycling with endogenous local policy. *Journal of Urban Economics*, *48*(3), 419-442.

Linderhof, V., Kooreman, P., Allers, M., & Wiersma, D. (2001). Weight-based pricing in the collection of household waste: the Oostzaan case. *Resource and Energy Economics*, *23*(4), 359-371.

McFarland JM. (1972) Comprehensive studies of solid waste management. In: Comprehensive studies of solid waste management, final report, sanitary engineering research laboratory, college of engineering and school of public health, report no. 72–3. Berkeley, CA: University of California; p. 41–106.

Morris, G. E., & Byrd, D. (1990). The effects of weight or volume-based pricing on solid waste management. Prepared for the US Environmental protection agency.

Morris, G. E., & Holthausen Jr, D. M. (1994). The economics of household solid waste generation and disposal. *Journal of environmental economics and management*, *26*(3), 215-234.

Nestor, D. V., & Podolsky, M. J. (1996). The Demand for Solid Waste Disposal: Comment. *Land Economics*, *72*(1), 129-131.

Pesaran, M. H., & Shin, Y. (1999). An autoregressive distributed-lag modelling approach to cointegration analysis. Chapter 11 in eds S. Strom, A. Chesher and M. Jackson *Econometrics and Economic Theory in the 20th Century: The Ragnar Frisch Centennial Symposium*. London: Cambridge University Press.

Pickin, J. (2008). Unit pricing of household garbage in Melbourne: improving welfare, reducing garbage, or neither?. *Waste Management & Research*, *26*(6), 508-514.

Podolsky, M. J., & Spiegel, M. (1998). Municipal waste disposal: unit pricing and recycling opportunities. *Public Works Management & Policy*, *3*(1), 27-39.

Sakai, S., Ikematsu, T., Hirai, Y., & Yoshida, H. (2008). Unit-charging programs for municipal solid waste in Japan. *Waste Management*, *28*(12), 2815-2825.

Skumatz, L. A. (1990). Volume-Based Rates in Solid Waste: Seattle’s Experience. *Report for the Seattle Solid Waste Utility (Seattle Solid Waste Utility, Seattle, WA)*.

Skumatz, L. A., & Breckinridge, C. (1990). Handbook for solid waste officials, Vol. 2. Washington DC: US Environmental Protection Agency.

Strathman, J. G., Rufolo, A. M., & Mildner, G. C. (1995). The demand for solid waste disposal. *Land Economics*, 71(1)57-64.

Usui, T. (2008). Estimating the effect of unit-based pricing in the presence of sample selection bias under Japanese recycling law. *Ecological Economics*, *66*(2), 282-288.

Van Houtven, G. L., & Morris, G. E. (1999). Household behavior under alternative pay-as-you-throw systems for solid waste disposal. *Land Economics*, 75(4), 515-537.

Ventosa I.P., Martínez, A.C.G. & Sora M.J. (2012). Landfill and waste incineration taxes in Catalonia, Spain. Chapter 16 in eds L. Kreiser, A. Yábar, P. Herrera, J.E. Milne, and H. Ashiabor *Green Taxation and Environmental Sustainability*. Cheltenham: Edward Elgar.

Wertz, K. L. (1976). Economic factors influencing households’ production of refuse. *Journal of environmental economics and management*, *2*(4), 263-272.

3 Externality costs

All the major studies of the externality costs associated with landfill note the existence of significant uncertainty in the calculations. The summary presented here is no different. The discussion is structured around consideration of carbon dioxide emissions issues, disamenity effect issues, and other effects. The chapter concludes with a summary that attempts to provide representative values for the current dollar impact of landfill externality costs in Western Australia.

3.1 Carbon dioxide emissions

Determining an appropriate carbon dioxide equivalent externality cost is complicated, and there are many factors to consider.

3.1.1 Discount rate assumptions

Within the externality cost literature it is important to recognise that studies can use different estimates for the statistical value of a life, or the discount rate, and that in turn these choices can have substantial implications for the landfill externality cost estimates reported. For example, consider Eunomia (2002), which is a study that includes estimates of externality costs for landfill in 21 European countries. The results contained in the main body of the report are for externality costs calculated using a discount rate of three percent and these are the values that one would naturally tend to select as the appropriate estimates. It is however noted in the report that the results are sensitivity to the discount rate assumption. Detailed calculations presented in the report appendix for Italian landfill sites show just how sensitive the externality cost estimates are to the discount rate assumption. With a discount rate of one percent the reported range for externality costs at Italian landfill sites, (including energy displacement but excluding disamenity value) is €12.63 – €15.40 per tonne of waste diverted from landfill. For discount rates of three percent and five percent, the respective values for the range of externality costs are €6.56 – €7.69 and €1.40 – €2.92 (Eunomia 2002, p. A33-A35). So depending on the discount rate assumption the externality cost for a well-run landfill could be anywhere from €1.40 to €12.63.

3.1.2 Quality of landfill site

Actual externality costs vary significantly between high quality and low quality landfill sites. For example, externality costs at relatively low quality landfill sites can easily be three times the externality costs of relatively well managed landfill sites (Eunomia 2002, p. A37-A38; Productivity Commission 2006, p. 76). If extreme comparisons are made, such as between a landfill with gas capture and electricity generation and a landfill with no gas management at all, then the externality cost difference per tonne of waste at each facility can be as high as $21 per tonne for C&I waste (Productivity Commission 2006, p. 76).

3.1.3 Marginal damage cost literature

A key input for any discussion of the externality cost of landfill is the marginal damage cost of carbon. Tol (2005) reviews 28 studies containing 103 estimates of the marginal damage cost of carbon dioxide. A number of features emerge as part of the review, including the finding that estimates of damage cost in the peer reviewed literature are substantially lower than those in the grey literature. Based on a pure rate of time preference rate of 1 percent, and using the sample mean, Tol finds a cost of carbon of $US51 per tonne, but using a time preference rate of 3 percent, which Tol suggests is consistent with the rate used by developed nations when considering long term investments, the cost of a tonne of carbon emissions falls to $US16. Although the Tol (2005) findings are widely cited it should be noted that the review is not (and does not claim to be) a structured meta-analysis. For example, the number of estimates from each study varies considerably, and no attempt is made to treat the data as an unbalanced panel with dependence across observations from a given study. These limitations mean that it is most appropriate to consider the review a narrative review.

It is also important to note that the review uses tonnes of carbon not tonnes of CO_2. In lay discussions the term carbon and CO_2 are often used interchangeably. Yet in scientific papers carbon and CO_2 remain two distinct concepts. The language in Tol (2005) is not as clear as it could be, but by considering the source data reviewed in the paper it can be established that the study is discussing the impact of carbon and not CO_2. For example, the value reported in Tol (2005) for the cost of carbon from the Nordhaus (1991) study is $US26.80, whereas the value reported in Nordhaus (1991) is $US7.33, which is for CO_2. The difference in the values is a function of the weight of the carbon content of a tonne of CO_2 relative to a tonne of C, and is 3.67.[4]

3.1.4 Social cost literature

The Regional Integrated model of Climate and the Economy (RICE) model is a highly respected climate change modelling tool. RICE-2011 was used to estimate the social cost of CO_2 emissions in Nordhaus (2011). The base case suggests a social cost per tonne of CO_2 emissions in 2015 of $US11 per tonne in 2005 dollars. Using the 2005 PPP exchange rate[5] and then inflating this to a current dollar value suggests a social cost of CO_2 emissions for Australian of around $19. With appropriate adjustments for inflation this value is lower than that reported in Anthoff et al. (2009), but comparisons are complicated by currency conversion issues.

Social cost of CO_2 emissions are, however, highly dependent on the assumptions made by the modeller. The extent of the impact of the choices made by the modeller is explored in Anthoff and Tol (2013). The authors use a different climate change model to RICE that is known as the Framework for Uncertainty, Negotiation, and Distribution (FUND) model. A number of highly cited studies have been published that use the FUND model. Ignoring equity weighting issues for the moment, the impact of different assumptions about risk aversion and the pure rate of time preference on the social cost of carbon calculated using the FUND model are shown in Table 2. As can be seen from the values in the table, differences in the assumptions have a significant impact on the estimated social cost of carbon.

[4] The ratio of CO_2 to C is (12+16+16/12) = 3.67.

[5] http://stats.oecd.org/Index.aspx?datasetcode=SNA_TABLE4 [accessed 7 June 2013]

Table 2 **Social cost of carbon (in $US1995 per metric tonne of carbon)**

	Risk aversion			
	Rates	1.0	1.5	2.0
Time preference	0.1	374	147	62
	1.0	103	45	20
	3.0	10	4	1

Note: The values are reported for tonnes of carbon not CO_2. To convert to a CO_2 value divide by 3.67. A pure time preference rate of 0.1 is consistent with the Stern review assumptions and a value of 3 percent is consistent with the value used in long term government investment projects.
Data source: Anthoff and Tol (2013)

In approaches that aim to estimate a social cost of carbon it is necessary to apply equity weights that reflect the idea that costs falling on the poor should receive a greater weight than costs falling on those that are relatively well off. Weights can be either cross-sectional, reflecting the current cross-country wealth distribution; or intertemporal, considering both the current country cross-sectional income distribution, and the income distribution for a given country through time. These issues add considerable complexity and uncertainty to the calculation of the social cost of carbon.

Using the mid-point value from Table 2 (Risk aversion = 1.5 and Time preference = 1.0), Anthoff and Tol (2013) subsequently explore, by region, the impact of different equity weightings on the estimated social cost of carbon. The first scenario considered ignores the negative impacts outside the specific region considered. For Australia, such partial analysis suggests that in 1995 US dollar terms the social cost of a tonne of carbon is $US0.41. When total global costs are considered the social cost of carbon for Australia varies between $51 and $3,433 per tonne of carbon ($14 to $935 per tonne of CO_2) depending on the assumptions made.

A number of other studies have been conducted that investigate the social cost of carbon. Yet, as the above example illustrates, social cost of carbon measures are very sensitive to the assumptions used in the model. In such an environment it will always be possible to find a value that is consistent with the position one wants to take. At a minimum, this suggests full disclosure of assumptions is required when considering social cost of carbon values.

A master student project, Schollum (2009) attempts to estimate the externality cost of landfill for Western Australia, but confuses the social cost of carbon with the cost of CO_2. Specifically, the study uses the carbon externality cost in Anthoff et al. (2009) rather than the CO_2 externality cost. Anthoff et al. (2009) reports that the social cost of carbon is $US44 in 1995 dollars or $US61 in 2008 dollars, and this is the value used in the study. However, the per tonne of CO_2 value in 2008 reported in Anthoff et al. (2009) is $US 17. The values in Schollum (2009) for the carbon externality cost per waste stream are thus overstated by a factor of 3.67.

3.1.5 Market price approach

The 2012/13 market price of a tonne of CO_2 in Australia was $23 per tonne of CO_2. This value is broadly consistent, if slightly higher than the central estimates for a number of studies that look at the social cost of carbon. The 2012/13 market price therefore seems a reasonable reference point for subsequent base case externality cost calculations.

There are several reasons to favour use of the current Australian market price of CO_2.

- The source of the value is transparent, and easily communicated

- There is no issue with the researcher imposing their own bias and preferences in terms of selecting the parameter values used in models that generate social cost of carbon estimates
- The market price value is the value actually used at landfills in Western Australia, and is also the value used in the pricing schedules that landfill operations have published.

For these reasons the current Australian market price is thought the most appropriate value to use for estimating CO_2 externality costs.

3.1.6 Alternative to a market based approach

The Federal government has indicated a clear intention to remove the price on carbon. Should this be the case, a reasonable alternative would be to use an average of the estimates from Anthoff et al. (2009) and Nordhaus (2011). As this would be an average of estimates from two different models such an approach is quite robust to criticism. If such an approach was used, the estimated marginal damage cost would also be $23 per tonne of CO_2: exactly the same value as for the market based approach.

For completeness the full set of estimates under a range of different scenarios are presented.

3.1.7 Greenhouse gas emissions externality

Calculation of the exact emission rate for any given tonne of waste is complex. For each waste component (food, timber, etc.) information is needed on the degradable organic carbon component, the fraction of degradable organic carbon, and the methane generation constant (Clean Energy Regulator 2011).

Fortunately, approximate emissions standards for a representative tonne of MSW, C&I and C&D waste have been developed. These rates are: 1.2 tonnes of CO_2-e for MSW; 1.1 tonnes of CO_2-e for C&I; and 0.2 tonnes of CO_2-e for C&D waste (MAV 2012). The maximum accepted value for methane capture from a landfill is 75 percent (MAV 2012). Using these representative values it is possible to establish some indicative values for the maximum and minimum CO_2 externality cost for landfill.

It is not clear that there is a correct CO_2 value to use. Here the preferred value is the 2012/13 market price. Representative externality cost values for each waste stream are presented below, where for completeness the implication of using different social cost of carbon estimates are also presented. The estimates are all in current 2013 dollars. The preferred set of estimates is presented in Table 4.

Table 3 **CO_2 cost for different waste streams: various benchmark (2013 dollars)**

	Units	Australian Price	Anthoff et al. (2009)	Nordhaus (2011)	Anthoff & Tol (2013) Low	Anthoff & Tol (2013) High
Assumed cost of CO_2 emissions	$ per T of waste	23.0	27.0	19.0	0.11	935
Municipal waste costs						
No methane capture technology	$ per T of waste	27.6	32.4	22.8	0.13	1,122
Best practice methane capture	$ per T of waste	6.9	8.1	5.7	0.03	280
Commercial and Industrial						
No methane capture technology	$ per T of waste	25.3	29.7	20.9	0.12	1,028
Best practice methane capture	$ per T of waste	6.32	7.42	5.22	0.03	257
Construction and Demolition						
No methane capture technology	$ per T of waste	4.6	5.4	3.8	0.02	187
Best practice methane capture	$ per T of waste	1.15	1.35	0.95	0.01	46.75

Note: Values based on the emissions factors noted in the text of 1.2 tonnes of CO_2-e for MSW; 1.1 tonnes of CO_2-e for C&I; and 0.2 tonnes of CO_2-e for C&D waste.

Table 4 **Preferred CO_2 externality estimates per tonne of waste (2013 dollars)**

	Units	MSW	C&I	C&D
Approximate emissions	CO_2 per T	1.2	1.1	0.2
No methane capture technology	$ per T of waste	27.60	25.30	4.60
Best practice methane capture (inc. energy displacement)	$ per T of waste	6.90	6.33	1.15

Note: Values based on the emissions factors noted in the text of 1.2 tonnes of CO_2-e for MSW; 1.1 tonnes of CO_2-e for C&I; and 0.2 tonnes of CO_2-e for C&D waste.

3.1.8 Existing literature and the issue of energy displacement

A number of studies have presented estimates of the greenhouse gas emission externality for landfill in Australia. Using a detailed time profile of emissions, a CO_2 price of $40 per tonne, and a discount rate of 7 percent, BDA (2009) estimate the greenhouse gas emissions externality from a tonne of waste in a dry temperate climate as between $4 (for landfill with gas capture) and $9 (for landfill without gas recovery). If energy recovery is considered BDA (2009) additionally consider the impact of the energy displaced by landfill electricity generation. For these scenarios the negative greenhouse externality associated with a tonne of landfill falls to zero for the case of energy generation over the life of the landfill, and a positive externality benefit of $1 per tonne of waste sent to landfill is derived if energy generation also takes place post landfill closure.

The comparison assumes the energy displaced is generated using coal, so that using 1998 pollution information there is a substantial reduction in sulfur dioxide emissions when landfill waste is used to generate energy.

Assumptions about CO_2 emission per tonne of waste have been revised since the publication of greenhouse gas emission externality impacts in Productivity Commission (2006). It is therefore more useful to discuss the methodology used in the report rather than the specific values reported. The method used in the report to calculate the CO_2 emission per tonne of waste liability can be understood as follows.

Let C denote the externality cost of a tonne of CO_2 equivalent emissions. Let k_i denote the tonnes of CO_2 equivalent emissions associated with waste stream i, i = MSW, C&I, C&D. Let r denote the maximum rate of energy recovery from waste, where r is greater than zero and less than one, and let d denote the energy displacement effect of generating energy from landfill waste such that $r + g = e$, where e can be greater than one, and where e is greater than one it indicates a positive externality.

For landfills without any gas capture technology the Productivity Commission methodology gives estimates of the per tonne of waste greenhouse gas emissions externality cost of $C \times k_i$, and for the case of gas capture and electricity generation the per tonne of waste greenhouse gas externality cost is $(1- e)\ C \times k_i$.

The Productivity Commission then relied on estimates of r of 0.75, and this is consistent with current policy regarding landfill liability calculations. For g the Productivity Commission relied on US data that suggested g could be 0.17. Using these parameter values, the current CO_2 market price, and current Australian values for k_i gives the range of values shown in Table 5 for the CO_2 externality cost of landfill.

Table 5 **Updated Productivity Commission impacts: CO_2 externality**

	Units	MSW	C&I	C&D
Approximate emissions	CO_2 per T	1.2	1.1	0.2
No methane capture technology	$ per T of CO_2	27.6	25.3	4.6
Best practice methane capture	$ per T of CO_2	2.2	2.0	0.37

Data source: Productivity Commission (2006) and current CO_2 price.

In an environment where there is a price on CO_2 emissions the correct treatment of the displaced energy externality is not entirely clear. The generation of energy from landfill waste does displace other energy generation, and to the extent there are differences in the emissions of these different technologies there are differences in the externality costs. However, with an appropriately set carbon price the externality has been internalised, and society appropriately compensated for the impact of burning fossil fuel for electricity generation. In such an environment it might reasonably be argued that the benefits of displaced energy could, and should be ignored.

That the maximum allowable claim landfills can make for the mitigation achieved through gas capture is 0.75 also suggests that the energy displacement effect not be included in current externality cost calculations.

3.2 Disamenity costs

Disamenity values are generally derived from hedonic price studies, where the disamenity is deduced as an implicit attribute price contributing to the total price of a house. An estimate of the dollar value per tonne of waste disamenity externality cost can then be determined based on the location of the landfill site and the surrounding population density.

To obtain an accurate estimate of the per tonne disamenity value of landfill sites it would be necessary to undertake a hedonic price study for Perth. It is however possible to obtain some insight into the magnitude of the likely disamenity cost from values reported in other studies and general concepts.

3.2.1 Existing information

The existing literature for Australia is somewhat circular, and does not seem to have an especially solid empirical foundation. For example, BDA (2009) suggest a disamenity value of $1 per tonne for well managed landfill sites and $10 per tonne for poorly managed sites. These values are informed by the disamenity value assumed in Productivity Commission

(2006) of $1 per tonne. The value assumed by the Productivity Commission is in turn informed by the value used in EPA NSW (1996), which says that the estimated range of values lies between $0 and $3.70 per tonne, where a hedonic model is used to derive the estimate.[6]

There are a number of reasons to question the relevance of the EPA NSW (1996) estimate for Western Australia, including:

- that the study used in the report has not subsequently appeared as part of the peer reviewed literature
- the evolution of hedonic price model theory and issues of omitted variables bias in hedonic models that do not account for repeat sales (see Triplet 2004; Jones 2010)
- differences in the density of housing in Perth and NSW
- improvements in environmental control systems at landfill sites over recent decades that have lowered externality costs, and
- that the estimated effect includes zero suggests a relatively low level of estimate precision. i.e. from the study it is not possible to reject the null hypothesis that there is no disamenity effect associated with landfill sites

Because of this an alternative measure of disamenity impact was sought.

3.2.2 Benefit transfer approach

When there is no direct information available on an externality cost in a specific location a benefit transfer approach can be used. A benefit transfer approach is used in Schollum (2009) to derive a WA specific disamenity value. In brief, the approach takes the estimates of the percentage house price disamenity effects from a well thought out and comprehensive UK hedonic price study (DEFRA 2003) and then applies these estimates to WA specific data for: house prices, the distribution of housing in relation to landfill sites, and the expected future life of landfill sites in WA. Using this approach an estimate of between $3.22 and $5.63 per tonne of waste is obtained.

The approach taken is both reasonable and clearly documented. However, the result is sensitive to a number of assumptions, including the discount rate used to convert waste flows into a Net Present Value equivalent, and the assumption about the remaining life of landfill sites in Western Australia.

Given the absence of clearly documented Australian specific hedonic price studies, a Benefit Transfer type approach is the most appropriate second best approach. As such the Benefit Transfer approach is used here. The specific approach proposed here requires a minimum of subjectively determined values and assumptions and can be understood as follows.

A given stock of housing provides an annual flow of consumption benefits. The flow of benefits is most readily understood as the annual rental income from housing, which in the case of owner occupier homes might be understood in opportunity cost terms. Although the annual housing stock rental yield is variable, in Western Australia, the housing rental yield appears to be around 4.0 to 6.0 percent.[7]

---

[6] The specific sentence in the report that relates to this estimate was read over the phone by an archives officer at the NSW EPA, as outside the official archived copy it has been difficult to source a copy of this report.

[7] RP Data reported in WA News article: Orr, A. Better Growth and rental yield in units, March 2013 http://www.watoday.com.au/wa-news/better-growth-and-rental-yield-for-units-20130313-2fzth.html

The estimated reduction in house prices applies only during the operational life of the landfill (DEFRA 2003; Kinnaman 2009). This suggests that a mapping of the flow of foregone housing rental income against the flow of waste to landfill provides a reasonable measure of the per tonne disamenity cost of landfill, without the need for discounting or an accurate assessment of the remaining life of landfills and the landfill compaction rate.

An essential qualification to this result is that the discount for proximity to a landfill in any given hedonic model is implicitly capturing the effect of some landfills that are just opening, some landfills that are in mid-life, and some that are almost closed.

Assuming the flow of material to landfill and the average size of landfills is approximately constant, this suggest that the raw estimate for the discount in the hedonic model should be doubled when used in any per tonne of waste calculation. This adjustment is the same as assuming all landfills are approximately half full, and the adjustment addresses the issue that the impact of a landfill on rental yield is only for the period of the operating life of the landfill.

Formally, the approach to estimating the disamenity effect can be understood as follows. Let y be the rent yield on housing expressed in percentage terms. Let r be the radius around a potential landfill site L, where the value of housing is impacted negatively by proximity to the landfill. Let h be the average value of a house located within r, and let there be n such houses. The value of the total stock of houses within the affected area is then $n \times h = H$. Let the impact of a being within r of an operating landfill site be d, where d is greater than zero and less than one. The impact of the landfill on the value of the housing stock is then $d \times H = D$, and the impact on the annual flow of benefits from this housing stock is $y \times D = C$, where C denotes the annual foregone dollar benefit attributable to the existence of the landfill. If W denotes the total annual flow of waste to landfill in tonnes, then the ratio $2C/W$ gives an expression for the disamenity cost per tonne of waste to landfill. Using the average waste volume to landfill in the metropolitan area over the past three years suggests a per tonne of waste disamenity value of $2.13 for a 4 percent rental yield; $2.66 for a 5 percent rental yield; and $3.20 for a 6 percent rental yield (see Table 6).

Table 6 **Estimates of the disamenity cost of landfill in WA**

Details	Units	Value
Volume waste (W)	T	3,088,967
Total implied Housing stock discount (2010) (D)	$	82,293,561
Low rent yield (4 percent) (C_1)	$	3,291,742
Mid rent yield (5 percent) (C_2)	$	4,114,678
High rent yield (6 percent) (C_3)	$	4,937,614
Low per tonne externality cost (C_1/W)	$ per T	2.13
Mid per tonne externality cost (C_2/W)	$ per T	2.66
High per tonne externality cost (C_3/W)	$ per T	3.20

Data source: Waste data DEC; housing stock value from Schollum (2009)

It is worth comparing the value implied using this approach to those values published in the literature for other jurisdictions. For the case of the full recovery in the value of the housing stock post landfill closure, Kinnaman (2009) suggests that the disamenity cost per tonne of compacted waste is between $US1.90 and $US2.73. It is unclear in the paper, but these values appear to be in 2001 dollars. The disamenity cost per tonne of waste sent to landfill estimate in DEFRA (2003) is £1.86. The report states this value is at current prices, which appears to be 2001 pounds. At the time, using PPP exchange rates this would translate to a value of around $3.92.

In light of these published estimates the current estimate derived here does not seem unreasonable. However, a number of caveats are warranted that suggest taking a cautious approach to the use of the estimated values. The reasons for concern are as follows:

- To the extent that the average landfill size in Western Australia has increased over recent decades the *2C/W* formula contains an average size effect error
- The diamentity cost falls rapidly as one moves away from the landfill site, so sites located away from high population centres have a relatively low disamenity cost. To the extent that larger waste facilities are located further away from areas of high population the *2C/W* formula overestimates the disamenity externality
- As odour controls and general regulation of landfill sites improve, disamenity costs falls. That there continue to be improvements in the regulatory environment that mitigate the negative effects of landfill externalities the current estimate overstates the disamenity externality. This is due to the externality cost impact of housing stock being derived from historical values.

A final point to consider that is relevant to discussions of disamenity cost relates to the use of the externality cost estimate. While there are disamenity costs associated with landfill there are also disamenity costs associated with other types of waste processing facilities. If comparing different waste treatment options the disamenity effect of such facilities must also be considered. The relative disamenity externality impact of landfill compared to other waste processing options depends on the relative housing density at each location.

So, while a central estimate of $2.66 per tonne of waste for the disamenity externality cost seems reasonable, it would be worth keeping in mind an error range of plus or minus 20 percent around this central estimate to reflect a realistic level of uncertainty in any such calculations.

Without any additional information, a reasonable estimate for the relative disamenity externality cost to other waste processing technologies might be zero.

3.3 Other externality costs

The other externality costs associated with landfill are minimal, but are considered briefly here. The literature on other externality costs is relatively sparse, but this is because other costs are small, and so for these costs it has not been seen as necessary to undertake detailed studies.

3.3.1 Leachate

Historically, there may have been grounds for considering leachate externality costs, but this no longer seems warranted for well regulated recently constructed landfill sites. The convention in a number of studies has been to assume the leachate externality is approximately zero (BDA 2009; Schollum 2010; Productivity Commission 2006). There appears no compelling reason to change from this convention when considering newly constructed landfills. It should also be noted that an externality cost only arises if the cost associated with leachate is passes on to an external party. If there is a leachate problem and a landfill operator is charged for the effective management of the issue this does not constitute an externality cost.

3.3.2 Other gas emissions

Although it relies on some Benefit Transfer values, BDA (2009) presents a detailed methodology for assessing non-greenhouse gas emissions effects. The calculations presented involved use of an unpublished spreedsheet evaluation tool, but the process used is clearly documented. As impacts relate to health effects, different values are reported for urban and rural landfills. For a dry temperate climate the estimates per tonne of waste are $0.67 with no gas recovery; $0.54 with gas recovery; and $0.96 with energy recovery. For a rural setting the estimates per tonne of waste are $0.21 with no gas recovery; $0.23 with gas recovery; and $0.08 with energy recovery. To inflate these 2008 values to current price it is necessary to multiply by 1.118.

Productivity Commission (2006, p.76) notes the externality impact of other emissions as less than a dollar for all waste streams.

3.3.3 Transport

Transportation of waste, it is sometimes argued involves negative externalities. The negative externalities associated with road transport are generally identified as comprising:

- Accident effects
- Road network damage
- Environmental and health impacts
- Congestion, and
- Depending on the country under consideration, oil dependency (Santos et al. 2010).

Accident externalities are due to the effect of each additional vehicle on the road increasing the probability of a collision accident. The public health literature generally considers the total cost associated with an accident when estimating the welfare loss associated with an accident. The total cost of an accident is, however, not an externality cost. Rather, it is only those costs not covered by insurance premiums that are an externality cost. The issue of road accident externalities is further complicated by the existence of two conflicting effects. An increase in the number of vehicles on the road leads to an increase in collision risk, but increased vehicle numbers on the road also leads to lower average speeds, which in turn implies a decrease in the severity of collision incidents.

Road damage costs are the costs that accrue to all levels of government for road repairs due to road usage, and to individuals that face higher vehicle maintenance costs due to driving on worn roads. As road damage increases in proportion to the axel load to the power of four, it is not the light vehicle fleet that is the main source of road damage externality costs, but rather the heavy transport sector (Newbery 1990).

There are a range of environmental and health externalities associated with the combustion of fossil fuels in vehicles. Although unpriced greenhouse gas emissions such as carbon dioxide (CO_2) have been given prominence in recent years, there are a range of other noxious emissions associated with vehicles that are of concern. Other emissions of concern include: nitrogen oxide, hydrocarbons, carbon monoxide, sulphur dioxide, and particulate matter. In terms of CO_2 emissions, although there is some variation in the rate of emissions depending on fuel type, the relationship is essentially proportional to the quantity of fossil fuel consumed.

Congestion externalities arise when additional traffic results in lower average travel speeds and unreliable travel times. The externality cost arises because under congested roads there is a difference between the cost faced by the individual and the total social costs for each additional car. The total social cost of an additional car on a congested road has two components. The first component is the private cost faced by those in the additional (marginal) car. The second cost is the additional (marginal) delay in travel time the additional vehicle imposes on all other road users. This relationship means that on any congested road the marginal private cost is below the marginal social cost. The extent to which there is divergence between the marginal and total social cost determines the extent of the externality.

Oil dependency arguments have been considered in oil importing countries, especially the US. The core issue that drives the oil dependency externality is the non-competitive nature of the OPEC influenced oil market (Leiby 2007). Despite importing significant quantities of certain fuel types, Australia is a net energy exporter. As such the oil dependency externality issue is not considered relevant to Australia.

These are all general externality issues and not landfill specific externalities and so these costs are not relevant to current considerations. Further, it has been foreshadowed that fuel excise tax will be reviewed by the Productivity Commission to address at least some of the market failure issues associated with the road transport sector.[8] Although the terms of reference are unknown, it is reasonable to expect the Productivity Commission to take a comprehensive approach to any review and consider the complex interactions between different transport sector externalities.

3.4 Other relevant linkages

Here, consideration has been given only to the role of externality costs associated with landfill. It is not possible to know the specific alternative options to landfilling that would be used for each tonne of waste not sent to landfill, but it is worth noting other potential environmental interactions that could be associated with lower landfill rates. For example, the UK imposes an aggregates tax of around two pounds per tonne of virgin aggregate, where the motivation for the tax was the non-market externality effects of aggregate extraction. To the extent that waste not sent to landfill produces lower externalities through use as an aggregate substitute, these external benefits are relevant.

Similar issues arise with respect to recycling. For example, it has been estimated that in Japan the external benefits from recycling are three times the external costs associated with waste disposal (Kinnaman, Shinkuma, and Yamamoto 2012).

While outside the terms of reference with respect to estimating the externality costs of landfill, to the extent that aggregate extraction and recycling externalities are not accounted for through other specific policy mechanisms, they can be considered relevant to the discussion of landfill externalities.

3.5 Summary

A summary of the estimated landfill externality costs for Western Australia are provided in Table 7. It should however be noted that the main externality effect – the effect of greenhouse gas emissions – is largely captured through current Australia taxation arrangements that address carbon externalities. This may of course change in the future. For this reason Table 7 presents cost estimates for each externality cost component. Should Commonwealth law regarding CO_2 liabilities at landfill sites change, it would be appropriate for State regulations to be adjusted to reflect these changes.

[8] See: www.pc.gov.au/carbon for details of the role of the Productivity Commission identified in the Commonwealth Government's *Clean Energy Future Plan*.

Table 7 **Landfill externality costs for Western Australia (current dollars)**

Details	MSW	C&I	C&D
	$ per T of waste	$ per T of waste	$ per T of waste
Greenhouse gas emissions			
No gas capture technology	27.60	25.30	4.60
Best practice	6.90	6.30	1.20
Best practice Inc. energy displaced	2.20	2.00	0.40
Other air emissions			
No gas capture technology	0.67	0.67	0.67
Best practice	1.07	1.07	1.07
Other externalities			
Disamenity	2.66	2.66	2.66
Leachate	0.00	0.00	0.00
Transport	NA	NA	NA
Total landfill externality			
No gas capture technology	30.93	28.63	7.93
Best practice	10.63	10.06	4.88
Best practice Inc. energy displaced	5.93	5.73	4.10

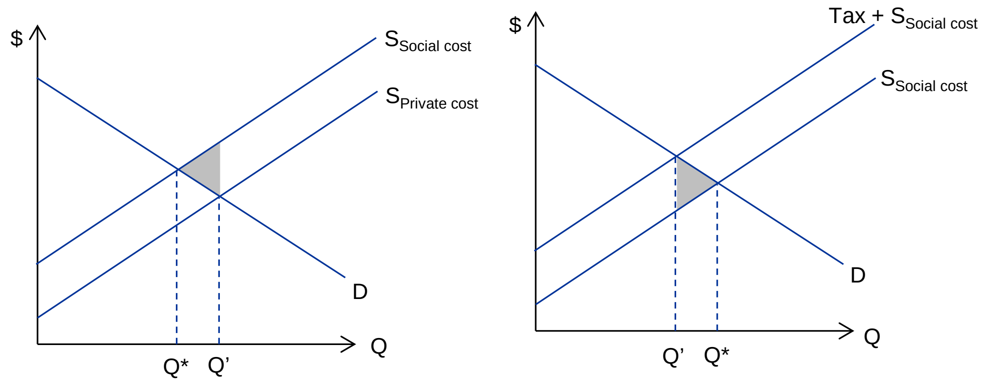
Data source: ACIL Allen estimates

Where there are externality costs it is possible that Pigouvian style taxes can increase total social welfare. Generating an improvement in total social welfare, does, however, require that the tax match the externality. Where the tax exceeds the externality cost -- even assuming that the tax revenue can be recycled as a lump sum payment to consumers -- total social welfare is not optimised. The two contrasting situations are illustrated in Figure 15.

The figure on the left illustrates a situation where there are unpriced externalities. Where there are unpriced externalities the volume of material sent to landfill is greater than the socially optimal quantity. In the figure the socially optimal quantity of landfill is Q*, while the volume of material actually sent to landfill is Q'. The grey shaded area represents, for each marginal tonne of material sent to landfill, the difference between the marginal benefit, represented by the demand curve, and the full social cost. As for these volumes the full social marginal cost is greater than the corresponding marginal benefit there is social welfare loss.

The figure on the right considers a situation where the tax rate has been set above the externality correcting tax rate. In the figure the socially optimal volume of material sent to landfill is represented by Q*, while the actual volume of material sent to landfill is Q'. For the volume of material from Q' to Q* the marginal benefit is greater than the full social cost. Too little material is sent to landfill and the grey area represents the associated welfare loss. In the figure on the right there is also a transfer of consumer and producer surplus to the government in the form of tax revenue. However, at least in principle this revenue can subsequently be recycled back to consumers and producers.

Figure 15 **Welfare implications of externalities and taxes**



Source: ACII Allen

References

Anthoff, D and Tol, R.S. (2013) The Uncertainty about the social cost of carbon: A decomposition analysis using FUND. Climatic Change, 117(3): 515-530.

Anthoff, D., Tol, RSJ & Yohe, G.W. (2009). Discounting for Climate Change. *Economics: The Open Access, Open Assessment E-Journal*, 3(24), 1-22.

BDA (2009) The full cost of landfill disposal in Australia. Report prepared for the Department of the Environment, Water, Heritage, and the Arts.

Clean Energy Regulator (2011). Supplementary guideline - Estimating an emissions profile at a landfill (2011-12 reporting period). Australian Government.

Eunomia (2002) *Economic analysis of option for managing biodegradable municipal waste*. Final report and appendices.

Jones, C. (2010). House Price Measurement: The Hybrid Hedonic Repeat-Sales Method. *Economic Record*, *86*(272), 95-97.

Kinnaman, T. C., Shinkuma, T., & Yamamoto, M. (2012). The Socially Optimal Recycling Rate. In *1st International EIMPack Congress. Lisboa*.

Leiby, P. N. (2007). *Estimating the energy security benefits of reduced US oil imports*. Oak Ridge National Laboratory.

MAV (2012). Landfills and Carbon Price Liability, Municipal Association of Vitoria

Newbery, D. (1990). “Pricing congestion: economic principles relevant to pricing roads.” *Oxford Review of Economic Policy*, (6)2, 22-38.

Nordhaus, W. D. (1991). To slow or not to slow: the economics of the greenhouse effect. *The economic journal*, *101*(407), 920-937.

Nordhaus, W. D. (2011). *Estimates of the social cost of carbon: background and results from the RICE-2011 model* (No. w17540). National Bureau of Economic Research.

Productivity Commission (2006). *Waste Management, Report no. 38.* Canberra, Australia.

Santos, G., Behrendt, H., Maconi, L., Shirvani, T., & Teytelboym, A. (2010). “Part I: Externalities and economic policies in road transport.” *Research in Transportation Economics*, 28(1), 2-45.

Tol, R. S. (2005). The marginal damage costs of carbon dioxide emissions: an assessment of the uncertainties. *Energy policy*, *33*(16), 2064-2074.

Triplett, J. (2004). Handbook on hedonic indexes and quality adjustments in price indexes. *Directorate for Science, Technology and Industry OECD (Paris)*.

4 Case studies

The following chapter presents information on a number of waste case studies. At the end of each case study a summary is presented that seeks to draw out the implications for Western Australia. The case studies cover a mix of Australian and international examples. Across each cases study different aspects are emphasised.

4.1 Netherlands

4.1.1 Historical overview

Following the end of the second world war the Netherlands underwent a period of rapid economic development. The industrialisation process saw a substantial increase in the volume of toxic waste material generated, and an increase in waste generation in general. Between 1960 and 2000 waste management in the Netherlands went from a largely unregulated system of locally operated landfill sites to a regulated system focused on resource recovery, strong regulation, a ban on landfilling most materials, and high taxes on landfilling. The brief history of developments in Dutch waste management presented here is a summary of the detail presented in Kemp (2007) and Verbong and Raven (2004).

In the mid 1960s about 75 percent of all waste was landfilled, and there were ten waste incinerators operating with little environmental controls on operations. In the early 1970s the Club of Rome "Limits to Growth" report and the subsequent oil supply shocks created a climate that heighted perceptions about the need for waste management policies that considered resource recovery.

With this context as the background a waste hierarchy emerged from a parliamentary discussion process in 1979, and was enacted as law in 1986. The hierarchy priority path, from highest to lowest priory is:

1. prevention
2. reuse
3. recycling
4. incineration (including energy generation)
5. landfilling.

Subsequent policy in the Netherlands has been driven by the waste hierarchy. Note that the waste hierarchy simply assumes the priority ordering is appropriate and does not consider cost effectiveness.

The hierarchy developed in the Netherlands is broadly consistent with that required under Article 4 of directive 2008/98/EC which specifies requirements for member states to adopt a waste hierarchy consistent with the following form:

1. prevention
2. preparing for re-use
3. recycling
4. other recovery, e.g. energy recovery, and
5. disposal.

During the 1980s there were high profile incidents where toxic pollution from incineration was found to be contaminating agricultural land, and there was also a case of a residential housing development constructed on a contaminated landfill site. This led to resistance to the creation of new landfill sites and the closure of some waste incinerators. With no decrease in the flow of waste, and shrinking processing capacity, by 1991 it had become necessary to store waste on barges. A crisis situation had developed due to a disconnect between policy decisions and the practical reality of waste management and the time lags involved in building new infrastructure.

It is also notable that at the time of the waste crisis recycling rates in the Netherlands were already quite high. For example Hill et al. (2002) reports overall recycling rates of 50 percent in the Netherlands for the mid 1980s.

The crisis situation galvanised politicians into action, and subsequently waste management rose up the national priority ranking to be an issue controlled and co-ordinated at the national level.

Core policy changes implemented following the crisis were:

- Requirement for LGAs to introduce separate collection of organics (1993)
- Ban on landfilling certain material (1994)
- Landfill tax (1995).

By 2010 the landfill tax in the Netherlands was over €100 per tonne of waste, the highest in Europe. In 2012 the landfill tax was removed due to the low volumes of material being sent to landfill.

Today, waste management policy in the Netherlands must be consistent with EU policy. The key targets in directive 2008/98/EC, which covers waste management policy appear to be a recycling/ reuse rate of at least 50 percent for the MSW and C&I waste streams, and a recycling/ reuse rate of at least 70 percent for the C&D stream by 2020. The Netherlands already meets these targets (Milios 2013; Hendriks and Janssen 2001).[9] [10]

4.1.2 Landfill tax details

The history of the landfill tax in the Netherlands is described in a number of documents. The details presented here report the values described in Fisher et al. (2012) and Milios (2013).

In 1995 a landfill tax was introduced at €13 per tonne, for all waste. In 2000 two landfill tax rates were introduced, one for dense material that, implicitly, is non-combustible, and a rate for combustible waste. The combustible rate was set at €65 per ton, and the non-combustible rate was left unchanged. Substantial further increases were applied to the combustible rate, and modest increases to the non-combustible rate such that prior to the repeal of the landfill tax in 2012, the tax rates stood at almost €17 per ton for non-combustible waste, and almost €110 per ton for combustible waste (banned waste that can only be landfilled with a permit).

There are long construction times for adding incineration capacity, and this means that processing capacity cannot be added quickly. For a number of years after the increases in the landfill tax that started in 2000, substantial waste was exported from the Netherlands, as it was cheaper to export waste than pay the landfill tax (Fisher et al. 2012).

Landfill tax revenue peaked in 2001 at around €185M and has subsequently fallen to the point where revenue collected does not justify the administrative burden. The landfill tax has therefore been removed.

---

[9] http://scp.eionet.europa.eu/facts/factsheets_waste/2009_edition/factsheet?country=NL [accessed 7 June 2013]

[10] The only reference that could be found to the 2009-21 National Waste Management Plan were in Dutch.

The landfill tax has been associated with additional diversion from landfill, but it is worth noting that prior to the introduction of the landfill tax in 1995 over 70 percent of total waste in the Netherlands was either recycled or composted (Bartelings and Linderhof 2006).

4.1.3 Other legislation: landfill ban

Although exemptions were made due to the lack of available infrastructure, around the time of the introduction of the landfill tax a ban on sending biodegradable and combustible waste to landfill was also enacted, and construction of new landfill sites restricted. The number of landfill sites was, however, already trending downwards as the Netherlands moved from a poorly regulated locally controlled landfill system to a national, well regulated system. For example, in 1977 there were 450 landfills in the Netherlands; in 1995 there were 46; in 2000 there were 40; in 2003 there were 30; and today there appear to be around 30 (Kemp 2007; Fisher 2012; Milios 2013).

The landfill ban for C&D waste became effective on 1 April 1997, and prohibits the landfilling of reusable or burnable C&D waste.

4.1.4 Construction and Demolition Waste

The volume of C&D waste in the Netherlands is equal to around 10 percent of the volume of raw material required for new construction each year, and the natural environment in the Netherlands is not abundant with raw aggregate material (Hendriks and Janssen 2001). This in turn means that C&D waste has a ready market in terms of use as road base. For example, long before the landfill tax was introduced, the recycling rate for C&D waste was over 60 percent and rising.

Table 8 **Construction and Demolition waste flows in the Netherlands (1990-2000)**

Detail	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000
	M T	M T	M T	M T	M T	M T	M T	M T	M T	M T	M T
Landfill/ Incinerated	5.0	5.0	5.1	4.2	3.8	3.1	2.9	2.6	2.1	1.7	1.4
Reuse/Recycled	7.7	7.6	7.6	8.6	9.2	9.9	10.5	11.1	11.8	12.4	12.8
Total	12.7	12.6	12.7	12.8	13.0	13.0	13.4	13.7	13.9	14.1	14.2
Reuse/Recycling rate	60.6	60.3	59.8	67.2	70.8	76.2	78.4	81.0	84.9	87.9	90.1

Note: Volumes exclude pre-crusher fines.
Data source: Hendriks and Janssen (2001).

Although use in road base remains the predominate use for material, a revised (flexible) hierarchy for the C&D waste steam of:

- Prevention
- Construction reuse
- Element reuse
- Material reuse
- Useful Application
- Immobilisations with useful application
- Immobilisation
- Incineration with energy
- Incineration
- Landfill,

has been proposed (van Dijk 2001). Consistent with such a hierarchy, work continues on technologies that will see higher recovery rates for C&D waste, and C&D waste used in higher value production activities, for example Mulder et al. (2007).

4.1.5 Economic evaluation of the system

The Netherlands was a leading nation in terms of the adoption of the waste hierarchy. The waste hierarchy does not consider cost effectiveness. It is therefore worth considering what cost, if any, the adoption of the waste hierarchy imposes on society. Some insight into the welfare loss imposed on society from the waste hierarchy can be obtained by considering the total social cost associated with landfilling waste and the incineration of waste.

Using a full social cost model Dijkgraaf and Vollebergh (2004) assess the appropriateness of the Netherlands preference for incineration of waste over landfilling. Under a wide range of assumptions they find no case where the full social cost of incineration with energy capture is less than the full social cost of landfilling. Details for the authors preferred set of values are shown in Table 9, and as can be seen, accounting for the full social cost of each option, landfilling is the cheaper option. The comparison is presented in both Euros and Australian dollars and as can be seen, the loss in welfare from landfilling relative to landfilling is meaningful.

Table 9 **Total social cost of waste disposal: Netherlands (2000)**

	Landfilling		Incineration	
	€ per T	$ per T	€ per T	$ per T
Gross private cost	*40.00*	*58.31*	*103.00*	*150.15*
Private cost savings				
Energy	4.00	5.83	21.00	30.61
Materials	0.00	0.00	3.00	4.37
Net private cost	***36.00***	***52.48***	***79.00***	***115.16***
Environmental costs				
Emission to air	5.85	8.53	17.26	25.16
Emissions to water	0.00	0.00	0.00	0.00
Chemical waste	2.63	3.83	28.69	41.82
Land Use	17.88	26.06	0.00	0.00
Gross environmental cost	*26.36*	*38.43*	*45.95*	*66.98*
Environmental cost savings				
Energy	4.76	6.94	22.62	32.97
Materials	0.00	0.00	5.76	8.40
Net Environmental costs	***21.60***	***31.49***	***17.57***	***25.61***
Total Social Cost	**57.60**	**83.97**	**96.57**	**140.78**

Note: Conversion to Australian dollars based on the 2003 PPP exchange rate information available at http://stats.oecd.org/Index.aspx?datasetcode=SNA_TABLE4 [accessed 7 June 2013]
Data source: Dijkgraaf and Vollebergh (2004)

In the above table, the land cost value assumes that the alternative use for the land is housing construction; hence the value is quite high. Specifically, if the value is adjusted for inflation and converted to Australian dollars it implies a current value of around $490 per m^2, or around $245,000 for a 500 square metre housing block. It would not be appropriate to use such a high value for less densely populated jurisdictions, such as Western Australia. Measured in terms of total social cost the relative performance of landfilling in Western Australia can therefore be expected to be substantially lower.

Other analyses, such as that contained in Bartelings et al. (2005) also show landfilling to be preferable in terms of total social welfare. For example the best total social cost estimates in Bartelings et al. (2005) are €45 per tonne for landfilled waste; €112 per tonne for incinerated waste.

4.1.6 General summary and implications for Western Australia

The Netherlands has had relatively high recycling rates since at least the mid 1980s. The relatively high recycling rate can be explained by a number of factors, but the lack of low cost virgin aggregate was clearly a factor in the reuse and recycling of C&D waste. That suggests that there is a strong link between the market for raw material in construction and the market for recycled C&D waste products.

Prior to the era of EU led waste management policy, circumstances in the Netherlands -- scandals with existing landfills and incinerators, and a crisis in terms of a lack of disposal capacity -- were such that the public were receptive to substantial changes in the approach to waste management, such as the adoption of the waste hierarchy and banning a wide range of material from landfill.

The process of transition in the Netherlands was not smooth, and there were several instances where a mis-match between policy decisions and the practical realities involved in waste management lead to undesirable outcomes.

In the long run, low rates of waste to landfill are driven by the regulatory environment -- in terms of restriction on the establishment of landfill and the material than can be sent to landfill -- not the landfill tax. This can be seen in that the landfill tax in the Netherlands was removed in 2012.

The waste hierarchy does not consider cost effectiveness. Direct application of the waste hierarchy can, therefore, be associated with substantial welfare loss.

4.2 Canada

The oil and mining industry is the largest generator of waste in Canada, and in 2008 the combined waste rock, sand tailing and other waste associated with the oil and mining industry was 1,118 million tonnes. These waste products are, however, stored in situ (Statistics Canada 2012).

In Canada, environmental regulation in general, and so landfill tax rates in particular, are determined at the Province level. Landfill taxes are not widely used in Canada and Kelleher (2012) reports the following landfill tax rates in Canadian Provinces (in Canadian dollars):

- Quebec $20.69 per tonne (including a temporary $9.50 charge to support infrastructure investment for organics)
- Manitoba $10 per tonne.

In both Quebec and Manitoba the majority of the revenue derived from the landfill tax is recycled back to municipalities to support investment in waste diversion infrastructure.

In terms of the relative importance of the landfill tax as a proportion of total landfill costs, it can be noted that in Winnipeg (Manitoba) the current gate fee (in Canadian dollars) for waste excluding the landfill tax is $22.50 per tonne for residents and $33.50 for commercial waste.

A levy of $10 per tonne is therefore a substantial proportion of the total cost of sending material to landfill.[11]

From the available information, no clear impact in terms of diversion rates seems apparent from the use of landfill taxes to raise the price of sending material to landfill in Canada. The lack of a consistent picture is detailed in Table 10. The table shows information on both the waste disposed of, and waste diverted from landfill for the two Provinces with landfill taxes, and Canada as a whole. To control for the effect of Province size the information is presented on a per capita basis.

Both Quebec and Manitoba dispose of more waste than the average Canadian Province. In terms of the change between 2008 and 2010, the rate of decline in waste disposed has been above average in Quebec, but below average in Manitoba. Considering the level of waste diverted information, Quebec diverts more waste than average for Canada, while Manitoba diverts less. Considering the change in waste diverted information, between 2008 and 2010 diversion performance in Manitoba improved, although the improvement was from a low base. In Quebec, waste diverted fell at a faster rate than for Canada as a whole.

Table 10 **Comparative waste performance: Canadian regions (per capita kg)**

	Canada			Quebec			Manitoba		
	2008	2010	% Δ	2008	2010	% Δ	2008	2010	% Δ
Waste disposal per capita	778	729	-6.2	793	733	-7.5	784	770	-1.8
Waste diverted per capita	249	236	-5.3	318	296	-7.0	137	144	5.1

Source: Statistics Canada (2010)

Although there is variation across the Provinces, in Canada the volume of solid waste that is diverted is attributable equally to the residential and non-residential sector (Statistics Canada 2012). The main materials recycled are Organics (39 percent), Paper and Cardboard (41 percent), and C&D waste (8 percent).

Although landfill taxes are not widespread in Canada, international comparisons suggest Canada performs somewhere in the mid-tier of countries in terms of waste diversion (World Bank 2012).

4.2.1 Construction and Demolition in Ontario

All construction and demolition projects in Ontario, where the total floor area is at least 2,000 m^2, are subject to waste minimisation regulations. The waste minimisation regulations are aligned with the triple R waste hierarchy of Reduce, Reuse, and Recycle. The specific requirements of Ontario Regulation 102/94 & 103/94 include:

- Preparation of a written waste reduction plan and conducting a waste audit
- Source separation for recyclables.

In terms of the waste audit, the issues that must be covered are:

- Quantity, including composition, of waste generated across all aspects of the project
- How the waste is produced
- How the waste is managed (Ontario Government 2008).

[11] Based on charging at the Brady Road Landfill available http://www.winnipeg.ca/waterandwaste/garbage/bradyroad.stm accessed 25 August 2013.

4.2.2 Municipal waste in Markham City, Ontario

In 2001 waste diversion in Markham City, Ontario, was 33 percent. Between 2001 and 2007 the diversion rate in the city rose to around 70 percent and had been broadly stable at that level until 2012 (City of Markham, 2012). In 2013 the city was able to increase the diversion rate to over 80 percent.

Since 1991 the Ontario government has required municipalities to provide a kerbside recycling service. The 33 percent diversion target might therefore be seen as consistent with diversion performance using a basic system approximating a two bin dry recyclables system. Markham City now uses a source separation system that is more involved than a simple kerbside program. Specifically, Markham requires residents to source separate waste into four streams: dry recyclables, organics, garden waste, and residue. To ensure compliance a system of clear plastic bags are used for the residual waste stream. The use of clear plastic bags means that collection officers can make a quick assessment of the extent of contamination. The city policy is that should a residual waste bag contain 25 percent or more recyclable material the bag is not collected and education material is left behind.

4.2.3 When landfill runs out: Toronto

The history of diversion performance in Toronto, the largest Canadian city, is documented in Krausz (2012), and the history presented here is based on the information in Krausz. In 1996 the city of Toronto was faced with the problem of very limited landfill capacity and negotiated to send waste to a landfill in Michigan, approximately 400 km away. Waste from Toronto was then landfilled in Michigan between 1998 and 2010, with Michigan local resistance to accepting waste from Toronto gaining substantial traction from 2004 onwards. From the start of 2011, the city of Toronto again had access to a landfill site in Canada.

In 2001 the cost of landfilling material was around $12 per tonne (diversion rate 27 percent), but rose to around $52 per tonne with the Michigan landfilling solution. At this time the city adopted a zero waste target. The target was announced by leading political figures in a context where it was well known that the available landfill options were extremely limited.

In 2002 Waste Diversion Ontario (WDO) was created to manage waste diversion activities, such as the dry recyclables (blue box) program, and by 2003 the diversion rate had increased to 32 percent. An increase in landfill costs of over 400 percent was therefore associated with an increase in the diversion rate (in the short run) of only 6 percentage points.

In 2005 the city moved to a single combined dry recyclables collection system, and made participation in recycling mandatory. Specifically, legislation was enacted to stop refuse collection for single residences that did not participate in the recycling program. At around this time the diversion rate had increased to around 40 percent.

In 2007 the city secured a new landfill site, and in 2010 the city had a diversion rate of 47 percent.

4.2.4 General summary and implications for Western Australia

A lack of access to landfill sites is a catalyst for changes in thinking, but with operating costs at landfill largely set to reflect just operating costs. Even when waste must be transported substantial distances, landfilling is still the most cost effective option.

A high level of source separation, and use of mechanisms, such as clear bags for residual waste, can result in very high diversion rates for municipal waste.

Regulations, in terms of a requirement to have a waste management plan for large and medium construction sites can assist with diversion, and waste reduction. The impact such programs have on housing construction costs would depend on interactions at different aspects of the C&D recycling supply chain.

4.3 United Kingdom

The UK is subject to the EU directive on waste to landfill and so was required to implement policies that would see the volume of waste sent to landfill reduce substantially before 2010. UK policy settings are also consistent with the EU waste hierarchy. Over time, both the Labour government and the Conservative – Liberal Democrat government have implemented a number of interrelated policies to ensure that the EU landfill diversion target was met. These policies include: the introduction of a landfill tax; the introduction of a virgin aggregates tax; and the use of a landfill cap and trade system for biodegradable municipal waste. These policies are discussed below.

4.3.1 Landfill tax

In 1996 a landfill tax was introduced in the UK. The impact of the landfill tax in the first years of operation is documented in Martin and Scott (2003). The summary of the impact of the tax in the early years of operation presented here is based on the detail in Martin and Scott.

When first introduced, the objectives of the landfill tax were to internalise the environmental and social externality costs associated with landfill, and support the diversion of waste from landfill. In terms of diverting waste from landfill the preference was for actions higher up the waste hierarchy so that reduction in waste generation and reuse are seen as preferred to recovery; and disposal is seen as the least preferable option.

The initial landfill tax rates were set with reference to externality costs, and the externality correcting landfill tax rates were estimated to be £7 per tonne for active waste and £2 per tonne for inert waste.

A 1998 review of the impact of the landfill tax by HM Customs and Excise determined that the tax was having little impact, especially in terms of the putrescible waste stream. Subsequent to the review, the landfill tax on putrescible waste was increased (see Table 11). The staged increase in the tax was an attempt to balance the need to encourage diversion, and at the same time allow operators and councils time to adjust to higher tax rates. The government has committed to maintain the 2014/15 landfill tax rate of £80 per tonne as a price floor until at least 2020. With a landfill tax rate of £80 per tonne, it is expected that by 2020 the landfill diversion rate for MSW and C&I will be around 75 percent (DEFRA 2011).

Table 11 **UK landfill tax rates**

Year	Putrescible	Inert
	£ per T	£ per T
1996-97[a]	7.00	2.00
1997-98	7.00	2.00
1998-99	7.00	2.00
1999-00	10.00	2.00
2000-01	11.00	2.00
2001-02	12.00	2.00
2002-03	13.00	2.00
2003-04	14.00	2.00
2004-05	15.00	2.00
2005-06	18.00	2.00
2006-07	21.00	2.00
2007-08	24.00	2.00
2008-09	32.00	2.50
2009-10	40.00	2.50
2010-11	48.00	2.50
2011-12	56.00	2.50
2012-13	64.00	2.50
2013-14	72.00	2.50
2014-15	80.00	2.50

[a] Tax introduced October 1996

Source: www.hmrc.gov.uk/rates/landfill-tax.htm for latest year, and HM Revenue and Customs Landfill Tax (LFT) Bulletin Table 6 for historic rates.

Despite concern in the initial years that the landfill tax was doing little to change behaviour in term of waste generation, there were significant changes in diversion rates for some waste streams following the introduction of the tax. For example, following the introduction of the landfill tax there was a material improvement in the landfill diversion rate for inert (largely C&D) material. Specifically, the proportion of inert material sent to landfill fell from 51 percent in 1996 to 24 percent in 2000.

Some indication of what happened to the inert material diverted from landfill can be obtained be considering trends in UK construction activity and sales of primary aggregates. Analysis in European Environment Agency (2008) shows that prior to 1996 there is a strong positive correlation between construction activity and primary aggregate use. In 1996 there is a structural break in the relationship so that over the next decade, while there is strong growth in construction activity, primary aggregate use falls sharply. A reasonable inference from this relationship is that much of the inert material diverted from landfill was used as a substitute for primary aggregates.

In the early years some waste streams, such as coloured glass, continued to be a problem due to a lack of final markets for products.

The UK was deemed to meet the 2010 EU landfill diversion targets, although meeting the target was assisted by some changes in the classification of material to different waste streams (DEFRA 2011).

4.3.2 Primary Aggregates Tax

Motivated by the idea that extraction of primary aggregates involves externality costs – biodiversity costs, lost amenity value, etc. – not captured by the market price, the UK introduced a primary aggregates tax in 2002. The impact of the aggregates tax is discussed in European Environment Agency (2008).

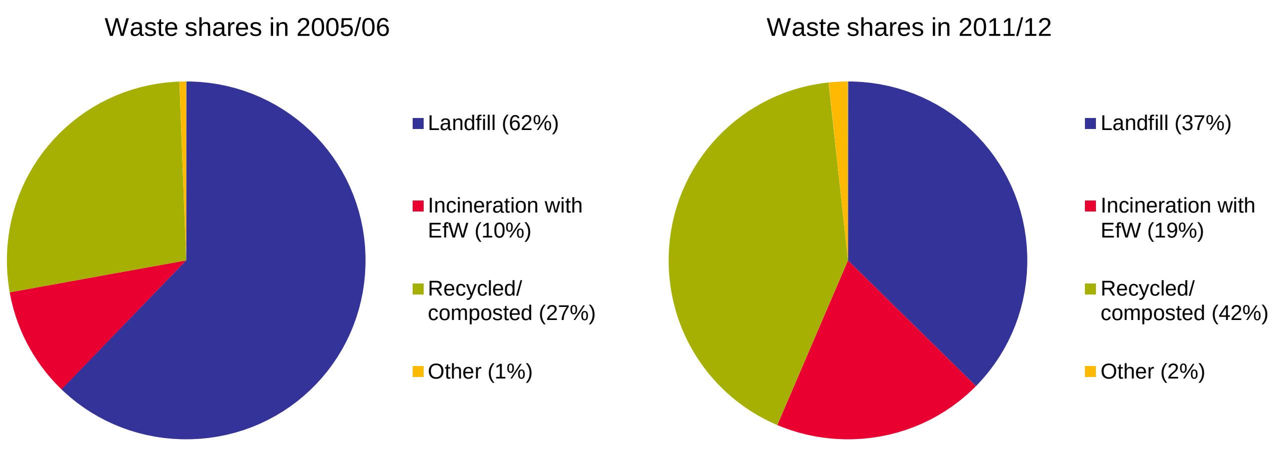
The tax was introduced at a rate of £1.60 per tonne and subsequently increased to £1.95 per tonne in 2008. Determining the impact the aggregates tax had on the volume of material sent to landfill is difficult. This is because the tax was just one of a number of interrelated policies. The estimate provided by the UK Quarry Products Association for the impact of the aggregates tax on material recycling is reported in European Environment Agency (2008). This estimate suggests that prior to the aggregates tax the supply of recycled material was increasing by around two million tonnes per annum, and that after the introduction of the aggregates tax the annual increase in the supply of recycled material rose to around three million tonnes.

4.3.3 Municipal sector actions

The final destination of municipal waste in England is reported in DEFRA (2012). Since 2000/01 the volume of waste collected by LGAs in England that is sent to landfill has fallen from around 22 million tonnes to less than 10 million tonnes. During the same period, the amount of material recycled/composted has risen from around 3 million tonnes to almost 11 million tonnes. The amount of waste incinerated for energy generation during this period has approximately doubled to 5 million tonnes, with much of the growth in incineration taking place recently. Information on the changes in waste destination since 2005/06 are shown in Figure 16, and the landfill tax continues to provide a strong incentive for LGAs to divert waste from landfill.

In 2012 the median landfill tax exclusive gate price for sending a tonne of non hazardous waste to landfill was of £21 (WRAP 2012). So, in the UK, a landfill tax of around 2.7 times the landfill gate price is associated with a landfill diversion rate of 63 percent for MSW.

Figure 16 **Changes in the municipal waste shares in the UK**



Note: Other includes incineration without energy recovery
Source: WasteDataFlow, Department for Environment, Food and Rural Affairs (Defra); Available: https://www.gov.uk/government/organisations/department-for-environment-food-rural-affairs/series/waste-and-recycling-statistics

The landfill tax was not the only measure used in the UK to target waste diversion in the Municipal Sector. Between 2005/06 and 2012/13 a cap and trade system was in place for biodegradable municipal waste. The system, known as the Landfill Allowance Trading Scheme (LATS), involved annual decreases in the total volume of biodegradable material that each LGA was allowed to send to landfill. Any LGA not able to meet the target could purchase allowances from other LGAs that had surplus permits. The system saw significant reductions in the volume of waste sent to landfill (Environment Agency 2011). More generally, where there is a fixed quantity target, as there is under the EU landfill directive, a cap and trade system can ensure the specific quantity goal is met. The LATS system is no longer in place as the landfill tax has risen to such a point that it is no longer needed (DEFRA 2011).

4.3.4 Commercial and Industrial waste

Summary information on C&I waste in England is reported in Jacobs (2011), and the values from the report are presented here. Waste generation in England is split evenly between commercial and industrial sources. Between 2003 and 2009 waste arising from both the commercial sector and the industrial sector fell, although in the case of the industrial sector the number of businesses also fell. Overall the proportion of C&I waste sent to landfill fell from 41 percent to 24 percent.

During this period the landfill tax rose from of £13 per tonne to £32 per tonne, with the landfill tax then rising to £40 per tonne later in 2009.

WARP (2008) reports that pre-tax landfill gate fees in the UK have been constant for a number of years at around £21 per tonne. Given the existence of a clear forward price path for the landfill tax that is increasing strongly, it is necessary to be cautious in interpreting the observed changes in landfill diversion rates. Nevertheless, it seems clear that for landfill tax rates that more than double the cost of landfilling, the diversion rate in the C&I sector are substantial.

4.3.5 General summary and implications for Western Australia

A low landfill tax rate can have a significant impact on some waste streams, such as C&D waste, but has little impact on the putrescible waste streams.

A cap and trade system for waste can be an effective way to reduce the volume of material sent to landfill.

High landfill tax rates result in greater diversion of waste from landfill, but also encourage the use of Energy from Waste technology.

Landfill tax rates and other charges on virgin aggregate extraction are broadly consistent with externality correcting values. With charges set at this level there is substantial diversion of material from landfill.

For putrescible waste, when the tax rate is set at a level to reflect the externality costs associated with landfill this has little impact on diversion rates.

4.4 Italy

As with other European nations, Italy is subject to the EU directive on landfilling waste and increasing landfill diversion has been a focus over recent decades. There is significant variation in waste management approaches at the local level in Italy and this variation allows the impact of different policies to be determined.

4.4.1 Municipal waste diversion

Using data from 1999 to 2006 for 103 provinces Mazzanti et al., (2009) investigate the impact of different policy and other variables on municipal landfill diversion rates. Key findings from the study are discussed below.

Landfill taxes did not have a statistically significant impact on landfill diversion rates. Landfill taxes were between €10 and €30 per tonne, with a sample average landfill tax rate of €15. Gate fees for landfill are around €80 per tonne, and the relatively small proportion of total cost that landfill taxes represent was suggested as a reason that no effect of landfill tax rate on diversion was detected. Lax enforcement in another possible reason suggested for failing to detect an effect of landfill taxes on landfill diversion rates.

Evidence is found that supports the idea that there are lock in effects from investment decisions about which waste technology solution -- landfilling, incineration -- is used. In a related fashion, the relative abundance of landfilling opportunities and incineration opportunities are found to influence the landfill diversion rate.

Although no statistically significant effect for landfill taxes was found, more generally the study reported that:

> A good performance on managing waste according to economic rationales (influencing prices) also helps reducing [sic] the amount that is landfilled. In association to the key features of the tariff system, we also underline the key role played by the share of separated collection at the very heart of the waste chain: where it is higher, landfill diversion is higher (p. 60).

In terms of diversion rates an interesting finding was that rather than the proportion of the population covered by a waste tariff, it was the share of municipal governments that introduced the tariff that matters. A possible interpretation of the result being that there is something additional in terms of the impact on diversion rates when many geographical locations implement a change.

4.4.2 Capital investment and tendering

A number of papers have considered the issue of waste management in the Campania region of Italy. Here attention is given to the tending process used when seeking waste management solutions and the impact decisions made at the tender level can have for longer term waste management. The issues associated with the tender are detailed in D'Alisa et al., (2010) and are summarised below.

In 1998 a tender was let for the provision of waste management services that was ultimately awarded to a conglomerate that relied on incineration technology. The tender evaluation format was similar to that used in many Australian government tender evaluations. There were four tender evaluation criteria:

1. Price
2. Speed of construction
3. Technical proficiency
4. Technical value.

The form of the assessment was a weighted additive assessment structure where each criterion is given a weight. A score is then determined for each criterion for each tender, and the final tender score is found as the sum of the product of the individual criteria scores and weights. For example, if each criterion had an equal weight, and a specific tender achieved scores of 5, 10, 10, 0 for the four criteria, the final tender score would be found as: $(0.25 \times 5) + (0.25 \times 10) + (0.25 \times 10) + (0.25 \times 0) = 6.25$.

A potential issue with such an evaluation process is that tenders scoring zero in terms of aspects such as value for money, environmental benefit, position in the waste hierarchy, effect on reducing waste to landfill, etc., can still be ranked as the best tender. Such a result can be counter to the desired policy outcome. For example, in the case of Campania, even if the incineration technology tender had scored zero for both Technical proficiency and Technical value it would still have been ranked the best tender.

If, rather than using an additive tender evaluation model a multiplicative evaluation model had been used, a ranking of zero for any given criteria would result in an overall evaluation of zero.

Subsequent to letting the tender other issues emerged. For example, although the operating costs for incineration were estimated to be €41.50 per tonne, they turned out to be €88.44 per tonne.

A final aspect of the process that is noteworthy is the relationship between investment costs and the time taken to recover capital investment. The scale of investment required for incineration technology is such that security of a minimum volume of waste material is required for a number of years. In the Campania region this supply uncertainty issue for the incineration project was addressed through the imposition of ‘deliver or pay’ contract terms for local government. With a deliver or pay contract there is no incentive for local government to invest in recycling or waste reduction programs. As can be seen from the detail in Table 12, source separation in Naples, the main city in the region is less than that observed in other major Italian cities.

Table 12 **Variation in waste management performance: Italian cities**

	Naples	Rome	Milan	Turin	Bari
Waste separation 2001 (%)	4.89	3.83	37.87	20.46	5.97
Waste separation 2005 (%)	7.71	11.79	43.35	36.36	10.31
Per capita, per day waste generation (kg)	1.43	1.82	1.35	1.43	1.34

Source: D’Alisa et al., (2010)

4.4.3 General summary and implications for Western Australia

The relationship between landfill diversion rates and the extent of source separation is relevant when considering the types of technology solutions to waste that are preferred.

That, for municipal waste, no effect of landfill taxes could be detected is consistent with the findings of the broader landfill price elasticity literature review and suggests landfill taxes have little impact of putrescible waste.

Landfill levy revenue provides the opportunity to support investment in different waste technology infrastructure projects. However, unless tender evaluation criteria are designed with care, it is possible that perverse outcomes will be realised.

4.5 Japan

In Japan, the Waste Management and Public Cleaning Law classifies waste as either industrial waste or municipal waste. Waste from offices, and restaurant garbage are, however, classified as municipal waste. This can make interpreting waste statistics from Japan confusing. Landfill construction costs in Japan are high. Putrescible landfill waste construction costs in 2003 were estimated to be between $US169 and $US338 per tonne, and inert landfill construction costs were estimated to be between $42 and $68 per cubic metre. Japan is therefore characterised by high gate fees for waste. Transport costs are also relatively high, at around $US25 to $US42 per tonne for typical waste journeys (Nakamura 2007). Similar total cost values for landfilling are also referred to in Nakayama et al. (2012). Incineration costs, at between $US 211 and $US 338 per tonne are also high (Nakamura 2007).

Historically, illegal dumping has been a problem in Japan. For example, in the ten years to 2004, across Japan approximately 5 million tonnes of waste was illegally dumped. Reasons for the high illegal dumping rate suggested in Nakamura (2007) include:

- High cost of landfill
- Close proximity of mountainous regions where waste can be dumped
- Inability to trace waste from point-to-point through the waste system
- Involvement of organised crime in waste disposal.

In terms of waste management policy, Japan has a triple R framework that is based around the concepts of reduction, reuse, and recycling. Similar to the waste hierarchy, reduction in waste generation is preferred to reuse; which in turn is preferred to recycling. Japan has, however, experimented with a number of different policy mechanisms for managing waste, and this experience provides some important insights that are relevant to Australia.

4.5.1 Municipal waste: Pay as you throw systems

Across the Municipal waste sector Japan's approach might be described as including some additional steps to the triple R model, where for residual waste there is also a volume reduction step (incineration) before final disposal (Tanaka 1999).

The literature review found that the landfill price elasticity is almost universally close to zero for Municipal waste. One potential reason for the low responsiveness of consumer to higher waste charges in the Municipal sector is that price changes are not clearly transmitted to customers. A number of local governments in Japan have experimented with Pay-As-You-Throw (PAYT) systems that involve a much clearer price signal to customers. Japan's experience with PAYT systems for municipal waste is described in Saki et al. (2008) and is summarized below.

There are a number of variations on the PAYT approach, but in Japan the main systems can be described as either simple unit pricing systems or two tier pricing systems. A simple unit pricing system involves payments that are proportional to waste generation. The mechanism for achieving this is usually something like a fixed price per bag of waste. There is some variation in two tier pricing systems but they generally involve something like a fixed fee for a basic service (a given number of free collections, say) and then charges after waste generation reaches a certain level. In 2003 approximately 30 pecent of local governments had some form of unit pricing scheme and these local government accounted for about 14.5 percent of the population. This in turn suggests that in Japan smaller local governments have adopted unit pricing to a greater extent than large metropolitan centres. The effect of the policy is described through four case studies.

Shingu city

In 2002 Shungu city (population 34,000) introduced a two-tier pricing scheme, along with an improved recycling service. Following the change there was:

1. a reduction in total per capita per day waste generated (including recyclables) of 18 percent
2. a reduction in per capita per day residual waste incinerated/landfilled of 25 percent
3. increase in the recycling rate from 15 percent to 22 percent.

Takayam city

At the time of the policy changes the population of Takayam city was 67,000. The city introduced a two tier system in 1992. The system involved a number of free collections followed by per unit charges for additional waste bags. However, in the initial period after the introduction of the system the volume of residual waste increased. Over time the city reduced the number of free bags, and raised the price of additional bags. Between 1992 and 2002 the city also introduced a sequence of measures related to increasing the recycling rate. Compared to 1991, by 2004 the:

1. waste generated per capita per day waste had fallen by 15 percent
2. waste incinerated/landfilled per capita per day waste had fallen by 22 percent
3. recycling rate had increased by almost 10 percentage points

Oume city

In 1998 Oume city (population 140,000) introduced a single tier pricing scheme, along with an improved recycling service. Following the introduction of unit pricing there was:

1. a reduction in total per capita per day waste generated (including recyclables) of 11 percent
2. a reduction in per capita per day residual waste incinerated/landfilled of 19 percent
3. an increase in the recycling rate from 7 percent to 16 percent.

Nagoya city

In 1999 Nagoya city (population 2.2M) introduced a single tier pricing scheme for commercial waste only, along with improved recycling services for residents. At the time of the change, due to difficulty securing residual waste sites, there was a heighted sense of the need for change to waste management practice. Following the changes there was:

1. a reduction in total per capita per day waste generated (including recyclables) of 8 percent
2. a reduction in per capita per day residual waste incinerated/landfilled of 14 percent
3. an increase in the recycling rate from 13 percent to 27 percent.

In the following years the volume of waste sent to landfilled/incinerated per capita has continued to fall, and the recycling rate has continued to increase.

Summary

Pay-As-You-Throw waste management systems appear to be a mechanism that helps transmit a clear price signal to residential customers. When faced with a clear price signal customers respond by reducing waste. PAYT systems can therefore effect modest, but real changes in waste generation. Although complementary measure were, in each case also implemented at the same time as the PAYT system, that combined residual and recyclable waste generation fell in each case suggests that there is a pure price effect.

4.5.2 Construction and demolition waste

In 2002 Japan implemented a Construction Material Recycling Law. The law applied to the following cases:

- Demolition of a building that has more the 80m^2 of total floor space
- Construction of a building with more than 500m^2 of total floor space

- Renovation work where the contracting fee is at least ¥100 M
- Civil engineering work where the contracting fee is at least ¥5 M.

The law makes it compulsory to recycle certain materials, including concrete, mixed concrete and iron, wood, and asphalt. There are substantial penalties for failing to comply with the recycling requirements. By 2003 the recycling rate in the C&D stream was 98 percent for concrete, 99 percent for Asphalt, and 90 percent for wood (Tam 2009). In the mid 1990s recycling rates for all these materials were substantially lower.

4.5.3 General summary and implications for Western Australia

In the municipal waste sector, if a clear price signal can be provided to consumers, it is possible that they will respond, at least initially.

In the construction and demolition sector, almost complete recycling of some material can be achieved through the implementation of demolition and construction licence conditions that apply at a relatively modest scale.

The need to demonstrate compliance with a detailed recycling plan when undertaking works can be a mechanism for increasing the overall level of professionalism in the C&D industry.

Where comprehensive systems are not in place to monitor waste, if there are options to dispose of waste illegally, it is possible for the illegal disposal of waste to take place at an industrial scale.

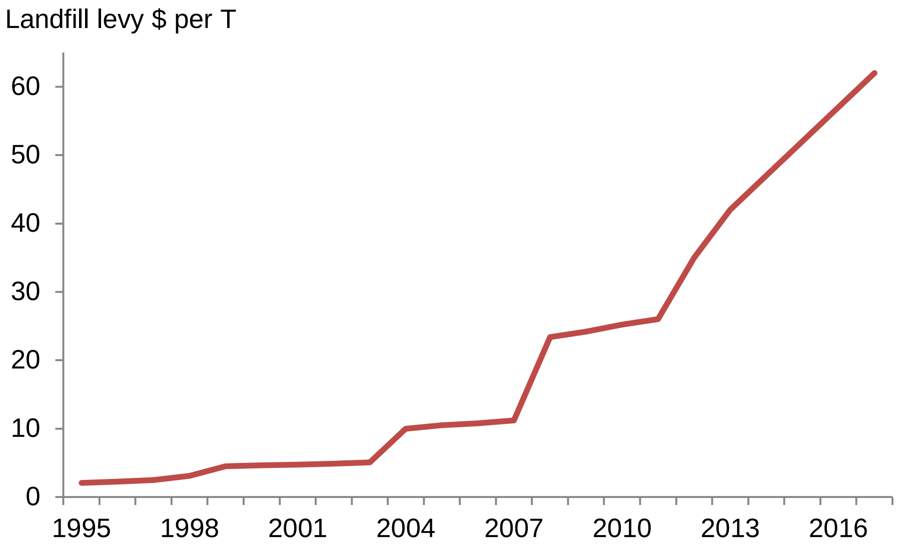
4.6 South Australia

South Australia has had a relatively high landfill diversion rate for some time. The evolution of waste management policy in South Australia can be traced through a number of documents:

- Integrated Waste Strategy for Metropolitan Adelaide: 1996-2015
- Background Paper to South Australia's Waste Strategy 2005-2010
- South Australia's Waste Strategy 2005-2010
- South Australia's Waste Strategy 2011-2015.

Until recently, the landfill levy in South Australia has been relatively modest. Landfill diversion has, however, been substantially above 50 percent since the early 2000s (Zero Waste 2005a). The current landfill levy rate for the metropolitan area is $47 per tonne of waste, and the levy does not distinguish between inert and putrescible streams. The non-metropolitain rate is set at 50 percent of the metropolitan area rate. The EPA in South Australia has announced a number of scheduled levy increases, and by July 2016 the levy will be $62 per tonne. Announcing a forward price schedule for the levy is a recent innovation. Levy revenue is shared between the EPA and Zero Waste SA. Zero Waste SA, in turn, fund a range of programs aimed at increasing diversions rates. The levy has always been indexed to inflation.

Figure 17 **Landfill levy in South Australia**



Note: Years are financial years.
Source: EPA South Australia.

The relative importance of the levy in total disposal costs is now substantial. For example, current subsidy calculations undertaken by Zero Waste SA for the high performance plus kerbside recycling program generally use a landfill disposal cost per ton of municipal waste of around $52 per tonne of waste, net of the landfill levy.

Improvements in diversion rates have varied across different waste streams through time. For example, comparisons of the waste share of material sent to landfill between 1998 and 2004 reveal substantial diversion of C&D material during this period (Zero Waste SA 2005a). As observed in other markets, modest changes in relative prices for waste disposal appear to have a more direct effect on the C&D waste stream.

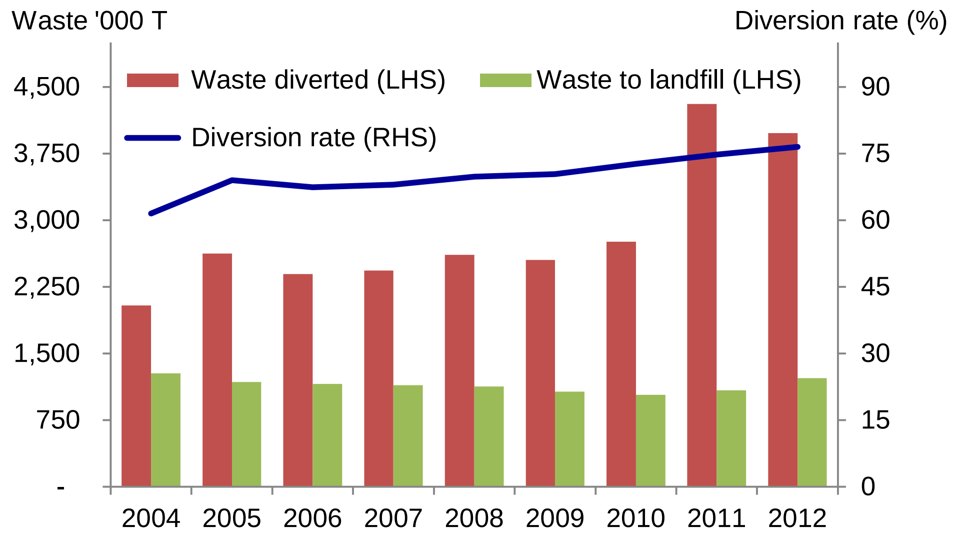
The current strategic plan for waste in South Australia contains diversion targets for Municipal, Commercial and Industrial, and Construction and Demolition waste in 2015 of, respectively, 70 percent, 75 percent, and 90 percent. Actual diversion rates in 2012 for Municipal, Commercial and Industrial, and Construction and Demolition waste, were, respectively, 54 percent, 83 percent, and 80 percent.

Table 13 **Waste diversion in South Australia: by sector**

	2005-06	2006-07	2007-08	2008-09	2009-10	2010-11	2011-12	2011-12 rate
	T	T	T	T	T	T	T	%
Municipal	388,938	408,338	402,677	398,495	340,000	440,000	446,000	53.9
C&I	87,273	870,636	951,355	788,664	1,120,000	1,400,000	1,390,000	82.7
C&D	1,134,571	1,155,154	1,257,182	1,365,043	1,300,000	2,470,000	2,146,000	79.5
Total diversion	2,395,582	2,434,128	2,611,214	2,552,202	2,760,000	4,310,000	3,982,000	76.5
Total to landfill	1,157,925	1,144,429	1,130,000	1,072,000	1,035,000	1,084,000	1,224,000	24.5

Source: Recycling activity statements, various years

Figure 18 **Waste diversion in South Australia**



Note: Financial years

Source: Recycling activity statements, various years

4.6.1 Kerbside recycling incentive payments

Throughout the 1990s and into the early 2000s metropolitan and municipal waste collection systems were variable across local government areas in South Australia. Although, in stylised terms, municipal waste collection systems had largely evolved to consist of a mix of two bin systems and single bin and recycling crate systems.

To encourage higher levels of waste diversion in the local government sector Zero Waste SA introduced a financial incentive for councils to introduce high performing kerbside recycling programs. Funding for the program was made possible due to the revenue generated through the landfill levy.

During the operation of the program the total annual amount of funding available was $11 per household, where receiving the full amount of funding was conditional on meeting a number of objectives, including residual waste reduction and education activities. To meet the criteria to receive funding under the programme the typical waste collection system introduced by metropolitan councils was:

- 140L (or smaller) residual bin collected weekly
- 240L green waste bin collected fortnightly
- 240L dry recyclables collected fortnightly

Important features of the financial incentive system were that: (i) payments were available to councils that had already made improvements; and (ii) the reduction in the size of the residual waste bin. It is notable that the high performance kerbside program has resulted in diversion rates for the Municipal stream of over 50 percent (see Table 13).

For metropolitan councils payments are no longer available under this scheme, but payments are now available to regional councils. As all metropolitan councils now have a 'high performing' kerbside collection service in place, the emphasis for Zero Waste SA has moved to programs targeted at increasing organics recovery through capture of food waste in the composting stream.

The specific strategy used by Zero Waste SA to encourage councils to consider kerbside systems that capture food waste is to subsidise 50 percent of the capital costs involved in providing new infrastructure, and make a contribution of 50 percent towards the net operating cost of the system for the first three years of operation. The calculation of the net cost involves subtracting the landfill cost savings for local governments from the additional operating and composting costs. In broad terms the additional net operating cost of including food waste in the composting stream is around $6 per household.[12] Although, as the landfill levy increases, this net cost will fall.

A key aspect of the kerbside plus program is the target on the reduction in waste to the residual bin, rather than just the increase in the composting stream. For example, if, subsequent to adopting the program, the residual waste stream fell below the target level, but the composting stream did not change much, because of say, an overall reduction in waste generation, local governments are still eligible for the payments. This target on a system that does not result in perverse effects was also evident in the original high performance kerbside program requirement to downsize the residual waste bin to a 140L or smaller bin from a 240L bin.

Organics diversion in South Australia is documented in Table 14. As can be seen, there is a substantial increase in organics recycling around 2004. This increase in organics recycling occurs at the same time as the transition to three bin systems across the metropolitan local government sector. There is a second major increase in organics recycling around 2010. This increase is due to a reporting change to capture information on industry organics, largely meat processing and wine production waste, where some form of reprocessing takes place that sees material diverted from landfill (Zero Waste 2012).

Table 14 **Organics recycling in South Australia through time**

Category	2003-04	2004-05	2005-06	2006-07	2007-08	2008-09	2009-10	2010-11	2011-12
	T	T	T	T	T	T	T	T	T
Food organics	-	10,540	6,005	3,981	5,796	4,820	5,800	4,400	5,600
Garden organics	130,100	188,610	222,499	209,725	202,397	203,558	220,000	230,000	212,000
Timber	116,700	300,980	255,728	275,385	241,387	254,866	262,000	280,000	281,000
Other organics	-	89,790	81,625	82,636	79,359	41,666	148,000	440,000	403,000
Total	246,800	589,920	565,857	571,727	528,939	504,910	635,800	954,400	901,600

Note:

Source: Recycling activity statements, various years

4.6.2 Other observations

Zero Waste SA has commissioned a number of studies into economic aspects of waste management, including cost-benefit analysis of different policies. It is unreasonable to expect agreement across all professional economists about the assumptions embedded in applied economic analysis, but research that makes the economic case for waste diversion policies may be seen as providing Zero Waste SA with a sound basis for the waste management policies advocated by the organisation. Further, despite any concerns about individual modelling assumptions, subjecting different policy combinations to cost-benefit analysis provides a consistent ranking of policy effectiveness. This in turn provides a sound, clearly documented reference point should any questions arise about specific policies.

Zero Waste SA has been active in funding research organisations and trials for new programs. Examples of this kind of activity include the Bio Basket and Kitchen Caddy trials

[12] The key metric for the obtaining payments from Zero Waste is the additional improvement in the diversion of waste achieved. Where actual diversion rate changes are lower than expected partial subsidy payments are made.

for capturing food waste, and the long term funding commitment to a research program with the University of South Australia (Centre for Sustainable Design).

4.6.3 General summary and implications for Western Australia

Zero Waste SA has a clearly documented economic case for pursuing higher recycling rates, and has subjected different policy options to formal analysis to determine which policy combinations are the most cost effective, where cost effectiveness measures include estimates of environmental impacts as well as pure financial costs.

The introduction of a three bin system made an appreciable impact on the amount of organic waste diverted from landfill. The three bin system results in diversion rates that would meet diversion targets in Western Australia.

Providing incentive payments to local government can be a very effective mechanism for driving change. Further, the design of incentive payments can be such that they evolve to continually drive ever higher diversion rates.

Careful consideration of the overall objective is an important element of all incentive programs.

4.7 Victoria

In Victoria amendments focused on reducing waste in general, and the amount of waste going to landfill in particular, were made to the *Environmental Protection Act 1970*, in 1992. A core element in the *Environmental Protection Act 1970* is the inclusion of a waste hierarchy. Specifically, Section 1.I of the *Environmental Protection Act 1970* adopts the following waste hierarchy:

> 1I. Principle of wastes hierarchy
>
> Wastes should be managed in accordance with the following order of preference-
>
> (a) avoidance;
> (b) re-use;
> (c) re-cycling;
> (d) recovery of energy;
> (e) treatment;
> (f) containment;
> (g) disposal.

In a 2009 review of the impact of environmental regulation on business in Victoria, the Victorian Competition and Efficiency Commission makes a compelling case that the waste hierarchy:

i) is an economically inefficient approach that fails to recognise the most appropriate means of disposing of different types of waste

ii) fails to recognise the changing nature of costs for different technologies, and

iii) could be seen as inconsistent with other parts of the Act (Victorian Competition and Efficiency Commission 2009).

The potential internal inconsistency in the Act was not explicitly identified, but a lay reading of the Act suggests the waste hierarchy is inconsistent with Section 1B(3) which requires measures to be cost-effective. As per the Act in February 2013, this inconsistency remained.

The 1992 revisions to the Act were associated with a government goal of reducing the volume of material sent to landfill by 50 percent. Progress towards achieving this objective was reviewed by the Auditor General in 2000. Some of the relevant findings of the review included:

- There was a lack of data for measuring progress against the objective
- The introduction of kerbside recycling had contributed to increased recycling rates, but that it was generally more costly to recycle than landfill
- Regional waste management plans were not well developed or costed
- Councils generally did not have direct information on resident preferences for recycling, and residents were generally not well informed about the cost of waste management and recycling services (Victorian Auditor General 2000).

Subsequent to the review, the Victorian Government developed a Towards Zero Waste strategy. The strategy was adopted in 2005 with targets to 2014. The key targets in the strategy were:

- A 1.5 million tonne reduction in the projected quantity of solid waste generated by 2014.
- 75 percent (by weight) of solid waste recovered for reuse, recycling and/or energy generation by 2014, with an interim 2008–09 target of 60 percent.
- 65 percent (by weight) of municipal solid waste recovered for reuse, recycling and/or energy generation by 2014 with an interim 2008–09 target of 45 percent.
- A 25 percent improvement, from 2003 levels, in littering behaviours by 2014.

In 2011, progress towards meeting these objectives, as well as the process of setting the objectives was reviewed by the Victorian Auditor General. In reviewing these objectives the Victorian Auditor General identified a number of serious problems. Core issues included that: (i) Sustainability Victoria and the Department of Sustainability could provide no evidence to demonstrate the targets were underpinned by any sort of robust analysis; (ii) consideration was not given to implementation costs; and (iii) a number of assumptions regarding the cost effectiveness of recycling technologies, and hence their commercial viability were incorrect (Victorian Auditor General 2011).

In response to the various reviews of the strategy, and the need to develop post 2014 targets, in 2012 the Victorian Government published a waste policy discussion paper (Department of Sustainability and Environment 2012a). Subsequent to receiving input into a review process, a draft waste policy paper was released (Department of Sustainability and Environment 2012b). The vision articulated in the draft policy is:

> Victoria's integrated, statewide waste management and resource recovery system provides an essential community service that protects the environment and public health, maximises the productive value of resources and minimises long-term costs to industry, government and households.

It is notable that cost minimisation is recognised in the waste management vision for Victoria.

More generally the principles to apply for waste management policy in Victoria are: (i) transparent decision making; (ii) evidence based decision making; (iii) recognition that waste management and resource recovery are an integrated system; and (iv) proportionate and appropriate government intervention (Department of Sustainability and Environment 2012b).

In recognition of the criticism that has been levelled at State governments that adopt specific waste targets (Victorian Auditor-General 2011; Productivity Commission 2006) the draft policy does not include explicit diversion targets. Rather it proposes a wide range of monitoring options. A number of metrics are still to be developed, but the four metrics that have been decided are: the rate of MSW, C&D, C&I, and total waste diverted from landfill (Department of Sustainability and Environment 2012b). The draft strategy also notes that the landfill levy will continue to be a core element of the overall waste policy framework.

4.7.1 Landfill level and recycling in Victoria

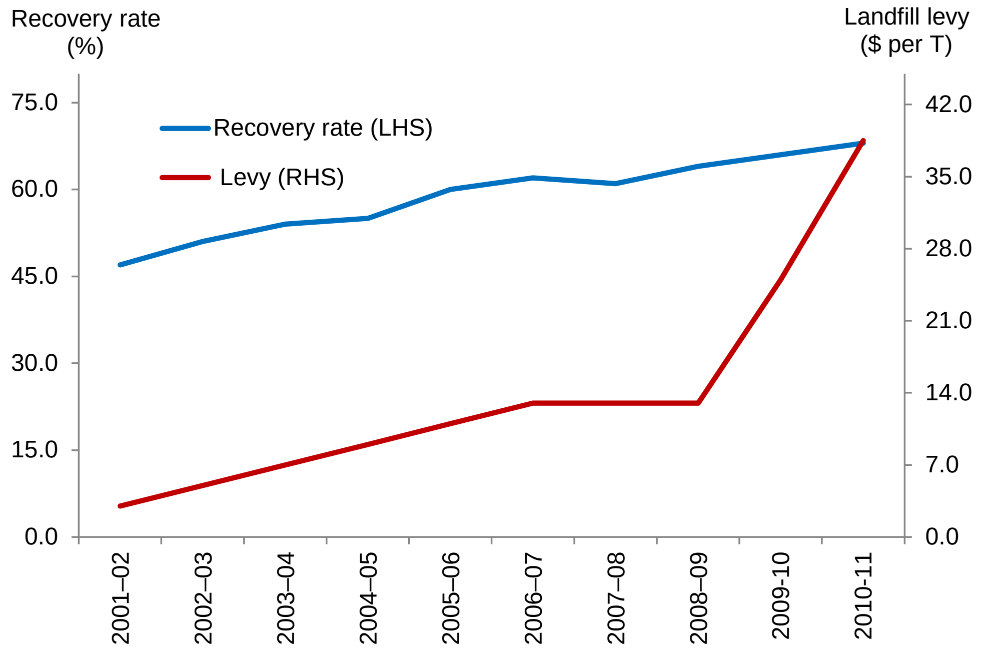
Historic and future landfill levy rates in Victoria are shown in Table 15; and the relationship between the recovery rate and the landfill levy is shown in Figure 19. As can be seen from Table 15, when first introduced the landfill levy was modest, but has increased substantially over recent years. In Figure 19 the sharp increase in the levy over recent years can be clearly seen. It is notable that despite the significant increase in the levy rate, the rate of change in the recovery rate seems approximately constant.

Table 15 **Landfill levy rates: Victoria**

	Regional		Metropolitan	
	Municipal	Industrial	Municipal	Industrial
	$ per T	$ per T	$ per T	$ per T
2001–2	2.00	2.00	4.00	4.00
2002–3	2.00	3.00	4.00	5.00
2003–4	3.00	5.00	5.00	7.00
2004–5	4.00	7.00	6.00	9.00
2005–6	5.00	9.00	7.00	11.00
2006–7	6.00	11.00	8.00	13.00
2007–8	7.00	13.00	9.00	15.00
2008–9	7.00	13.00	9.00	15.00
2009–10	7.00	13.00	9.00	15.00
2010–11	15.00	25.00	30.00	30.00
2011–12	22.00	38.50	44.00	44.00
2012-13	24.20	42.40	48.40	48.40
2013-14	26.60	46.60	53.20	53.20
2014-15	29.30	51.30	58.50	58.50

Data source: Southern Councils Group (2012).

Figure 19 **Victorian recovery rate and metropolitan landfill levy (2001-11)**



Data source: Sustainability Victoria (2012).

However, when interpreting the figure recall the discussion presented earlier about requiring models that explicitly account for a lagged effect when the price of landfill changes. Within such a framework we would not expect to see a change in the recovery rate following a change in the levy for a number of years.

It was noted by the Victorian Auditor-General that the reason for Victoria exceeding the aggregate by weight diversion target in 2008-09 was a high rate of diversion of C&D waste (Victorian Auditor-General 2011). At the time of the review, the landfill levy that applied to this waste stream was $14 per tonne. It was subsequently noted in Department of Sustainability and Environment (2012b) that the reason for the high diversion rate was that recycled C&D waste was a low cost product in high demand.

Details on recycling in Victoria are contained in the annual Victorian Recycling Industry Surveys. The most recent data is for the 2010-11 year, and between 2001-02 and 2010-11 the recovery rate from solid waste in Victoria increased from 47 percent to 68 percent (Department of Sustainability Victoria 2011).

Table 16 contains information on the total amount of waste recovered in Victoria, by waste type and waste stream. For example, by reading down column (2) and (3) of Table 16 it can be seen that in 2010-11, for C&D waste 4.2 million tonnes was recovered and that this represented 52 percent of the total volume of waste recovered. The contribution of the three main waste streams to this 4.2 million tonnes is then shown in columns (3), (4), and (5). The final column of the table provides details on the relatively small volume of recovered waste that was not generated in Victoria.

Table 16 **Recycling in Victoria by waste stream (2010-11)**

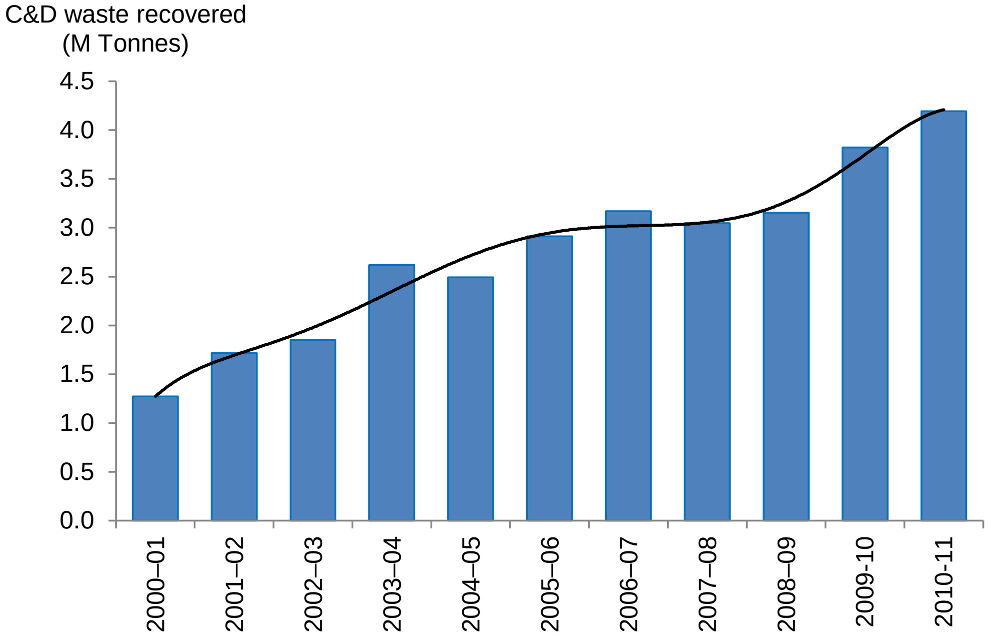
Waste type	Total	Share	MSW	C&I	C&D	Imports
	T (‘000)	(%)	T (‘000)	T (‘000)	T (‘000)	T (‘000)
Construction & demolition	4,193	51.97	177	55	3,960	0
Metals	1,389	17.22	232	1,065	91	55
Paper/ cardboard	1,212	15.02	315	897	0	79
Organics	871	10.80	377	445	47	40
Glass	195	2.42	115	79	0	39
Plastic	146	1.81	55	86	3	0
Rubber	54	0.67	2	51	<1	6
Textile	4	0.05	3	<1	0	0
Total	8,068	100.00	1,281	2,683	4,102	221

Data source: Sustainability Victoria (2012).

The change in the volume of C&D waste recovered through time is shown in Figure 19. The volume of material recovered fluctuates with construction and demolition activity but over the past decade there has been a significant increase in the amount of material recovered from this waste stream.

Again, at this point it is worth recalling the earlier discussion on the role of substitutes when considering changes in resource recovery rates. As the range of economically viable substitute options for the use of C&D waste is greater than for other waste streams we observe greater responsiveness.

Figure 20 **Volume of C&D waste recovered (2000-11)**



Data source: Sustainability Victoria (2012).

The relative importance of the different waste streams within C&D, and their evolution through time is shown in Table 17. As can be seen, recycled concrete is the dominate component of the overall C&D waste stream.

Table 17 **Detailed construction and demolition recovery volumes (2000-11)**

Year	Asphalt	Bricks	Concrete	Mixed C&D	Plaster-board	Rock/ stone	Soil & Sand	Total
	T ('000)	T ('000)	T ('000)	T ('000)	T ('000)	T ('000)	T ('000)	T ('000)
2000–01	68	318	811	-	4	56	16	1,273
2001–02	65	293	942	-	8	359	49	1,716
2002–03	84	250	1,161	-	21	293	42	1,852
2003–04	170	425	1,525	-	22	428	49	2,618
2004–05	162	395	1,477	-	24	367	68	2,492
2005–06	139	385	1,734	-	27	419	209	2,913
2006–07	190	438	1,695	81	22	505	239	3,170
2007–08	152	293	1,717	111	33	668	72	3,047
2008–09	226	244	1,731	91	37	656	170	3,155
2009-10	196	518	2,438	81	27	452	108	3,823
2010-11	223	497	2,174	167	32	980	118	4,193

Data source: Sustainability Victoria (2012).

Although it might be thought that the higher the levy the greater the likelihood of recycling, the link between higher landfill levy rates and recycling is complex. For example, in the metal recycling industry, due to subsequent higher cost of waste disposal, the net margin per tonne of material recovered falls slightly as the landfill levy increase (Marsden Jacob Associates 2007).

More generally, the issue raised here is relevant for any reprocessing/ recycling process that also generates significant volumes of residual waste that must subsequently be landfilled.

4.7.2 Municipal waste

Price responsive of the municipal waste sector in Melbourne has been considered and the results found a price elasticity of -.02 that was not statistically different from zero (Pickin 2008). Other findings in the same study found that introducing bin capacity reductions resulted in a reduction of waste of around 7 percent, and the introduction of co-mingled recyclables resulted in an increase in recycling of 6 percent. These findings suggest that for municipal waste, where there price signal can be weak, regulation based approaches can be more effective than price based approaches.

4.7.3 General summary and implications for Western Australia

Victorian policy development has benefited from a number of formal reviews. These reviews have identified a number of important features of waste management policies that are relevant for Western Australia. These include:

- A need for clear documentation and analysis to support decisions
- Good policy is consistent with the avoidance of explicit diversion targets that are likely to result in the imposition of excess costs on society
- The waste hierarchy is inconsistent with the principle of cost-effectiveness.

Other aspects of the review that are relevant for Western Australia include the impact of landfill levy rates on recycling and or reuse options that include significant volumes or residual waste that must be sent to landfill, and that non-price mechanisms can be effective at lifting diversion rates.

4.8 New South Wales

4.8.1 NSW Summary

The landfill levy in NSW applies to different regions. The *Protection of the Environment Operations (Waste) Regulation 2005* provides definitions of the relevant regions for the application of the waste levy as follows:

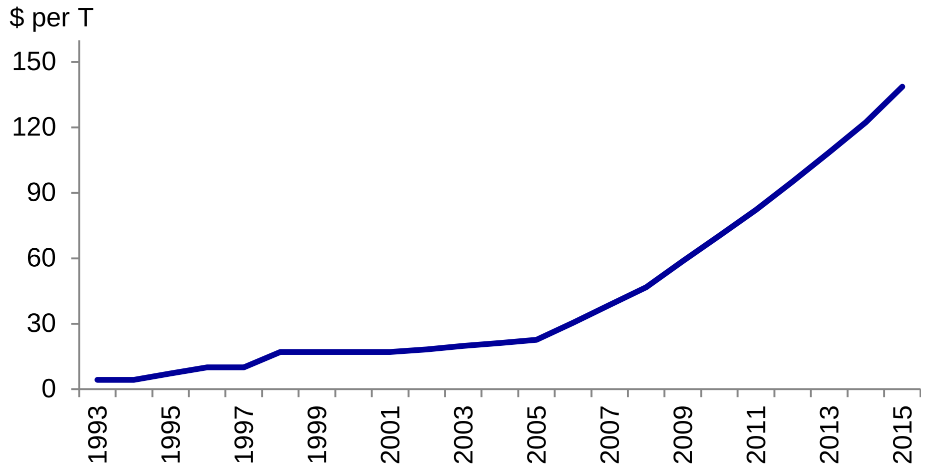
- The Sydney Metropolitan Area (SMA) is defined as the local government areas of: Ashfield, Auburn, Bankstown, Baulkham Hills, Blacktown, Botany, Burwood, Camden, Campbelltown, Canada Bay, Canterbury, Fairfield, Holroyd, Hornsby, Hunters Hill, Hurstville, Kogarah, Ku-ring-gai, Lane Cove, Leichhardt, Liverpool, Manly, Marrickville, Mosman, North Sydney, Parramatta, Penrith, Pittwater, Randwick, Rockdale, Ryde, Strathfield, Sutherland, Sydney, Warringah, Waverley, Willoughby and Woollahra.
- The Extended Regulated Area (ERA) is defined as the local government areas of: Cessnock, Gosford, Hawkesbury, Kiama, Lake Macquarie, Maitland, Newcastle, Port Stephens, Shellharbour, Shoalhaven, Wingecarribee, Wollongong and Wyong, and
- The Regional Regulated Area (RRA) is defined as the local government areas of: Ballina, Bellingen, Blue Mountains City, Byron, Clarence Valley, Coffs Harbour City, Dungog, Gloucester, Great Lakes, Greater Taree City, Kempsey, Kyogle, Lismore City, Muswellbrook, Nambucca, Port Macquarie-Hastings, Richmond Valley, Singleton, Tweed, Upper Hunter Shire and Wollondilly.

In NSW there are separate levy rates for some classes of waste, such as liquid transferable waste and coal washery rejects. Here the discussion here is focused on what is understood to be material in the three generally recognised waste streams: MSW, C&D, and C&I.

A modest landfill levy of $0.51 per tonne was first introduced in NSW in 1971. The levy was then periodically adjusted. Starting in 2006, there have been substantial, staged increases in the real landfill levy rate in the SMA and the ERA. Starting in 2010 a landfill levy was introduced in the RRA.

In real 2006 dollar terms the levy price increases described in the regulations map out a price path for the SMA and ERA that would see the levy reach around $110 per tonne in 2016. The formula detailed in the regulations implies that the real value of the levy would then be maintained through indexation. In nominal dollar terms this means a levy of around $120 per tonne for these areas in 2016. In nominal dollar terms the landfill levy for the RRA is set to reach around $75 dollars in 2016, before then being indexed at the inflation rate.

Figure 21 **Sydney metropolitan area levy rate through time**



Data source: Southern Councils Group (2012).

In 2012 the NSW Environment Protection Authority commissioned a review of the landfill levy. The review findings and the response of the NSW government are briefly summarised below. The following abridged summary is based on information provided on the department website. The format, in terms of sub-headings also follows that of the department.[13] The summary is useful as it highlights factors that have emerged during a period of substantial increase in a landfill levy in an Australian jurisdiction.

Household organic waste

R1. Develop best practice guidelines to encourage source separation at the household level (accepted).

R2. Provide non-contestable funding to local government to allow appropriate, locally relevant education and information material and programs to be developed that will assist with improving recycling rates (accepted, $70M provided over four years).

R3. Use levy funding to provide a contestable pool of funds to assist with the development of recovery infrastructure (accepted, total funding in two programs of $130M over four years).

Recycling

R4. Use levy revenue to support C&I recycling initiatives (accepted, $15M over five years).

[13] Full details available: www.environment.nsw.gov.au/waste/levyrecommendations.htm [accessed 21 May 2013].

R5. Implement programs specifically targeting a reduction in metal and paper recyclers' residual waste (accepted, and will be a focus of the $15M recycling fund, additional concessions also provided to metal and paper recyclers).

Use of levy funds

R6. Allocate a greater portion of levy revenue to waste reduction programs (rejected).

R7. Local governments should be able to access funding for relevant initiatives (accepted, $70M in non-contestable funding provided over four years).

Waste infrastructure

R8. A regional waste infrastructure strategy should be developed and funding provided to support installation of the required infrastructure (accepted, $9M to support planning and applications to the infrastructure fund; $13M for non-levy regional areas; and $7M for upgrading facilities in rural and regional areas).

R9. Establish an expert panel to facilitate new infrastructure planning, procurement, and delivery (accepted, operationalized through the *Environmental Trust Act 1998*).

Illegal dumping

R10. Develop and implement a state-wide illegal dumping strategy (accepted, EPA developed strategy, $58M in funding).

R11. Increase community involvement in anti-illegal dumping activities (accepted, EPA to engage with relevant community groups and local government).

R12. Trial drop-off collection points for problem waste. (accepted, $70M funding over five years).

R13. Provide rebate to small renovators for the levy where the renovator attends an approved waste education session. (accepted, EPA co-ordinating with relevant local governments).

Energy from waste policy

R14. Develop an energy from waste policy (accepted, EPA has a paper out for consultation).

Carbon price

R15. The levy should not be adjusted in-light of a carbon price (accepted).

Application of the levy

R16. A uniform rate of the levy should apply across NSW (rejected, further consultation ongoing).

Landfill operation

R17. A deduction (10 percent) should be allowed for a portion of clean fill accepted (accepted, regulation amendment to be progressed).

4.8.2 General summary and implications for Western Australia

In 2000, when the landfill levy was around $17 per tonne of waste the recycling rates were around 26 percent for MSW, 28 percent for C&I, and around 65 percent for C&D. This highlights the different responses of each waste stream to higher landfilling costs and the relative responsiveness of the C&D waste stream. Note that this responsiveness is in turn driven by the greater substitution opportunities to landfilling available for C&D waste, relative to C&I waste and MSW.

The MSW stream has been the least responsive to price changes, and has the lowest recycling rate, increasing from 26 percent in 2000 to 44 percent in 2009-09.

The 2014 targets for each waste stream in NSW are broadly comparable with the 2020 targets in Western Australia. In 2014 the metro area levy will be around $110 per tonne, and with the levy at this level diversion targets for MSW (66 percent) and C&I waste (63 percent) are unlikely to be met without the provision of substantial funds to assist with: (i) new waste infrastructure investment and local council initiatives, and (ii) government infrastructure planning leadership. The quantum of money subsequently allocated to support LGAs and new infrastructure investment is especially notable.

As the landfill levy has increased a number of local councils submitted evidence to the review that illegal dumping of waste has become more prominent. Actions targeted at small renovators may, however, be a mitigation strategy for targeting illegal dumping.

References

References: Netherlands

Bartelings, H., Kuik, O., & Linderhof, V. (2005). Effectiveness of Landfill Taxation. *Amsterdam: Institute for Environmental Studies*, Vrije Universiteit, Netherlands.

Dijkgraaf, E., & Vollebergh, H. R. (2004). Burn or bury? A social cost comparison of final waste disposal methods. *Ecological Economics*, *50*(3), 233-247.

Fischer, C., Lehner M. & McKinnon, D.L (2012). Overview of the use of Landfill Taxes in Europe. Prepared for the European Environment Agency.

Hendricks, F. & Janssen, G.M.T. (2001). Reuse of Construction and Demolition Waste in the Netherlands for Road Construction. *Herron*, 46(2), 109-117.

Hill, J., Bégin, A-E. & Shaw B. (2002). Creative Policy Packages for Waste: Lessons for the UK, Green Alliance, London.

Kemp, R. (2007). An Example of a "Managed Transition": The Transformation of the Waste Management Subsystem in the Netherlands (1960–2000). In *Innovations Towards Sustainability* (pp. 87-94). Physica-Verlag HD.

Mulder, E., de Jong, T. P., & Feenstra, L. (2007). Closed Cycle Construction: An integrated process for the separation and reuse of C&D waste. *Waste Management*, 27(10), 1408-1415.

Milios, L. (2013). Municipal Waste Management in the Netherlands. Prepared for the European Environment Agency.

Verbong, G., & Raven, R. (2004). Dung, sludge, and landfill: Biogas technology in the Netherlands, 1970-2000. *Technology and culture*, *45*(3), 519-539.

von Dijk, K., Boedianto, P., te Dorsthorst, B.J.H., & Kowalczyk, T. (2001). Strategy for Reuse of Construction and Demolition Waste: Role of Authorities. *Herron*, 46(2), 89-94.

References: Canada

Ontario Government (2008). A Guide to Waste Audits and Waste Reduction Work Plans for Construction and Demolition Project, Ontario Government, Ontario.

City of Markham 2012 The best of the best: Markham's road map to 80 percent diversion, City of Markham.

Anon 2013, 10-point strategy, residents take Markham, Ont past 80% diversion rate, *Solid Waste and Recycling*, 19 August 2013.

Statistics Canada (2012) Waste Management Industry Survey: Business and Government Sectors, catalogue no. 16F0023X, Ottawa.

World Bank (2012) *What a Waste: A Global Review of Solid Waste Management*, Annex L, World Bank.

Krausz, R. 2012, All for Naught? A critical study of zero waste to landfill initiatives, PhD thesis, University of Lincon.

References: UK

DEFRA (2011) Government Review of Waste Policy in England 2011, Department for Environment, Food and Rural Affairs, London, UK.

DEFRA (2012) Local Authority Collected Waste Management Statistics for England – Final Annual Results 2011/12, Release Date 8th November 2012, Department for Environment, Food and Rural Affairs, London, UK.

DEFRA (2013) Statistics on Local Authority Waste Management in England – Provisional Results October to December 2012/13, Release Date 8th August 2013, Department for Environment, Food and Rural Affairs, London, UK

EEA (2008) Effectiveness of environmental taxes and charges for managing sand, gravel and rock extraction in selected EU countries, European Environment Agency, Copenhagen, Denmark.

Environment Agency (2011) Report on the Landfill Allowance Trading Scheme 2011/12, Bristol, UK.

Jacobs (2011) Commercial and Industrial Waste Survey 2009, Report prepared for Department for Environment, Food and Rural Affairs.

Martin, A., & Scott, I. (2003). The effectiveness of the UK landfill tax. *Journal of environmental planning and management*, *46*(5), 673-689.

WRAP (2012) Gate Fees Report, 2012:Comparing the cost of alternative waste treatment options.

WRAP (2008) Gate Fees Report, 2008:Comparing the cost of alternative waste treatment options.

References: Italy

D'Alisa, G., Burgalassi, D., Healy, H., & Walter, M. (2010). Conflict in Campania: Waste emergency or crisis of democracy. *Ecological economics*, *70*(2), 239-249.

Mazzanti, M., Montini, A., & Nicolli, F. (2009). The dynamics of landfill diversion: Economic drivers, policy factors and spatial issues: Evidence from Italy using provincial panel data. *Resources, Conservation and Recycling*, *54*(1), 53-61.

References: Japan

Nakamura, Y. (2007). Waste Management and Recycling Business in the United States and Japan. *USJP Occasional Paper*, 07-09.

Nakayama, H., Tsuchida, D., & Shimaoka, T. (2012). Estimation of cost reduction and increase for the final disposal associated with the categorization of inert waste landfills in Japan. *Waste Management & Research*, *30*(2), 190-199.

Sakai, S., Ikematsu, T., Hirai, Y., & Yoshida, H. (2008). Unit-charging programs for municipal solid waste in Japan. *Waste management*, *28*(12), 2815-2825.

Tam, V. W. (2009). Comparing the implementation of concrete recycling in the Australian and Japanese construction industries. *Journal of Cleaner Production*, *17*(7), 688-702.

Tanaka, M. (1999). Recent trends in recycling activities and waste management in Japan. *Journal of Material Cycles and Waste Management*, *1*(1), 10-16.

References: South Australia

Integrated Waste Strategy for Metropolitan Adelaide: 1996-2015.

Zero Waste SA (2005a) Background Paper to South Australia's Waste Strategy 2005-2010.

Zero Waste SA (2005b) South Australia's Waste Strategy 2005-2010.

Zero Waste SA (2007) SA Recycling Activity Report 2005-06.

Zero Waste SA (2008) SA Recycling Activity Report 2006-07.

Zero Waste SA (2009) SA Recycling Activity Report 2007-08.

Zero Waste SA (2010) SA Recycling Activity Report 2008-09.

Zero Waste SA (2011a) South Australia's Waste Strategy 2011-2015.

Zero Waste SA (2011b) SA Recycling Activity Report 2009-10.

Zero Waste SA (2012) SA Recycling Activity Report 2010-11.

Zero Waste SA (2013) SA Recycling Activity Report 2011-12.

References: Victoria

Department of Sustainability Victoria (2011). Victorian Recycling Industry Annual Survey 2010-11.

Department of Sustainability and Environment (2012a) Waste Policy Review Discussion Paper.

Department of Sustainability and Environment (2012a) Draft Victorian Waste and Resource Recovery Policy.

Marsden Jacob Associates and Warnken ISE (2007) Impact of Landfill Levy on the Steel Recycling Sector in Victoria. Prepared for EPA Victoria.

Productivity Commission (2006). *Waste Management, Report no. 38.* Canberra, Australia.

Pickin, J. (2008). Unit pricing of household garbage in Melbourne: improving welfare, reducing garbage, or neither?. *Waste Management & Research*, *26*(6), 508-514.

Southern Councils Group (2012). Southern Councils Group Submission to the NSW Office of Environment and Heritage. Prepared for the Independent Review of the Waste and Environment Levy.

Victorian Competition and Efficiency Commission (2009). A Sustainable Future for Victoria: Getting Environmental Regulation Right, Final Report.

Victorian Auditor General (2000). Reducing Landfill: Waste Management by Municipal Councils.

Victorian Auditor-General (2011). Municipal Solid Waste Management.

References: New South Wales

KPMG (2012). Review of the NSW Waste and Environmental Levy, Prepared for the NSW Environment Protection Authority.

Southern Councils Group (2012). Southern Councils Group Submission to the NSW Office of Environment and Heritage. Prepared for the Independent Review of the Waste and Environment Levy.

Appendix A Models for estimating the landfill own-price elasticity

A.1 Deriving the Autoregressive distributed lag model

The ADL model can be difficult to interpret, so it is worth explaining the approach as a combination of two underlying sub-models that have a more natural interpretation: the partial adjustment model and the distributed lag model.

A.1.1 Partial adjustment model

The first sub-model to consider is the partial adjustment model, and in the context of estimating the landfill own-price elasticity the model can be understood as follows.

We can operationalize the partial adjustment model in terms of observable data to obtain an estimate of the long-run response in a number of different ways. The most common approach is to assume that the adjustment process towards the target level of waste to landfill depends on the difference between the unobservable target level for waste volumes in the current period and the observable actual level of waste volumes to landfill in the previous period.

In its simplest form we would have the following expression for the theoretical unobservable relationship between the target level of waste to landfill and the price of landfill:

$$Q_t^* = a + bP_t + e_t. \tag{8}$$

In the above expression Q_t^* denotes the target level of waste to landfill at time *t*, given landfill price P_t, *a* and *b* are parameters to be estimated, and e_t is a zero mean random error term that accounts for the various small random shocks that are always present in the system, but not explicitly accounted for in the model. As it is the theoretical target level of waste to landfill we do not actually ever observe Q_t^*, rather, what we observe is Q_t, the actual level of waste going to landfill in each time period.

Assuming the adjustment process depends on the difference between the volume of waste sent to landfill in the previous period and the target level of waste sent to landfill this period gives the formal expression:

$$Q_t - Q_{t-1} = \gamma[Q_t^* - Q_{t-1}], \tag{9}$$

where γ is the speed of adjustment parameter. The above expression say that the observed change in the quantity of material sent to landfill is proportional to the difference between the quantity sent to landfill in the previous period and the target level of waste to landfill this period. Values for γ are bounded by zero, which implies no adjustment, and one, which implies complete adjustment. That γ will lie somewhere between no adjustment and complete adjustment is the result that gives rise to the name "partial adjustment model". This relationship can also be seen by rearranging equation as:

$$Q_t = Q_t^*\gamma + Q_{t-1}[1 - \gamma], \tag{10}$$

which says that the quantity we observe going to landfill today is a weighted average of the amount we sent to landfill in the previous period, and the optimal amount we would like to send to landfill today.

To obtain a model completely in terms of observable information note that equation (10) can also be written as:

$$Q_t^* = \frac{Q_t}{\gamma} + \frac{Q_{t-1}}{\gamma} - Q_{t-1}. \tag{11}$$

The right hand side of equation (11) can then be used to replace the left hand side of equation (8) to give:

$$\frac{Q_t}{\gamma} + \frac{Q_{t-1}}{\gamma} - Q_{t-1} = a + bP_t \ + e_t. \tag{12}$$

Following simplification, the expression collapses to:

$$Q_t \ = \gamma a + \gamma b P_t \ + (1-\gamma)Q_{t-1} + \gamma e_t. \tag{13}$$

The term γb in equation (13) then describes the short-run effect of price on the quantity of material sent to landfill. As, in the long-run, we move from one equilibrium point to another, to find the long-run effect we substitute into equation (13) long-run equilibrium values. Using Q' and P' to denote long-run equilibrium values this substitution first gives:

$$Q' = \gamma a + \gamma b P' + (1-\gamma)Q', \tag{14}$$

which can then be rearranged to give:

$$Q' = a + bP'. \tag{15}$$

Equation (15) says that the long-run impact of the price of landfill on the volume of waste sent to landfill is given by the b term.

So, for the partial adjustment model the underlying model estimated is:

$$Q_t = \gamma a + \gamma b P_t + (1-\gamma)Q_{t-1} + \gamma e_t, \tag{16}$$

and the actual model regression output (estimated via least squares) is of the form:

$$Q_t = \alpha + \beta_1 P_t + \beta_2 Q_{t-1} + e_t'. \tag{17}$$

Long-run and short run impacts are then determined via a process of matching terms between the theoretical model (equation (16)) and the estimated model output (equation (17)). Through this matching process it can be seen that we have $\beta_1 = \ \gamma b$, so that in the estimated regression model the β_1 parameter gives the short-run impact of landfill price on the quantity of material sent to landfill. Further, as the short run estimate is estimated directly via the regression model, the standard error associated with the β_1 estimate provides a clear measure of the uncertainty associated with the estimated short-run effect.

As b gives the long-run impact, through matching the estimated model output to the theoretical model output the long-run impact can be found as: $b = \gamma b/\gamma = \beta_1/1 - \beta_2$. That is, the long-run impact is found from the estimated regression model as the ratio of the β_1 term and one minus the β_2 term. Given the long-run impact is found as the ratio of two estimated coefficients a measure of the uncertainty surrounding the estimated long-run effect can be obtained via the delta method approximation.

When the above model is estimated using the log of quantity and price, the parameter values have a direct interpretation as the short-run and long-run own-price elasticity. When the model is estimated in the original price and quantity scale, the short-run and long-run elasticity values are usually evaluated at the sample mean price and quantity level.[14]

A.1.2 Distributed lag model

The second sub-model to consider is the distributed lag model. The distributed lag model says that there can be a delay in responses to a change in the operating environment today. In the context of landfill price responsiveness, the model says what happen today, in terms of price changes, matters, but because we have a slow moving process, what happened in the past also matters for what happens today. The motivation is similar to that outlined for the partial adjustment model.

Using the same notation as for the partial adjustment model, the distributed lag model implies a model of the form:

$$Q_t \ = a + bP_t + cP_{t-1} + e_t. \tag{18}$$

If it was thought appropriate, the model could also include $P_{t-2}, P_{t-3}, \dots, P_{t-k}$ as additional explanatory variables. Although for illustration purposes, here the discussion is restricted to a single period lagged effect.

In equation (18) the b term describes the current period, or short-run impact, and the c term describes the delayed impact. Similar to the partial adjustment model, the total impact, or long-run impact is found by substituting equilibrium values into equation (18) to give:

$$Q' \ = a + (b + c)P'. \tag{19}$$

The long-run effect is therefore the sum of the terms on the current and past price variables. As with the partial adjustment model, if the regression model is specified in log form the values are interpreted directly as elasticities. As the short-run impact is estimated directly, and the estimate of the parameter standard error provides a measure of the uncertainty surrounding the estimate. For the long-run impact a measure of estimate uncertainty can be derived using the delta method.[15]

A.1.3 The Autoregressive Distributed Lag model

With an understanding of the relevant sub-models, and the motivation for the way these models are estimated, it becomes possible to consider the ADL model. The specific ADL model presented here is known as an ADL(1,1) model, and based on the literature review the ADL(1,1) model is the most flexible model that has been used to estimate the price

---

[14] The partial adjustment model can be criticised as a backward looking model. It might reasonably be argued that waste management organisations are forward looking. A forward looking organisation forms a view about the expected future (relative) price of landfill and then plans accordingly. Such a framework is known as the rational expectation model. However, from a statistical point of view the rational expectations model and the partial adjustment model are statistically equivalent. This means that as an empirical matter it does not matter whether one thinks people are forward looking or backward looking, the empirical model estimated takes the same form.

[15] When several lags of price are introduced as part of the regression model multicollinearity can be a problem. A practical approach to resolve the issue is to use algebraic manipulations to derive estimates of the long-run effect using a combination of variables in first difference form and level form. Such an approach provides an estimate of the long-run effect and a direct estimate of the standard error that is not affected by multicollinearity.

responsiveness of landfill volumes to changes in the price of landfill. The ADL(1,1) model may be thought of as a model that combines the distributed lag model and the partial adjustment model, and is specified as:

$$Q_t = \alpha + \beta_1 P_t + \beta_2 P_{t-2} + \beta_3 Q_{t-1} + e_t. \tag{20}$$

As the model combines the partial adjustment model with the distributed lag model the short-run and long-run effects are derived by working through the same steps. Specifically, in equation (20) the short-run effect is given by β_1, and the long-run effect is found by substituting long-run equilibrium values into equation (20) to give:

$$Q' = \alpha + (\beta_1 + \beta_2)P' + \beta_3 Q', \tag{21}$$

which, after grouping common terms and rearranging gives:

$$Q' = \frac{\alpha}{(1-\beta_3)} + \frac{(\beta_1 + \beta_2)}{(1-\beta_3)} P'. \tag{22}$$

As the term $(\beta_1 + \beta_2)/(1-\beta_3)$ describes the equilibrium change in the quantity of material sent to landfill following a change in the landfill price it describes the long-run impact. Although the expression looks complex, note the expression is simply a composite of the way the long-run effect is derived in both of the sub-models described.[16]

A.2 Consultation

In preparing this report representatives from a wide variety of organisations found time to provide comments and input, and or attended a presentation at the EMRC. The following organisations are thanked for their time:

- City of Nedlands
- City of Canning
- City of Rockingham
- EMRC
- SMRC
- Waste Management Association of Australia
- Eclipse Resources
- Moltoni Group
- Zero Waste SA
- RichGro
- NutraRich
- WALGA
- Transpacific Industries Group
- Veolia
- Capital Recycling

[16] None of the literature reviewed considers the issue of non-stationary data. Non-stationary data is a situation where the mean value of the data series varies through time, and can result in what is known as the spurious regression problem. Note, however, that the traditional ADL model is still valid for trend stationary data and has been shown to generate consistent estimates of long-run effects even with I(1) regressors (Pesaran and Shin 1999). The ADL model can also be reformulated as an Error Correction model for co-integrated data.

- Bowman and Associates
- Geo-Environmental
- Instant Waste
- Various Members of the Department Environmental Regulation
- Amcor
- All Earth Group, and
- AnaeCo
- Perthwaste Green Recycling.