NZIER NEW ZEALAND INSTITUTE OF ECONOMIC RESEARCH

Willingness to pay for six end-of-life products

NZIER report to Ministry for the Environment

24 October 2013

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Key points

NZIER and MWH Global reviewed the end-of-life treatment of six products[1] and investigated the international literature to determine potential consumer willingness to pay for alternatives to disposing of the products in landfills.

- The context must recognise the New Zealand Waste Strategy, which focuses in part on reducing overall waste. Willingness to pay for alternatives to landfilling end-of-life products contributes to the value of the Waste Strategy.
- Alternatives to landfilling end-of-life products are reuse (finding someone who will use the end-of-life product for its original or an alternative use), recycling (extracting material from an end-of-life product for reuse or processing for other uses), and recovery (finding a useful purpose for otherwise waste material. In this report, these alternatives are collectively referred to as diversion from landfill.
- Economic drivers – including willingness to pay but also considering the value of resources to be recovered – is only one element of a regulatory approach to waste diversion. Political and institutional drivers must be considered to achieve an integrated and robust solution.
- There are few estimates of willingness to pay for end-of-life treatment of the six identified products. With most studies, there are significant concerns using benefit transfer methods to apply them to New Zealand, including poor design of the studies and differences between source countries and New Zealand.
- Survey research found that New Zealanders were willing to pay an estimated $2.22 per tyre for recycling (Denne, Atreya, & Robinson, 2007), which suggests an aggregate figure of $10.7 million annually to recycle the annual 4.8 million end-of-life tyres (Tyrewise Working Group, 2012).
- E-waste is a wide category with complex end-of-life treatment. Survey research suggests that consumers are willing to pay a few dollars per item (in local currency – United States, Australia, Macau) to divert e-waste into recycling systems. Mobile phones, another subset of e-waste, generally retain some value even after they have been replaced. Consumers are unlikely to pay for disposal or recycling of mobile phones, at least in the short term.
- Mercury-containing lamps could be viewed as a subset of e-waste; no willingness to pay estimates are available.
- Lead-acid batteries tend not to be a disposal problem because the materials are valuable enough to support recycling by commercial concerns. No willingness to pay estimates are available.

[1] E-waste, batteries, mercury-containing lamps, tyres, agricultural chemicals and products containing refrigerants

- Household batteries are probably not much of a disposal problem, with the exception of button-type batteries that contain mercury. No willingness to pay estimates are available.
- Agricultural chemicals have little to no possibility of reuse, recycling or recovery. Landfilling is not a safe option. The focus is on safe disposal. No willingness to pay estimates are available.
- Management of end-of-life refrigerant gases can be improved by greater reuse of residual gases and safer destruction of gases; studies show a willingness in other countries to pay for less harm.

Recommendations

- **Well-structured willingness to pay research should be conducted for selected products to inform durable waste policy in New Zealand.** Primary research of willingness to pay in New Zealand would help inform the implementation of the Waste Strategy and future decisions on regulatory intervention under the Waste Minimisation Act. Such research could focus first on e-waste, batteries (except used lead acid batteries) and mercury-containing lamps. Tyres, agricultural chemicals and refrigerants gases have other pressures driving end-of-life treatment.
- **Overseas estimates for willingness to pay can be used in New Zealand for some products, such as tyres and (initially) e-waste.** However, further research will be needed to confirm willingness to pay and understand the underlying dynamics of disposal and recycling, particularly for e-waste.
- **Further work should be conducted on implementation design.** No primary research addresses the question about willingness to pay at different points in product life cycles.
- **Point of sale levies should be preferred to end-of-life levies in New Zealand for these products.** This is because:
  - the products investigated are imported, which makes it difficult for New Zealand to regulate producer behaviour during design and manufacture directly
  - the initial purchase puts the products into use, leading to the eventual end-of-life, so is a good point for influencing consumption decisions
  - fees at the time of disposal would, by contrast, encourage free riders, hoarding and illegal dumping.

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Glossary

Diversion	sending end-of-life products somewhere else than a landfill
Diverted material	any thing that is no longer required for its original purpose and, but for commercial or other waste minimisation activities, would be disposed of or discarded (New Zealand Waste Minimisation Act 2008)
Dumping	improper disposal of waste
End-of-life	describes a product that a consumer no longer requires for its original purpose
Fly-tipping	disposal of waste in a location that is not a landfill
Free-riding	not contributing to diversion but relying on other people to deal with waste issues; not 'doing your part'
Hazardous waste	any material that could cause a problem if improperly disposed of. There are many types of problems, including environmental, public safety, worker safety
Illegal dumping	fly-tipping, or improper disposal at a landfill (where the facility is not approved to handle the specific type of waste)
Landfill	an approved facility where waste is collected and buried
Prisoner's dilemma	a 'game' or situation studied in economics that describes the pay-offs to cooperating or 'defecting' (refusing to cooperate). In the waste context, everyone benefits most when everyone goes to the trouble of reducing, reusing, recycling and recovering. However, individuals can improve their personal situations by just sending everything to landfill and depending on everyone else to do the recycling. This is one explanation of the incentives for 'fly-tipping'
Recovery	any operation the principal result of which is for waste to serve a useful purpose (European Commission, 2008)
Recycling	any recovery operation by which waste materials are reprocessed into products, materials or substances whether for the original or other purposes
Resource recovery	a general term for diverting material from landfill, generally via recycling or recovery (rather than reuse)
Reuse	any operation by which products or components that are not waste are used again for their original purpose (European Commission, 2008)
Waste	any substance or object which the holder discards or intends or is required to discard (European Commission, 2008)

1. Introduction

This study explored the international literature to determine where there might be unrealised benefit in current treatment of six end-of-life products, with a focus on willingness to pay studies. We examined the products in the context of the New Zealand Waste Strategy and the Waste Minimisation Act 2008. In theory there are several ways in which markets may not capture all benefits, as discussed in Appendix B; evidence was sought to show whether these benefits might exist in New Zealand.

1.1. Scope of the project

The Ministry for the Environment (the Ministry), required research that allows it to further understand the extent to which public concern exists about the disposal/low levels of recycling of the following end-of-life products:

- e-waste
- batteries
- mercury-containing lamps
- tyres
- agricultural chemicals
- products containing refrigerant gases.

More specifically, the Ministry would like to understand New Zealanders' willingness to pay (WTP) for higher levels of recovery, improved treatment, or higher levels of recycling. The Ministry considers that WTP may be one element in an overall waste-minimisation strategy.

The focus of the research is the willingness to pay for the safe disposal or recycling of end-of-life products. Little New Zealand data is available, therefore the assessments will require examining overseas studies (if available) to be used as a proxy for the economic value that the New Zealand public place on the environmentally sound management of these products.

The key policy questions to be answered are:

- does the New Zealand public want to recycle the product at the end-of-life enough to pay for it? And if yes, how much are they likely to want to pay per unit of end-of-life product?
- will they prefer to pay at point of purchase or point of disposal?

1.2. Background and policy context

Government policy on waste is presented in the New Zealand Waste Strategy. Pivotal to government policy is the Waste Minimisation Act (2008). It encourages a reduction in the amount of waste generated and disposed of in New Zealand and aims to lessen the environmental harm of waste.

Product stewardship is part of the Waste Minimisation Act. Product stewardship encourages (and in certain circumstances require) people and organisations involved in the life of a product to share responsibility for:

- ensuring effective reduction, reuse, recycling or recovery of products
- managing environmental harm arising from the product when it becomes waste

Product stewardship schemes are initiatives that help reduce the environmental impact of manufactured products. When a product stewardship scheme is introduced anyone involved in the product life cycle such as producers, brand owners, importers, retailers and consumers accepts responsibility for its environmental effects.

The revised New Zealand Waste Strategy, which sets out the Government's long-term priorities for waste management and minimisation, was released in 2010. The strategy provides high-level direction. Its two goals are:

- Goal 1: Reducing the harmful effects of waste. There is a potential for waste to cause harm. Waste should therefore be reduced or its effects managed
- Goal 2: Improving the efficiency of resource use. Waste represents resources that could otherwise be used for production or consumption.

The strategy states that, to support New Zealand's moves towards improving efficiency, a change in the way we all buy and dispose of goods and services is needed and pricing policies that reflect the full costs associated with waste are one way to instigate this behaviour change.

New Zealand is currently a party to four international agreements that cover hazardous waste: The Basel Convention, the Waigani Convention, OECD Decision C/2001 107/Final and the Stockholm Convention.

1. The Basel Convention restricts and regulates the transboundary movement and ensures environmentally sound management of hazardous wastes.
2. The Waigani Convention governs transboundary movements of hazardous waste in the Pacific.
3. The OECD Decision governs transboundary movements of hazardous waste for recovery operations in OECD countries.
4. The Stockholm Convention places restrictions on the production, use, emission, trade and disposal of certain chemicals which are persistent organic pollutants.

A further convention, the Minamata Convention, may be signed by New Zealand in October 2013 and will further control the transboundary movement of mercury wastes (in similar ways as the Basel Convention controls hazardous waste transboundary movement). New Zealand Government is considering whether to sign this Convention.

There are two international agreements dealing with the emission of ozone depleting substances and synthetic greenhouse gases. These agreements are:

1. The Montreal Protocol, which restricts use of ozone depleting substances; and
2. The United Nations Framework Convention on Climate Change (and its Kyoto Protocol), which aim to reduce greenhouse gas emissions from synthetic greenhouse gases.

International agreements focus on a large number of different products that can be recycled. This study focuses on six products whose disposal at end-of-life would be affected by these international agreements, and domestic concerns. These include:

E-waste

Ownership of electronic items such as televisions and computing equipment is rising rapidly, and with it, environmental and human health concerns about the disposal of end-of-life equipment (e-waste). There is evidence of growing concern in the community (eDay Report 2010) about sending e-waste to landfill when it could be recovered and the potentially valuable resources recycled. Although e-waste makes up a relatively small percentage of the total waste going to landfill, it is a growing waste stream and triggers such as the Going Digital, technological changes (e.g. shorter life cycles) and innovation (e.g. new products) are contributing to a surge in its disposal. Current recycling rates are low and there are no nationwide, comprehensive product stewardship schemes in place that cover this type of equipment. As a hazardous waste and one with components that may contain persistent organic pollutants, e-waste comes under the requirements of both the Stockholm and Basel Conventions.

Batteries

The number and diversity of electrical and electronic products that use batteries has grown dramatically in the past few decades. Most homes and businesses contain many pieces of equipment that depend on batteries to operate such as portable computers, mobile phones, cameras, toys, watches, torches, power tools, and electric toothbrushes.

The widespread use of batteries has created a number of environmental concerns. Recycling or proper disposal of batteries can prevent dangerous elements (such as lead, mercury and cadmium) found in some types of batteries from entering the environment. As with mercury-containing lamps, there may, in the future, be new requirements on New Zealand in relation to the trade and disposal of mercury-containing products and wastes that could impact mercury-containing batteries.

Rechargeable batteries that are embedded in electronics products like mobile phones can be considered as a subset of e-waste.

Mercury-containing lamps:

The use of mercury helps to make lighting more efficient. A mercury-containing compact fluorescent lamp is up to 80% more efficient than a standard incandescent lamp for example. The majority of environmental impacts from lighting occur during the use of a lamp, through the consumption of electricity. Therefore, while mercury is a potentially dangerous substance it offers environmental benefits that potentially outweigh its impacts.

A new international agreement, the Minamata Convention, if it comes into force in New Zealand, would place restrictions on the import, export and manufacture of selected lamps. New Zealand has not signed or ratified the Minamata Convention on mercury. It opens for signature in October 2013.

Lamps and lighting equipment could also be considered a subset of e-waste.

Tyres

Tyres in landfills create management issues, while in stockpiles they pose risks of fire and spread of disease vectors. In general, they may also pose leachate risks (including in engineering, farming and artificial reef applications). Tyres are also a potential resource, particularly as a source of energy.

Tyre Track, a voluntary product stewardship initiative for end-of-life tyres, ran for five years (2004-2009). This was a partnership between the Motor Trade Association and the Ministry for the Environment, but was found to be unsuccessful due to 'free rider'[2] issues.

In March 2012, the Minister for the Environment awarded Waste Minimisation Fund funding for the development of a stewardship programme for end-of-life tyres (ELTs). The project enabled industry to work together to deliver a consistent nationwide approach to the responsible disposal of tyres. Various industry stakeholders are working together to develop the programme. Stakeholders include companies and organisations representing tyre importers and suppliers, new and used car importers, tyre recyclers, motor services, and motorists as well as local and central government.

Agricultural chemicals

Chemicals that are used on farms are known as agricultural chemicals, or agrichemicals. New Zealand's agriculture sector uses a variety of chemicals for pest and weed management, animal health, sanitation and export requirements.

Improper use or disposal of agricultural chemicals can pose a significant risk to the environment, human and animal health, and trade. Leaking chemical containers can poison waterways, soil and groundwater.

Agrichemicals may become surplus when farming methods or owners change, chemicals expire, or chemicals previously in use are deregistered. Some surplus agrichemicals are inappropriately stored, are in damaged containers, or are unlabelled.

For the disposal of some agrichemicals, New Zealand also has international obligations under the Stockholm, Rotterdam and Basel Conventions.

The Agrecovery programme has been designed as a solution for responsible brand owners/manufacturers to provide for the collection of their customers' used containers. The programme is designed to recover used containers that are Agrecovery-branded. This means it will only be used for products from member companies. Membership of Agrecovery is open to all producers in the sector. Waste agrichemicals and their containers are collected by regional councils and the Agrecovery programme. This voluntary product stewardship scheme was accredited on 23 September 2010 and the programme has also raised issues about 'free riders'.

Although the bulk of legacy agrichemicals are considered cleared in most regions, residual amounts still remain and waste continues to be generated from use of more modern chemicals.

[2] Free-riders are those who don't contribute or comply with a product stewardship scheme, but benefit from it.

Products containing refrigerant gases:

CFCs (Chlorofluorocarbons) were developed in the 1930s as safe, non-flammable and non-toxic refrigerants. They were widely used until it was confirmed in the 1980s that they were the main source of harm to the ozone layer. CFCs have not been imported into New Zealand since 1996. However, there are still CFCs left in New Zealand in older industrial air conditioning and refrigeration systems, car air conditioning systems and domestic refrigerators.

CFCs, as well as harming the ozone layer, are also strong greenhouse gases that can cause climate change. The 80 grams of CFC in a domestic refrigerator can destroy 3 tonnes of ozone.

HCFCs or hydrochlorofluorocarbons were developed as 'transitional' refrigerants. They do much less damage to the ozone layer than CFCs, and they are being used by the refrigeration industry while other longer-term replacements are developed. Imports of HCFCs will be phased out by 2015. HFCs or hydrofluorocarbons do not harm the ozone layer but they are highly potent greenhouse gases (from 140 to 12,000 times more potent than CO2). HFCs are part of the 'basket of gases' covered under the Kyoto Protocol.

To support the international protocols established to protect the environment from the release of synthetic refrigerants, the Trust for the Destruction of Synthetic Refrigerants collects and disposes of (destroys) in an environmentally acceptable way, CFCs, HCFCs, HFCs and any other synthetic refrigerant and mixes containing them which have been used in the refrigeration and air conditioning industries.

Under signed agreements with refrigerant wholesalers, a voluntary levy is received on the sale of ozone depleting refrigerants and HFCs/Synthetic Greenhouse Gases.

This voluntary product stewardship scheme was accredited in September 2010[3] however, the scheme does not collect the voluntary levy on all refrigerant-containing products imported into New Zealand, such as in new and second hand cars, nor does it cover all of the bulk importers of refrigerants.

1.3. Framework for analysis

To understand the literature on waste, it is necessary to put WTP in to an economic framework to explain behaviour. An extended framework is set out in the Appendix B. The main points of the framework are as follows:

- WTP is a measure of consumer demand. In its broadest sense, it captures the total economic value (TEV) of a product
- TEV allows us to talk about all the ways that product contribute to our well-being: through direct and indirect experience, through using products or conserving resources for future use
- negative impacts also affect TEV: values can be reduced, particularly through wider impacts known as 'externalities'
- the market value of end-of-life products is best explained with a supply-demand framework that focuses on the provision of alternatives to landfills and the demand for such alternatives

[3] http://www.refrigerantrecovery.co.nz/index.shtml)

- interventions that recognise the TEV of end-of-life products may produce more waste minimisation
- economic drivers are not, however, the only consideration. A wider view also considers the political and institutional drivers. These drivers – such as international obligations or industry coordination – may be crucial in determining appropriate policy.

2. E-Waste

2.1. Introduction

Disposing of society's waste is an on-going issue. In the last two decades, the growth in electrical and electronic equipment production and consumption has been significant. This is largely due to increased demand in developing economies, a generally high product obsolescence rate, decreasing in prices, and the growth in internet use.

Electrical and electronic waste (hereafter referred to as e-waste) is the fastest growing waste stream (about 4 per cent growth a year world-wide). There is no standard definition of e-waste. The Organisation for Economic Co-operation and Development (OECD) defines e-waste as 'any appliance using an electric power supply that has reached its end-of-life' (UNEP, DTIE, 2007a). The most widely accepted definition of e-waste is as per European Commission Directive 2002/96/EC: 'electrical or electronic equipment, which is waste … including all components, subassemblies and consumables, which are part of the product at the time of discarding' (European Commission, n.d.(a)).

Most batteries for consumer electronics are disposed of in the general waste stream.[4] The batteries contain elements that can be harmful, such as cadmium, or materials that are potentially valuable (Tanskanen & Butler, 2007). Mobile phones are also of interest because many are still functioning when they are replaced (Wilhelm, Yankov, & Magee, 2011). Over 60 percent of phones are retained as back-up phones (Ongondo & Williams, 2011; Wilhelm et al., 2011), which provides an indication of the proportion of phones that are still working.

In general, large household appliances represent the largest proportion (about 50 per cent) of e-waste globally, followed by information and communications technology equipment (about 30 per cent) and consumer electronics (about 10 per cent). The composition of e-waste is very diverse and differs across product lines and categories. Overall, it contains more than 1000 different substances which fall into 'hazardous' and 'non-hazardous' categories; significantly, the toxicity of many of the chemicals in e-waste is unknown. Broadly speaking, electronic products consist of ferrous and non-ferrous metals, plastics, glass, wood and plywood, printed circuit boards, concrete and ceramics, rubber and other items. Iron and steel constitutes about 50 per cent of e-waste followed by plastics (21 per cent), non-ferrous metals (13 per cent) and other constituents (UNEP, DTIE, 2007a).

[4] For example in Australia one survey suggested 70% of batteries are disposed of in general waste with 6% recovered for processing (http://recyclingweek.planetark.org/documents/doc-513-battery-research-report-final.pdf p3)

2.2. Issues

For New Zealand we have focused on televisions, mobile phones, and computers since they are the main contributor of e-waste. In the Table below we have set out the volume of cathode ray tube (CRT) televisions, CRT monitors and volume of desk top monitors. While the number of mobile phones being used is not known, Statistics NZ report that the number of mobile broadband users is roughly 2.5 million.[5]

While the numbers may seem quite large, it represents a tiny portion of the estimated 2.5 tonnes of total waste each consumer disposes of each year.

Table 1 Estimated volume of computers and televisions in New Zealand

Item	No.	Ave weight	Total weight	Lead weight	Per head of population	
	Millions	Kg	Tonnes	Tonnes	Number	Kg
Desktop boxes	7.8	11	85,800	n.a.	1.8	19.8
CRT monitor (bus)	3.1	16	49,600	4,340	0.7	11.2
CRT monitor (home)	1.9	16	30,400	2,660	0.4	6.4
CRT TV	2.2	24	52,800	7,040	0.5	12.0
Totals	15		218,600	14,040	3.4	49.4

Source: MacGibbon (2011) p31

Disposal of e-waste creates public concerns because of perceptions that:

- it adds to the volume of waste in landfills
- there is a loss of scarce resources embodied in the items.
- there are potentially hazardous materials in e-waste that damage human health. E-waste is classified as hazardous waste (Tsydenova & Bengtsson, 2011) having adverse health and environmental implications. Approximately 40 per cent of the heavy metals found in landfills comes from electronic waste (Montrose, 2011)
- the public perception is that e-waste imposes many challenges on the recycling industry (Smith, Sonnenfeld & Naguib Pellow, 2006) as it contains many different materials that are mixed, bolted, screwed, snapped, glued or soldered together. Toxic materials are attached to non-toxic materials, which makes separation of materials for reclamation difficult. Hence, responsible recycling requires intensive labour and/or sophisticated and costly technologies that safely separate materials[6]
- there is an overhang of e-waste in the community because consumers perceive that it is difficult to dispose of e-waste effectively and cheaply

5 http://www.stuff.co.nz/technology/digital-living/7808200/2-5-million-Kiwis-using-mobile-Internet

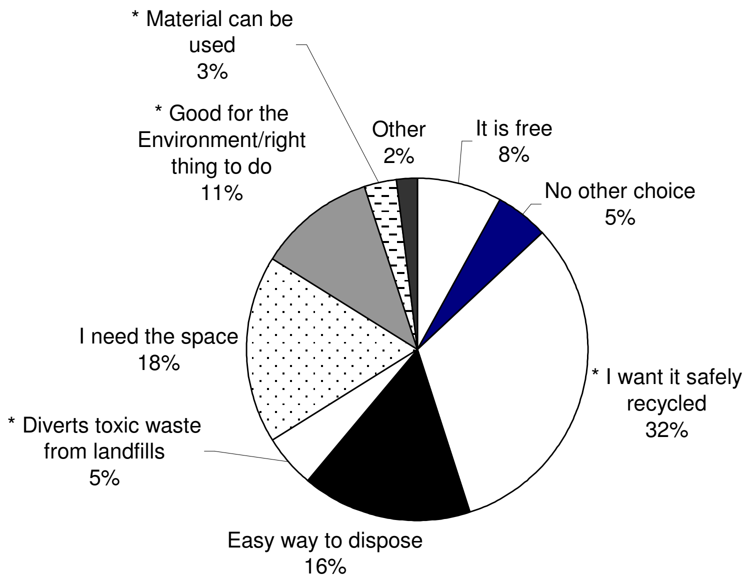
6 http://www.ilo.org/wcmsp5/groups/public/---ed_dialogue/---sector/documents/publication/wcms_196105.pdf

- internationally, New Zealand is seen as an outlier since it is one of the only countries that does not have active policies to divert e-waste from landfills.

The public concern translates into some consumers paying for disposal, some illegally dumping and others hoarding e-waste leading to an overhang of e-waste in the community.

While we do not know the relative magnitudes associated with each behaviour, the last e-day survey is instructive. In New Zealand, consumers are concerned with the amount of environmental damage caused by e-waste given the survey responses taken in e-Day 2010.

Figure 1 What is the main reason for bringing your computer waste here today?



Source: e Day report 2010

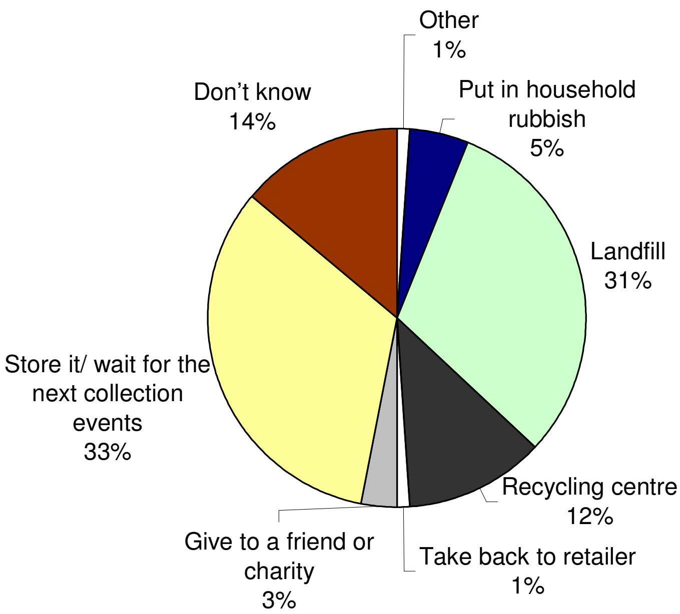
While the environmental damage (e.g. leakage of hazardous waste from landfills) may not be very large in the short term, even small amounts of hazardous waste are still important enough for consumers to hoard rather than attempt to dispose of the waste through the general waste stream (Nixon 2011 p54-72).[7] This behaviour suggests that there is demand for keeping these end-of-life products out of the landfill, but that this demand is not currently being met by a ready supply of alternatives.

Government and local government behaviour reinforces this by supporting services for e-waste recycling that charge at point of disposal e.g. e-waste is recyclable for between $0 and $25 per item depending on the location, service provider, level of subsidy and type of e-waste being disposed. The lack of affordable disposal increases the likelihood of e-waste hoarding.

[7] Also see: http://www.mfe.govt.nz/publications/waste/eee-survey-report-jan06/html/page3.html

Furthermore, the annual e-Day recycling event between 2006 and 2011 in selected New Zealand cities has increased expectations that the public will be able to freely dispose of any e-waste they might have. Figure 2 sets out the responses from an e-Day survey asking consumers what might they do if there was no e-Day. Substantial numbers were not sure what they would do with their e-waste or would store it (33%) or landfill it (31%).

Figure 2 If there was no e-Day how would you dispose of e-waste?



Source: e Day report (2010)

Covec (2012) points to the lack of New Zealand data for charges on waste minimisation. The Covec (2012) report examines the rationale for government intervention to minimise waste, examines issues around disposal pricing, and examines the costs and benefits of waste minimisation. While the report is about waste in general, it points to an uncertain impact where the costs may or may not outweigh the benefits. The figures range from net benefit of between -$292,000 and $950,000 of diverting waste from landfills.

Recycling of e-waste is costly because of the nature of the recycling process. Computers and televisions are all differently configured and are made of multiple materials that are sometimes stuck together with glue and resin. Some materials have value and others have minimal value. Recycling e-waste is labour intensive.

In 2010 the RCN group (e-waste recyclers) partnered with the Community Recycling Network and set up a public collection and recycling solution (RCN e-Cycle) with a grant from the Waste Minimisation Fund. No manufacturers are involved in the RCN e-Cycle programme. The costs ($5 to $20 per item) lie with the end consumer and/or ratepayer.

2.3. International e-waste connections

For most e-waste materials, an optimal level of abatement will not result in zero waste, because not all e-waste is equally avoidable. For instance, recovering pre-used products (including potentially reusable materials) from the community is the easy part of e-waste recovery. A harder problem is how to disassemble the components and extract value from the e-waste products in an efficient manner when exporting to countries with abundant labour resources is restricted.[8] This disadvantages nations such as New Zealand, because labour is relatively expensive, the amount of e-waste generated is relatively small, there are no onshore recycling facilities and New Zealand is a long way from markets for recyclable materials.

The lack of scale and relatively high labour costs in New Zealand are some of the reasons why New Zealand has not developed waste schemes that manage products such as e-waste. However, as time goes on it is likely that New Zealand will come under increasing pressure from NGOs, councils and recyclers to adopt product stewardship models operating in nearly all OECD nations (Tripathi 2011).

In Table 2 and subsequent product tables the descriptors represent:

- An overview of waste disposal issues of each item
- Who bears the cost of disposal within the waste disposal cycle of each item
- The costs borne by the end-of-life holder of the item being recycled
- A description of the extent of recycling of each item
- The extent of illegal dumping (if known)
- International agreements that impact on the disposal and shipment of the recycled item
- The sufficiency or durability of the system/policy that supports the disposal of the item
- A description of the most efficient way of disposing of the item
- Who bears the costs of recycling within the recycling system of each item.

[8] The Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and their Disposal came into force in 1992. Its intent was to put in place strict rules around the import and export of hazardous waste. It requires verifiable notice, consent and tracking of waste across national boundaries and encourages the recycling of waste in home countries. New Zealand signed the Basel Convention in 1989 and ratified it in 1994.

Table 2 Overview of NZ end-of-life situation for e-waste

Descriptor	
Overview	E-waste is increasing in size as a proportion of overall waste. Concern by the public is increasing with a large overhang of e-waste products in the community.
Cost of disposal	User pays through community or local government-based operations, CRN/RCN e-Cycle network or other commercial pay-on-disposal e-waste recycling services. User pays approach with a subsidy from the Waste Minimisation Fund (e.g. TV TakeBack Programme) and some District councils.
Cost to end-of-life holder to recycle	Typical prices per item are between $0 and $25 per item.
Extent of recycling	Limited. No comprehensive data available. The recycling services are uneven and New Zealand relies on the global recycling market
Extent of 'dumping'	Unknown. Large overhang in the community.
International commitments	Basel Convention, Waigani Convention, Stockholm Convention and in the future potentially the Minamata Convention (if New Zealand becomes a Party).
Sufficiency	Question as to the durability of policy given growing public concern and international pressure through international agreements.
Efficiency	Technical costs of disposal outweigh benefits. Does not take into account WTP.
Fairness	Some consumers paying for e-waste disposal, others hoard e-waste and others have incentives to illegally dump. Although the size of each category is unknown (MacGibbon, 2011)

Source: NZIER

2.4. WTP

We have used total economic value (TEV) as a way of examining the costs and benefits associated with e-waste management. Use values are relatively easily to define into commercial and indirect values. Use values involve the direct commercial activities associated with recycling materials. An indirect use value might be the value minimisation of waste (and e-waste) being put into landfills so that potentially e-waste streams are more efficiently used.

We are focused on the non-use values and how we might examine their value. These include option values (i.e. to preserve landfills for future use), existence value (i.e. to preserve what we have or improve what we have e.g. this includes the knowledge that e-waste is being dealt with in a socially acceptable way), and other non-use values (i.e. preserving something for future generations e.g. a durable solution is found).

One method used to value the non-market benefits is to use Willingness to Pay (WTP) surveys to gauge community e-waste recycling values. Given the scarcity of New Zealand WTP studies on waste issues we examined studies in overseas jurisdictions with a view to assessing the portability of results.

The large quantities of e-waste being generated in developed countries has caused wide spread concern in those communities over the appropriate disposal of end-of-life electronic equipment.[9]

For example, the Environment Production and Heritage Council (EPHC) in Australia commissioned URS (2009) to examine community e-waste values since opinions had been expressed that the level of e-waste recycling was too low and that too much e-waste was being disposed in landfills.

URS (2009) found that consumers were willing to pay an extra A$0.50 cents per new item bought for a 1% increase in e-waste diversion. This translated into a WTP of A$3.9 million over 5 years for the five biggest Australian cities. Younger people and higher income earners were more likely to support recycling e-waste options.

For similar concerns, Nixon and Saphores (2007) in California found that consumers were willing to pay an extra 1% on the purchase price for improved e-waste disposal.

Nixon et al (2008) show that consumers are willing to pay for e-waste recycling. The willingness to pay is made up of bundles of time and money. To understand consumer preferences Nixon et al compares various e-waste recycling programmes. To calculate WTP for increased e-waste recycling convenience (i.e. the trade-off at the margin between cost and convenience) Nixon et al test a number of options.[10] They show that Californian households prefer drop off recycling at regional centres. In this preferred approach (using a contingent ranking model) they show that households are willing to pay approximately $US0.13 per mile per month to increase e-waste convenience.

In a national survey of US households, Saphores et al (2012) find that the most important variables in explaining willingness to pay were social pressures to recycle followed by convenience, knowledge of toxicity and prior experience with e-waste recycling. It advocates more work being done on stimulating public awareness of e-waste recycling.

In a series of surveys in Wisconsin (2006, 2010, 2011)[11] consumers have been willing to pay up to $US5 to have electronic devices recycled. These surveys have shown slight increases in willingness to pay as time goes on.

A study in Macau (Song et al 2012) found that households were willing to pay as much as $US 2.50 per household for e-waste recycling (an annual figure of $US 5 million).

In Europe the dominant e-waste issue has been the introduction of the Waste Electrical and Electronic Equipment Directive (or WEEE). All member countries of the EU are mandated to develop national product stewardship regulations within the WEEE Directive including a requirement that producer pays for recycling with no direct costs for consumers. Achillas et al (2012) showed that in Greece there was a large degree of public ignorance on the impact of e-waste. However, the average fee consumers were willing to pay exceeded existing recycling charges. They argue that existing recycling programmes in Greece could be augmented to include e-waste disposal and provide an improved network of disposal sites and bins.

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9 Note that while there are many definitions of e-waste we focus on televisions and computers as being the main sources.

10 These include 1. Pay as you throw, 2. Drop Off Recycling a Regional Collection Centres, 3. Curb side recycling, 4. Retail drop off, and 5. Deposit-Refund programme at retail locations.

11 See for an overview: http://dnr.wi.gov/files/PDF/pubs/wa/wa1604.pdf

We were not able to find an estimate of consumer WTP for recycling or diversion of mobile phones. The fact that mobile phones are retained suggests that they maintain some residual value for consumers, an option value to be exercised if the principal phone breaks or is lost. Consumers may therefore require compensation to be induced to recycle old phones (Ongondo & Williams, 2011). Research has shown that mobile phone recycling is affected by attitudes to recycling and general environmental attitudes (Milovantseva & Saphores, 2013), although the exact relationship between attitudes and phone recycling are complex (Hansmann, Bernasconi, Smieszek, Loukopoulos, & Scholz, 2006).

Mobile phones have been studied in the New Zealand context (URS, 2006a). The research noted that the treatment of mobile phones should be properly viewed as a product lifecycle issue, essentially arguing for a product stewardship model. However, the scope of the research was on end-of-life treatment of mobile phones. The conclusion of the report was that the industry – Vodafone and Telecom at the time – were working through the associated issues. Their drivers were environmental and social responsibility as well as customer service. As a result, government intervention was thought unnecessary. The report did note that 10-15% of cell phones in New Zealand were sent to landfill while the rest were stockpiled informally.

Indications from the e-day surveys also show that there is increased concern about how e-waste is disposed of.

Table 3 Evidence of WTP for e-waste

Country	Key findings	Limitations	WTP
NZ	Concern is growing about how NZ should dispose of e-waste. International and domestic pressure to develop a product stewardship scheme	Little information on how much e-waste is disposed of	No national WTP survey
AUS	The first e-waste product stewardship programme has been established	Large sample size conducted in five cities	Consumers WTP $A0.50 cents per item for every 1% increase in e-waste diversion
US	Product stewardship schemes underway. Business participating for branding purposes	Mainly city based surveys	A number of surveys undertaken. Most show positive WTP of up to 1% of product value
EU	Product stewardship required under the WEEE Directive	Limited number of surveys	Show slightly positive WTP

Source: NZIER

2.4.1. International agreements

There is a growing level of policy and regulation on e-waste product stewardship globally. Most OECD nations are using some sort of product stewardship scheme or extended producer responsibility regulation to target e-waste. Of importance to New Zealand are:

- The Basel Convention that restricts the movement of e-waste internationally. Companies exporting e-waste from New Zealand must obtain a Basel Permit from the Environmental Protection Authority
- The Waigani Convention governs transboundary movements of hazardous waste in the Pacific. It stops e-waste from being exported to South Pacific nations but allows e-waste exports from the Pacific to New Zealand and Australia
- OECD Decision C/2001 107/Final governs transboundary movements of hazardous waste for recovery operations in OECD countries.
- The proposed Minamata Convention would have implications for the way mercury containing products are handled in New Zealand and are exported, if New Zealand becomes a Party. This may have an impact on LCD computer and television screens e.g. it will phase out import/export and manufacture of cold-cathode fluorescent lamps (CCFL) and external electrode fluorescent lamp (EEFL) lamps by a certain date (which are primarily used in LCD screens).
- The Stockholm Convention Persistent Organic Pollutants (POPs) aims to protect human health and the environment from POPs by banning the production and use of some of the most toxic chemicals known. In May 2009 nine new Persistent Organic Pollutants (POPs) were listed on the Convention's annexes, including certain brominated flame retardants[12]. These brominated flame retardants have been used in the past in the plastic components of e-waste.

2.4.2. How e-waste is paid for in selected developed nations.

Funding e-waste recycling is a major concern in developed nations since each nation's attempts to find a mechanism that covers the costs. Most have opted for product stewardship schemes where the producer or importer is responsible for covering the costs of recycling. Some countries have an explicit advanced fee added to the price of a new product.

Costs of collection and recycling differ. Approaches are mixed, however a common approach is that consumers bear the costs of taking the e-waste to local collection centres which are typically supported by local government. Producers then take responsibility for the recycling costs. The roles in e-waste diversion in several jurisdictions are shown in Table 4.

[12] commercial pentabromodiphenyl ether (c-pentaBDE) and commercial octabromodiphenyl ether (c-octaBDE)

Table 4 E-waste: roles in waste diversion

Jurisdiction	Collection	Recycling costs
Australia	No-one has obligations to provide collection service. State governments have partnered with product stewardship schemes to assist in collection	Producers (importers)
Ireland	Retailers/Suppliers must collect for free if consumer buying similar new product.	Producer Consumer (legacy products)
Japan	Producers (take back obligation)	Consumer (financial obligation) responsibility to purchase a recycling ticket.
South Korea	Retailers/Suppliers must collect for free if consumer buying similar new product. Consumer pays fee for local government collection	Producers pay an advanced deposit covering recycling costs. Annual sales drive recycling targets. Consumer pays fee for local government recycling.
Sweden	Local government	Producer
Switzerland	Local government	Producer
USA, California	Consumer and local government	Partial industry but mainly local government
USA, Maine	Producer	Producer
USA, Minnesota	Producers (manufacturers), with retailers having reporting requirements	Producers, through the recyclers they use. Targets based on manufacturer's sales in Minnesota (not legacy products)

Source: Adapted from Tripathi (2011), http://www.environment.gov.au/settlements/waste/publications/pubs/product-stewardship-na-eu.pdf, and http://www.environment.gov.au/settlements/waste/publications/product-stewardship-asia.html

2.4.3. Implications

Non-market value estimates for New Zealand for e-waste are not available. While we know that the use costs of recycling e-waste is likely to be higher than the benefits we are unsure of the benefit value that New Zealanders put on recycling of e-waste (non-use values).

From the overseas literature it is known that consumers are prepared to pay extra for proper disposal of e-waste. Estimates range up to 1% of the product value. Also manufacturers are also engaged in product stewardship schemes because of state and national regulations (in the US and Europe) and also for branding purposes.[13]

[13] Possibly, for larger sized companies the cost of stewardship schemes is much less than for smaller companies therefore they can take advantage of economies of scale.

With little data on New Zealanders WTP the cost of the potential externality is unknown. Also with the only option available being user pays for disposal the likelihood of illegal dumping is high.

The exporting restrictions under the Basel Convention are also a constraint since processing e-waste in New Zealand is not of sufficient scale to be cost effective. With transaction costs high, small scale recycling and limited export markets, the cost of e-waste disposal is relatively high compared with other OECD nations.

Table 5 E-waste and issues with the market for diversion

Descriptor	
Heterogeneous preferences	Unknown in New Zealand. Overseas consumers are willing to pay more for products that are disposed of in an environmentally friendly way. Also, firms are willing to be involved in product stewardship for branding purposes.
Non-market values	Unknown in New Zealand
Externalities	The full cost of disposal may not be paid by the consumer/manufacturer
Prisoner's Dilemma	Illegal dumping risk is high under current system[1]
Economies of Scale	Little scale in New Zealand, therefore recycling installations are unlikely to be as cost effective as those operating on other OECD nations
Information	Little information on e-waste disposal available.
Transaction costs	Relatively high compared to the general waste stream and other OECD nations
Note (1) The current system creates incentives for dumping that are potentially high. However, we do not have any data to test this proposition.	

Source: NZIER

3. Batteries

3.1. Introduction

Batteries are not a unified category of product when considered for purposes of waste, recovery and recycling. It is important to be clear about which specific product is under consideration, because some are highly toxic, some contain hazardous materials, and some are inert.

End-of-life batteries can be categorised in different ways to give different pictures of the situation (Warnken Industrial and Social Ecology Pty Ltd, 2010):

- household and commercial, or large and industrial – the categories use different types of batteries and have different preferences for collection and recycling
- technology/chemistry type, such as nickel-cadmium versus lead-acid batteries
- level of hazard, with some batteries containing hazardous material
- typical uses, in particular whether the battery is embedded or stand-alone. This affects the recovery technique, with embedded batteries essentially fitting the e-waste stream
- single use or rechargeable, which affects the product life and impact on waste stream.

One key difference is the chemistry, which then affects the level of hazard. As MfE makes clear (Ministry for the Environment, n.d.), there are essentially three groups of batteries:

- household batteries – the common single-use battery, which are largely inert and can be put in landfills
- rechargeable batteries – usually embedded in consumer electronics products, and often containing hazardous or valuable substances
- lead-acid or wet batteries – used for motor vehicles and unsafe for landfills.

These different types are three different waste streams that need to be considered separately.

Lead-acid batteries are by far the largest component of battery waste by weight. Australia research indicated that they were 90 percent of waste battery tonnage (Warnken Industrial and Social Ecology Pty Ltd, 2010). Household batteries are extremely common by count, but make up only 5 percent of waste battery tonnage (Warnken Industrial and Social Ecology Pty Ltd, 2010). While the standard alkaline battery (e.g., AA, AAA, C cells) is largely inert, some button-type batteries contain mercury. Mercury-containing batteries are considered hazardous and not appropriate for landfills.

Finally, rechargeable batteries are currently less numerous but are a growing issue. In addition, they contain hazardous materials so are not safe for landfills. The main use of rechargeable batteries is as embedded power sources in consumer electronics. They are therefore discussed in the section on e-waste.

3.2. Issues

Table 6 Overview of NZ end-of-life situation for batteries

Descriptor	
Overview	Three main issues are: - lead-acid batteries contain lead - some button-type single-use batteries contain mercury - rechargeable batteries may contain valuable or hazardous materials. They are covered in the e-waste section of this report.
Cost of disposal	Lead-acid batteries not permitted in landfills. Household batteries are a minor contributor to landfills.
Cost to end-of-life holder to recycle	Lead-acid batteries: possibly free; may require trip to transfer station. Button-type batteries: recycle at transfer station, by watch repairer or medical service provider or other option.
Extent of recycling	Most to all lead-acid batteries recycled. Unknown percentage of other batteries recycled – likely to be a low percentage.
Extent of 'dumping'	Unknown.
International commitments	Lead is a known environmental toxin, so covered under relevant agreements. Mercury could be covered by the Minamata Convention if New Zealand ratifies, and is already covered by the Basel Convention.
Sufficiency	The market solution for lead-acid batteries leads to good waste reduction. Other batteries are likely to be put in the general waste stream – current situation does not provide diversion.
Efficiency	Recycling of lead-acid batteries is apparently efficient. The small amount of other hazardous batteries may make it difficult to create efficiencies in waste diversion.
Fairness	The ubiquity of batteries suggests that a generalised recycling programme could be fair and less confusing for public. Applying charges to producers may lead to technological change, but may be infeasible for New Zealand to attempt unilaterally.

Source: NZIER

WTP for diverting the three types of batteries should be assessed separately.

Lead-acid batteries are a small concern for waste streams. The lead in the batteries is valuable, and the technology for recovering the lead is well established and cheap enough for waste batteries to be recycled. It is estimated that 75 percent of lead-acid batteries in Australia are recycled (Warnken Industrial and Social Ecology Pty Ltd, 2010), while an even larger proportion of batteries in California are diverted from landfill. We assume in the New Zealand context that the WTP for recycling is not an important issue; the value of the waste material is sufficient to encourage diversion. However, we do not have enough data to test this proposition.

Alkaline/household batteries seem to be of little concern. Australia research notes that, excluding lead-acid batteries, 64 per cent of other batteries by weight go to landfill. A large proportion of them would be single-use, alkaline batteries (also referred to as primary batteries (CM Consulting, 2012)). They have been banned from Californian landfills since 2006 but provisions for their recycling have not been made.[14] There appears to be no research investigating the WTP for recycling such batteries.

Mercury-containing batteries comprised 11 per cent of New Zealand's mercury anthropogenic releases in 2008[15] (mercury containing lamps comprised approximately 3 per cent). This assumed that most small batteries were disposed of in domestic rubbish to landfill, with the amount imported each year the same as the amount disposed of: 168 kg Hg/year (Chrystall, Rumsby, & Pattle Delamore Partners Ltd, 2009).

A number of different types of batteries contain mercury. These include 'button cells' of silver oxide, zinc air and mercury oxide composition, in addition to some alkaline, and zinc carbon batteries. Silver oxide batteries are used in devices such as watches and calculators, zinc air and mercury oxide batteries are used in hearing aids, alkaline batteries are for general purposes, and zinc carbon batteries are commonly used in smoke alarms and clocks (Chrystall et al., 2009).

Mercury-containing batteries are all imported. Data from Customs New Zealand compiled by Chrystall et al. (2009) p33 sets out the assumed annual weight of batteries consumed in New Zealand.

Table 7 Consumption of mercury batteries in New Zealand, 2008

Battery type	Quantity	Av. Weight (kg)	Assumed total weight (kg)
Mercuric Oxide	59,240	0.001	59
Silver Oxide	994,984	0.001	995
Zinc Air	5,761,039	0.001	6,417
Zinc Air (exceeding $300cm^3$	97,651	0.001	91
Alkaline (Mn-Dioxide)	41,322,632	0.012	497,938
Alkaline (Mn- Dioxide) exceeding $300cm^3$	638,766	0.129	82,385

Source: Chrystall et al. (2009)

The final type of battery is rechargeable batteries embedded in electronics. WTP for recycling these batteries is covered in the section on e-waste. There is some research, mainly on mobile phones.

14 http://www.calpsc.org/products/batteries.

15 Note that an up to date mercury inventory undertaken in 2012 will revise these figures. It will be available on the Ministry for the Environment website.

3.3. WTP

Table 8 Evidence of WTP for diversion of batteries

Country	Key findings	Limitations	WTP
Lead-acid batteries			
Industrialised countries	Already widely recycled.	No inventory or national-level information available	Not relevant given the estimated extent of recycling.
Household batteries			
NZ	Only mercury-containing batteries are of concern.	No data available.	Unknown.
AUS	Australian Battery Recycling Initiative is promoting free drop-off locations for recycling.	No data available.	Unknown.
US	Regulations vary by state. E.g., all batteries should be recycled in California[16].	No data available.	Unknown.
EU	Covered by 2006 Battery Directive.	No data available.	Unknown.

Source: NZIER

3.3.1. New Zealand

There are no studies to demonstrate the WTP for battery recycling in New Zealand.

3.3.2. Australia

The Australian Battery Recycling Initiative (ABRI) has been formed by a group of battery manufacturers, recyclers, retailers, government bodies and environment groups to promote the collection, recycling and safe disposal of all batteries. ABRI is making information available to consumers about take-back and drop-off locations[17]. Various State and Federal regulations govern the transport and disposal of lead-acid and household batteries. Recycling of lead-acid batteries is very high, while recycling of household batteries is very low (Warnken Industrial and Social Ecology Pty Ltd, 2010). There are no studies of the WTP for battery recycling.

3.3.3. United States

There is battery recycling or diversion policies in several states. For example, batteries are classified as hazardous waste in California and are not permitted in general household waste or landfills. However, there is no programme for recycling household batteries and a large percentage of household batteries do go to landfill. These regulations do not appear to be linked to research on household WTP for diversion of batteries.

Most lead-acid batteries are recycled by for-profit reprocessors.

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16 See the California state government's CalRecycle programme: http://www.calrecycle.ca.gov/reducewaste/Batteries/. Note that while batteries are classed as hazardous waste, a low proportion of household batteries are recycled.

17 For example, a list of locations is available here: http://www.batteryrecycling.org.au/recycling/handheld-batteries.

3.3.4. Europe

Batteries are covered by the 2006 Battery Directive (2006/66/EC). In the Summary of the Directive[18], two goals are apparent:

> *The Directive prohibits the placing on the market of certain batteries and accumulators with a proportional mercury or cadmium content above a fixed threshold. In addition, it promotes a high rate of collection and recycling of waste batteries and accumulators and improvement in the environmental performance of all involved in the life-cycle of batteries and accumulators, including their recycling and disposal.*

One goal is to exclude toxic materials from the environment. The second goal is to promote recycling all end-of-life batteries. This second goal does not appear to be linked to any WTP estimates.

3.4. Implications

Table 9 Batteries and issues with the market for diversion

Descriptor	
Heterogeneous preferences	There appears to be a preference in some parts of the community to divert all batteries from landfill, including inert household batteries that make a minor contribution to landfills. This preference may not be widely shared.
Non-market values	The heterogeneous preferences activated appear to revolve around non-market values, e.g., the option value of scarce/non-renewable resources.
Externalities	Environmental damage from toxins (e.g., lead, mercury) creates externalities. Existing and prospective regulations and international agreements are contributing towards dealing with the externalities.
Prisoner's dilemma[19]	This appears to be important for toxic materials in batteries.
Economies of scale	Economies of scale are required to accumulate and recycle batteries; New Zealand lacks the scale[20].
Information	There is information on recycling available on the internet for many jurisdictions, so information is available for those interested in recycling.
Transaction costs	Household batteries make a small contribution to waste and must be taken to transfer stations or other locations for recycling. This approach creates high transaction costs for the amount of waste diverted.

Source: NZIER

18 http://europa.eu/legislation_summaries/environment/waste_management/l21202_en.htm.

19 See glossary.

20 Lack of scale is suggested by the experience of the sole lead-acid battery recycler in New Zealand, Exide Technologies. It does not have sufficient feedstock to run its facility full-time, while used batteries are also being sent to other countries for reprocessing (Watkins, 2011).

4. Mercury-containing lamps

4.1. Introduction

The mercury-containing lamps have major advantages over standard bulbs. Lower energy consumption (they consume roughly 75% less energy that standard bulbs), longer life expectancy (up to ten times), and some jurisdictions phasing out incandescent lighting have greatly increased the use of mercury-containing lamps. While the costs of such lamps/bulbs can be 3 to 10 times the alternatives, the benefits will compensate the consumers.

Garrett and Collins (2009) estimated that there were 32 million lamps imported into New Zealand in 2006. Of these 8.7 million were mercury-containing lamps. The lighting market is complex, with different technologies and a wide range of end users. There have also been government interventions in the market that have or will change the technology in use in New Zealand. These include the Electricity Commission subsidising compact fluorescent lamps and EECA introducing Minimum Energy Performance Standards (MEPS) for linear fluorescent lamps and incandescent lamps (proposed).

The study estimated that in 2007 approximately 5.6 million lamps entered the waste stream in New Zealand, equating to 717 tonnes. Of these 5.6 million lamps, just 500,000 were collected and recycled, meaning 91% of lamps were disposed of to landfill. Lighting represents a small portion of solid waste generation in New Zealand – just 0.02%. Lighting is a relatively minor source of mercury arising from products. The total load of mercury from gas discharge lamps (linear fluorescents, compact fluorescents and high intensity discharge) was at the time estimated to be 45kgs. It was expected to almost double in the 5 years from 2007. This increase would result from the compact fluorescent lamps already in use.

The main issue is that when these items reach the end-of-life they are typically discarded into the general waste stream. Mercury is a necessary part of fluorescent lamps including compact fluorescent bulbs (CFLs). Only a small percentage of CFLs are being recycled: e.g., 98% are put into landfills in the US. These are typically crushed with the potential that the vapours are inhaled by workers.

4.2. Issues

The problem is the potential harm to humans and the environment caused by mercury poisoning.

Mercury residue in landfills forms methyl mercury gas which is especially toxic. It can harm the nervous system, damage kidneys and liver and in sufficient quantities cause death.

A secondary issue is that CFLs contain rare earth metals (REMs), an important and scarce raw material.

In New Zealand, Chrystall et al. (2009) have undertaken a mercury inventory in New Zealand and concluded for CFLs that the volume of mercury is small and it is unlikely to contribute greatly to New Zealand's total mercury emissions (Chrystall et al., 2009, p. 53).

Table 10 Overview of NZ end-of-life situation for Mercury-Containing Lamps

Descriptor	
Overview	Mercury residue in landfills forms methyl mercury gas which is toxic. Chrystall et al. (2009) conclude that the volume of mercury generated is small and unlikely to contribute greatly to New Zealand's mercury emissions, although the number of lamps in use is increasing rapidly
Cost of disposal	91% of all lamps are sent for landfill disposal in New Zealand
Cost to end-of-life holder to recycle	Cost to the consumer similar to general waste
Extent of recycling	Limited – 9% of total mercury-containing lamps. This is similar to overseas recycling rates
Extent of 'dumping'	Very large both in New Zealand (9% recycling rate) and overseas
International commitments	Potentially the Minamata Convention if New Zealand ratify.
Sufficiency	Dumping of Mercury-containing lamps will continue. Question over regulatory durability
Efficiency	Landfill disposal is technically efficient. No knowledge of consumers WTP
Fairness	A limited number of consumers recycle mercury-containing lamps. Most are dumped into landfills

Source: NZIER

4.3. WTP

Tunsu et al (2011) examine the recycling literature on mercury-containing lamps. Its main focus is on the viability of industrial processes that recover mercury and REMs. Other documents focus on the problem of mercury-containing lamps and stewardship schemes which have been mandated around the world.[21]

One of the interesting features is that despite the number of product stewardship schemes around the world, the amount of CFLs dumped in landfills can still be very high. For example, in California there is a ban on CFLs being dumped in landfills. Despite this, 90% of CFLs are dumped in landfills.

We could not locate any WTP surveys on mercury-containing lamps. One of the major barriers to such a survey is the lack of understanding of the issue amongst the public.

[21] In the USA (see for example http://www.calpsc.org/products/fluorescent-lamps and in Australia (see http//www.fluorocycle.org.au/).

4.3.1. What has been done

A variety of European Union recycling regulations (e.g. Waste Electrical and Electronic Equipment Directive) make it unlawful for EU residents to dispose of CFLs in the general waste. In the United States, six states have product stewardship schemes, however though most localities rely on consumers to voluntarily mail the lamps back to manufacturers for recycling. In many other places the ban on CFLs in landfills has been delayed because of the lack of options for disposal.[22]

In developing countries, recycling is less available, and proper landfills often do not even exist in the event that the lamps are discarded as general waste.

Below we set out some examples of how mercury-containing lamps are being managed.

Table 11 Mercury-containing lamps: roles in waste diversion

Jurisdiction	Collection	Recycling costs
Australia	Local government & partially industry	Local government & partially industry
European Union[1]	Government	Producer
Japan	Local government	Producer & local government
South Korea	Local government	Producer & local government
United States[2]	Mainly local government, some voluntary industry schemes	Mainly local government, some voluntary industry schemes
USA, California	Consumer and local government	Partial industry but mainly local government
USA, Maine	Consumer and local government	Partial industry but mainly local government
Notes: (1) The EU control the use of mercury through the Directive on the Restriction on use of certain Hazardous Substances in electrical and electronic equipment (RoHS). Under RoHS CFLs are limited to 5 mg per lamp. Also, Mercury-containing lamps are one of the many products subject to Waste Electrical and Electronic Equipment Directive. (2) Rules on mercury-containing lamps are subject to both federal and state rules. Federal rules include the Universal Waste Rule. State rules can be more stringent that Federal rules so practice varies from state to state.		

Source: NZIER

4.3.2. International agreements

The Minamata Convention is open for signature in October 2013. Signing the convention does not create legally binding obligations, which apply only after a country chooses to ratify the Convention. Among its provisions is waste management of mercury. It would control the movement of mercury across international boundaries in largely the same way that the Basel Convention controls e-waste movement internationally. Despite the relatively low impact of mercury in our

[22] See for example http://www.cbc.ca/news/canada/manitoba/story/2011/12/30/mb-cfl-bulbs-recycling-ban-manitoba.html

country, New Zealand imports mercury-containing products, and imports and exports mercury and waste containing mercury.

4.4. Implications

Currently the amount of mercury emissions from mercury-containing lamps is relatively small compared to other sources of New Zealand mercury emissions. Around the world the attempts to recycle mercury-containing lambs have meet with only partial success. Product stewardship schemes are in place in Europe and United States but still many mercury-containing lamps are dumped into landfills.

Table 12 Mercury-containing lamps: and issues with the market for diversion

Descriptor	
Heterogeneous preferences	Unknown in New Zealand. Overseas consumers are disposing mercury-containing lamps directly into landfills despite product stewardship schemes being in place.
Non-market values	No non-market studies have been attempted either in New Zealand or overseas
Externalities	The small scale of the problem in New Zealand means that externalities are limited
Prisoner's Dilemma	Most consumers dumping mercury-containing lights directly into landfills
Economies of Scale	No ability to take advantage of economies of scale
Information	Information on disposal available. All disposed directly into landfills
Transaction costs	High transaction costs because of small size of the market

Source: NZIER

5. Tyres

5.1. Introduction

End-of-life tyres pose several problems for disposal (Campbell, Crane, Schaaf, Tracy, & Vo, 2008). In landfills, whole tyres do not remain in place but can rise through the landfilled material to the surface. Also, they provide a space in the landfill in which methane from other decaying material can collect, leading to potential fire risk. Stockpiles of tyres are also potential problems. They pose fire risks and, once alight, they are difficult to extinguish. They also collect water, which can harbour mosquitoes and other insect pests.

There are several uses for end-of-life tyres (Campbell et al., 2008):

- whole, they can be used for retaining walls or to weigh down silage covers
- pyrolysis to produce the following useful resources; carbon black, oil, steel and non-condensable gases.
- they can be turned into rubber crumb, which has several uses, including as an additive for asphalt mix
- they can also be used as fuel – the same burning potential that makes them a hazard in stockpiles can make them useful as fuel.

The end-of-life treatment of tyres varies considerably. The trend is toward greater diversion of tyres from landfills, with Europe, Japan and the US achieving 90 percent rate of recovery or higher (Brindley, Mountjoy, Mountjoy, Council of Australian Governments, & Standing Council on Environment and Water, 2012). Australian recovery rates are much less, around on-half (Brindley et al., 2012; Houghton, Preski, Rockliffe, & Tsolakis, 2004). In New Zealand, 67 percent of the tonnage of end-of-life tyres had an unknown end use (landfill, on farm, illegally disposed) (Tyrewise Working Group, 2012).

5.2. Issues

Table 13 Overview of NZ end-of-life situation for tyres

Descriptor	
Overview	Tyres are hazardous in landfills and large stockpiles. They are potentially valuable as a resource or fuel.
Cost of disposal	Varies. For example, Wellington City Council fee is $4 per car tyre, $10 per truck/tractor tyre.
Cost to end-of-life holder to recycle	Varies. There are several groups, initiatives and companies who will take tyres.
Extent of recycling	32 percent (Tyrewise Working Group, 2012).
Extent of 'dumping'	Unknown.
International commitments	Basel Convention - If another country has banned the import of tyres under the Basel Convention, the EPA will not be able to grant an export permit to that country, and export to that country without a permit will be illegal.
Sufficiency	Current situation is insufficient to promote high levels of recycling. Alternatives to landfill are insufficient in New Zealand. Successful international examples suggest possible alternatives.
Efficiency	New Zealand has some difficulty in achieving efficiencies. First, some reprocessing technologies require considerable scale. Secondly, some end-of-life uses (e.g., shredded rubber) require markets not well developed in New Zealand.
Fairness	Use of tyres is not uniform, either across consumer or across production activities. A user-pays or dumper-pays approach would tend to be more equitable.

Source: NZIER

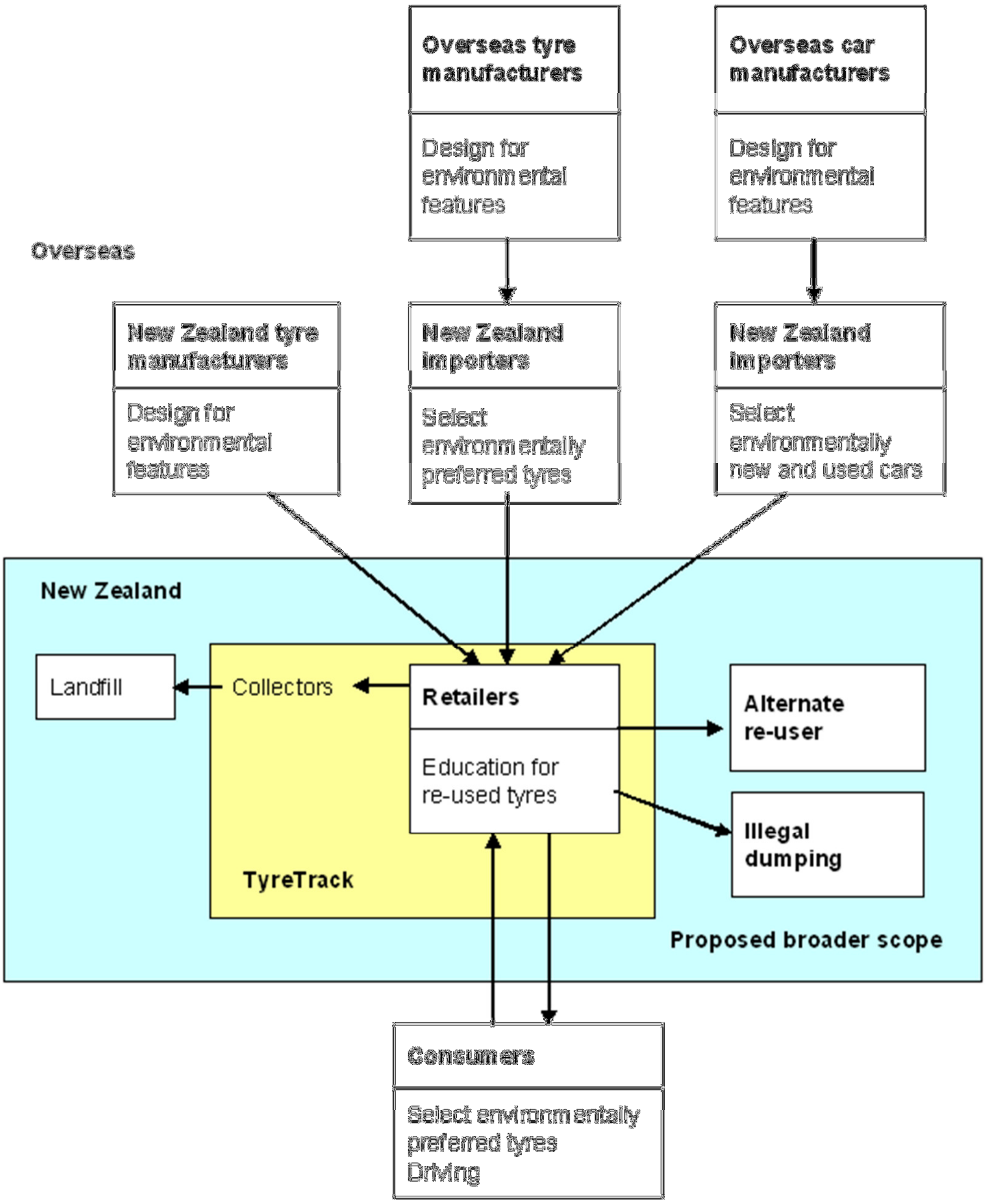
An estimated 4.8 million end-of-life tyres are generated annually in New Zealand (Tyrewise Working Group, 2012). Survey research found that New Zealanders were willing to pay an estimated $2.22 per tyre for recycling (Denne et al., 2007). These figures suggest that New Zealanders are willing to pay $10.7 million annually to recycle tyres.

A considerable focus with respect to tyres has been on the supply of diversion: on reducing landfilling of tyres and increasing the attractiveness of other end-of-life options. For example, URS (2006b) focused on product stewardship, and specifically the role of domestic business in improve end-of-life options. This is seen in the figure from the report below, which excluded the consumer from the scope of the study.

The Tyrewise programme (www.tyrewise.co.nz) is a Product Stewardship Foundation programme to establish a product stewardship scheme for end-of-life tyres. Tyrewise has completed research, development and consultation phases for its scheme.

Figure 3 Tyre product lifecycle

Showing scope of research for URS (2006b) (blue box)



Source: (URS, 2006b)

5.3. WTP

Table 14 Evidence of WTP for tyres

Country	Key findings	Limitations	WTP
NZ	Evidence available.	Single study. Estimated value is lower than disposal fees.	$2.22 per tyre; $10.7m annually
AUS	Fees for dumping are similar to NZ, e.g., $4.40 per tyre (Gold Coast).	No WTP for recycling available.	Unknown.
US	Varies. California has a tyre-management programme.	WTP not estimated; could be higher than fee charged.	$1.75 (California) – actual fee.

Source: NZIER, Denne et al. (2007)

5.3.1. Australia

Ten years ago, Australia was diverting only about half its end-of-life tyres (Houghton et al., 2004). Landfill is a cheap way to dispose of tyres in Australia, so it was often used. Regulations on the treatment of tyres in landfill vary by State (Brindley et al., 2012). In general, there is little use of tyres as fuel (Brindley et al., 2012). Australia has fees on tyres charged at landfills, and the money is used to fund collection and storage of tyres (Campbell et al., 2008). For example, the fee for a passenger tyre in the Gold Coast is currently AU$4.40[23]. Modelling showed that an average fee of about AU$2 per passenger tyre or its equivalent should be sufficient to ensure that only 10 percent of tyres were sent to landfills (Houghton et al., 2004).

A later report (Brindley et al., 2012) stated that

> *Findings from the latest study show there were approximately 48.5 million tyre equivalent passenger units (EPU) tyres entering the waste stream in 2009-10 (compared to 41.8 million EPU tyres entering the waste stream in 2007-08). Of these, approximately 66% were disposed either to landfill, stockpiled, illegally dumped or categorised as unknown compared to 64% in 2007-08; 16 % were domestically recycled compared to 11% in 2007-08 and 18% were exported compared to 10% in 2007-08.*

A comparison of the earlier and later figures suggests that diversion rates have changed little over the last ten years.

In 2013, a decision by the Australian Competition and Consumer Commission cleared the way for a voluntary, industry-led tyre product stewardship scheme.

5.3.2. California, USA

California produces more than 40 million waste and reusable tyres each year ("California. Tire Management Overview," n.d.). CalRecycle estimated that over 80 percent of tyres were diverted from landfills and stockpiles in 2010. The alternative

[23] http://www.greengc.com.au/find-a-service/Council-commercial-waste-disposal-and-recycling-facilities#landfill pricelist.

uses of end-of-life tyres, however, leave a portion of tyres that proper management and stockpiling. To help pay for these programmes, a fee is charged on each new tyre. Starting in 2005, the fee is US$1.75[24].

5.3.3. Hong Kong

In 2008, 75 percent of tyres from Hong Kong were recycled (Campbell et al., 2008). However, landfill capacity is under severe constraint as the existing landfills come to the end of their lives. Researchers found that there was interest amongst many stakeholders in a tyre recycling fee. Members of the public as well as tyre retailers, producers and recyclers were 'open to a tyre levy' (Campbell et al., 2008). The researchers suggested that the best use of the levy would be to educate the public about recycling techniques and to fund collection points.

5.3.4. Summary comparison

Table 15 Tyres: roles in waste diversion

Jurisdiction	Responsibility
Europe	
Belgium, Finland, France, Greece, Hungary, Italy, Norway, Netherlands, Poland, Portugal, Romania, Spain, Sweden, Czech Republic	Producers
Denmark, Latvia, Slovak Republic	Government
Austria, Germany, Ireland, Switzerland, United Kingdom	Consumers
North America	
British Columbia, Canada	Producers
Other provinces, Canada; most states, United States	Government
Some states, United States; Mexico	Consumers
Asia	
Taiwan, South Korea	Producers
Japan	Consumers
Australia	Consumers

Source: (Brindley et al., 2012)

[24] http:/-/www.boe.ca.gov/sptaxprog/tax_rates_stfd.htm#2.

5.4. Implications

Table 16 Tyres and issues with the market for diversion

Descriptor	
Heterogeneous preferences	Does not appear to be a major issue.
Non-market values	An issue to the extent that tyre stockpiles represent lost amenity values.
Externalities	Tyre stockpiles create externalities by harbouring pests. Uncontrolled burning of tyres creates pollution. Abandoned tyre stockpiles costs landowner/Council to clean up
Prisoner's Dilemma	Tyres hidden in loads of mixed waste create problems for landfills.
Economies of Scale	Considerable issue with tyres, both for reprocessing technology and for outlets for the reprocessed materials.
Information	Possible problem – verifying whether tyres are actually being recycled/reprocessed.
Transaction costs	Potentially low. The majority of end-of-life tyres are the result of tyre replacement, so the diversion can be part of an existing transaction.

Source: NZIER

The frameworks set out earlier are useful for understanding tyre disposal. The economic-institutional-political framework demonstrates why there is momentum for diverting tyres from landfill. Economically, tyres have clear value as a resource that can be unlocked with current technology. Institutionally, the interests of stakeholder firms are generally aligned. Tyres are problematic for waste management firms – they are problems in landfills and tyre stockpiles are hazardous. Tyre producers and distributors are interested in demonstrating social and environmental responsibility through good product stewardship. Politically, there is domestic pressure to reduce the use of landfill and make good use of resource, while international 'best practice' is aiming for 90 percent or greater recovery. Thus, all three sets of drivers are aligned.

The market assessment of tyre recovery indicates the strength of the economic drivers. The supply of alternatives to landfill is generally strong. There are several alternative uses for tyres that are economically viable. The demand for alternative is also clear: research in New Zealand and overseas as well as actual fees charged for disposing of tyres demonstrates that people are willing to pay a few dollars per tyre for proper handling. In New Zealand, the aggregated willingness to pay can be estimated at $10.7 million annually.

6. Agricultural chemicals

6.1. Introduction

Agricultural chemicals, as discussed with the 2013 Global Chemicals Outlook, are the fertilisers and pesticides (comprising herbicides, insecticides, fungicides and others) used within pastoral, cropping and horticulture farming systems. There are also the chemicals used by veterinarians. These aforementioned products do not have an End-of-Life use. There is high temperature incineration that produces energy for cement production but this is more an efficient disposal method rather than recycling.

Within the scope of the New Zealand Agrecovery Rural Recovery programme, there are also other associated products such as plastic wrap and the plastic chemical containers. These products can be recycled into other plastic products.

6.2. Issues

The issues with agricultural chemicals largely relate to the use and disposal of the chemicals, namely the potential for harm to the user and more widely to other people, and to animals, the environment and New Zealand's export trade. Some of New Zealand's responsibilities are dictated under international conventions.

Table 17 Overview of NZ end-of-life situation for Agricultural chemicals

Descriptor	
Overview	All ag chems imported. New Zealand Agrecovery Rural Recovery (NZARR) programme collects chemicals and plastics; plastics are recycled, chemicals are destroyed her and overseas. Otherwise chemicals go to urban or on-farm landfills.
Cost of disposal	NZARR accepts chemicals for free for member brands at depots around NZ. Voluntary disposal levy on imported bulk chemicals. End users must pay for all other non-member brands of chemicals to be collected and destroyed.
Cost to end-of-life holder to recycle	Not applicable
Extent of recycling	Only relevant for plastics. Believed to be low.
Extent of 'dumping'	Believed to be high, but unknown (GHD Limited, 2012). The traditionally accepted disposal methods for agricultural waste, including agricultural chemicals, have been burning, burying (farm pits) or bulk storage.
International commitments	**Stockholm Convention:** New Zealand to collect and destroy polychlorinated biphenyls (POPs) **Basel Conventions**: Permit required to ship hazardous material offshore
Sufficiency of current schemes	Risk exists that NZARR funds may be insufficient to collect /recycle/destroy end-of-life chemicals although obligation limited to member brands.
Efficiency of recycling schemes	To be determined.
Fairness of current schemes	Brand scheme excludes non-payers but that leaves externality existing (and discourages volunteers)

Source: NZIER

From an end-of-life perspective, the challenge is to

- safely and efficiently collect and dispose of chemicals
- set pricing and provide information that encourages efficient and safe disposal (and use)
- fairly share the costs of disposal.

From a whole-of-life perspective, the challenge is to

- reduce inefficient use of chemical products, both at the production stage and during use
- reduce the external impact of any chemical use
- encourage safe handling.

The current voluntary system for agricultural chemical collection for recycling or safe disposal does not have large coverage, so that potential harm remains for many chemicals; this is both unfair and inefficient. The Agrecovery programme does not have the risk of accepting non-member end-of-life products, thus reducing its financial risk, but it does remain exposed to the risk that collection and disposal costs increase significantly.

6.3. WTP

There are no studies found that examine the WTP of farmers, or other end-users, to pay for recycling or disposal. There are more general WTP surveys that indicate a willingness amongst the wider public to pay for reduced chemical use and there are patterns of behaviour that point to both willingness and unwillingness amongst different farmers.

Table 18 Evidence of WTP for agricultural chemicals

Country	Key findings	Limitations	WTP
NZ	No survey found of WTP by product importers or end users. Some brands volunteering to pay at present.	It is not known why some brands do not volunteer for scheme i.e. what is behind their unwillingness.	-
AUS	No survey found of WTP by product importers or end users.	-	-
US	No survey found of WTP by product importers or end users.	Diverse arrangements between US states creates opportunity to observe variations in willingness but not such study found.	-
EU	No survey found of WTP by product importers or end users.		-
Surveys addressing pollution in general (rather than in end-of-life product)			
US	Rural households WTP for less groundwater contamination	Relates to general use, not just disposal	Equivalent to US$ 197-730 / annum
UK	Household WTP more for bread produced in less damaging manner	Budget constraint at point of sale may also be of influence	Extra 1% / loaf to avoid 1-in-100 human harm
Italy	Household WTP more for food produced in less damaging manner	As above.	EUR 15 / month for 1% less pesticide contamination
Korea	Household WTP higher taxes to increase plastic ag. wrap recycling from 30% to 60%	Included large proportion (60%) who were unwilling to pay	US$ 3-5 / annum for 5 years

Source: NZIER

6.3.1. Farmer attitudes

The varied response of farmers in different situations and surveys provides evidence of the mixed attitudes amongst farmers to disposal (and use) of chemicals.

The 'free-riding' noted by the MfE in New Zealand voluntary Agrecovery scheme is a general challenge. It is a wider problem also noted by the UNEP (2013). Although the UNEP did not determine why voluntary schemes were achieving low recycling rates across jurisdictions, the failure to volunteer was couched in two dimensions: "for lack of capacity or willingness".

Research into plastic containers also implies a significant proportion of farmers have either low or no capacity or willingness to pay for recycling. Studies show that farmers are sensitive to the recycling fee (Levitan et al, 2005), that convenience is important (Levitan et al, 2005; Gilbert, 2001) and that regulation such as a ban of burning increases recycling (Negra and Rogers, 1998).

Turning to pesticide use generally, there is support for the case that a lack of information may be one factor behind any low willingness to pay for disposal. Hasing (2010) found US farmers were sensitive to labelling, being willing to pay US$27/acre to avoid use of a herbicide that included a label with the words 'Warning' and US$38/acre with a 'Danger' label.

The importance of information can also be inferred from the low price elasticity of pesticide demand found by Skevas et al (2010) in the EU, given an earlier finding (Norgaard) that the major motivation for pesticide application is the provision of some insurance against damage (i.e. to manage uncertainty).

6.3.2. Public attitudes

There have been numerous studies looking into the willingness of the general public to pay to reduce pesticide use or increase recycling. Skevas (2010) considers there are sufficient studies and insufficient sensitivity to local incomes to be able to infer a consumer willingness to pay for reduced pesticide use. These findings are consistent with the existence of an externality effect of agricultural chemical usage on the average consumer.

A selection of studies include:

- Travisi (2004) who found an Italian household WTP of EUR15/month on the food bill to avoid 1% soil and water contamination due to pesticides, EUR24/month for one less bird loss, and EUR3/month for one less human ill-health (note, these figures capture both expectations of consequence and probability);
- Kwak (2004) who found a South Korean household WTP higher taxes equivalent to US$3-5 per annum for 5 years to increase agriculture wrap recycling from 30% to 60%;
- Foster and Mourato (2000) who found a UK household WTP of an extra 1% for bread to avoid 1-in-100 cases of human illness and 9% to protect 1-in-9 farmland bird species;
- Crutchfield et al. (1995) who calculated an WTP by rural households of US$197-730 million per annum to protect groundwater from chemical contamination.

Interestingly, within these 'average' WTP there can be large proportions of people who were unwilling to pay, again indicative of the mixed attitudes in society. For example, Kwak et al (2004) had 60% say 'no' to higher taxes.

6.4. Implications

Table 19 Potential benefits not captured for end-of-life agricultural chemicals

Descriptor	
Heterogeneous preferences	Variation in WTP
Non-market values	Potentially large as the future generation cannot price damage that they might incur, including potentially irreversible effects.
Externalities	WTP surveys show externalities exist, although issue largely relates to use of products.
Prisoner's Dilemma	Free-riding is evident but not clear whether due to gaming or lack of perceived value.
Economies of Scale	Economies of scale likely with incineration plants but limited opportunity with collection.
Information	There is likely to be incomplete information about harm to the user and the user's immediate circle (people and environment), leading to the under-valuation by individuals of WTP for safe use and disposal.
Transaction costs	There are significant costs to transport chemicals to collection sites (either by the end holder or the collector). Costly to ascertain information.

Source: NZIER

The Australian Productivity Commission (2008) noted three market failures relating to environmental risks with chemicals. These failures represent benefits that are not being taken into account in market transactions, and potential benefits that may justify higher levels of, in this case, safe waste disposal.

A noteworthy market failure relates to information: chemical users may be unaware of the risks that chemical use poses to the environment; the complexity of risks may mean that management of risks is not properly conducted; and information may be costly to obtain.

The second two market failures concern externalities, the first resulting from use and the second resulting from waste disposal. These externalities will relate to the current generation and yet-to-be-born generations.

Not so much a market failure but a barrier to trade is the high costs of waste collection and disposal. In many cases these waste collection and disposal costs are reduced by scale. These economies of scale will be difficult to capture without coordination.

7. Refrigerant gases

7.1. Introduction

Gases such as CFC, HCFC and HFC have been used as refrigerant gases in household fridges, commercial chillers, home and industrial air conditioning systems, and air conditioning systems within vehicles. These gases are generally toxic, and many have either ozone or greenhouse impacts.

There is no refrigerant manufacturing sector in New Zealand. Gases enter the country in bulk or within existing products, including new and used cars. Chillers perform a significant role in the dairy, meat, seafood, wine and horticulture export industries.

Refrigerant gases can be reused, after being recovered, recycled or reclaimed. Recovery is simply the extraction and transfer of the gas for reuse. Recycling involves a basic cleaning of the gas before reuse. Reclamation entails a quality controlled cleaning to a high standard of purity, and then reuse.

Otherwise safe disposal is by way of high temperature incineration.

A voluntary scheme exists in New Zealand to collect, store and dispose of ozone depleting substances and greenhouse gases, including refrigerants. Disposal is via export to an incineration plant in Australia. Funding is provided by a voluntary levy collected by the scheme on imports (e.g. $1.50/kg of HFC or HCFC in 2009 (Hennessy, 2011).

The global demand for commercial refrigeration equipment is forecast to rise by 4.6% annually until 2016 (Freedonia, 2012). Strong growth in dairy production and strong population growth is likely to see strong growth in refrigerant demand in New Zealand.

7.2. Issues in New Zealand

Table 20 Overview of NZ end-of-life situation for refrigerants

Descriptor	
Overview	All refrigerants imported. Bulk imports of HFC/PFCs 411 tonnes in 2008; pre-charged imports 317 tonnes. Plus within insulation foam and aerosols. (Hennessy, 2010) Trust for Recovery of Synthetic Refrigerants, working as Refrigerant Recovery NZ Ltd (RRNZ), collects waste refrigerants, extracted from end-of-life products, for export to Dascem/Coffey Environments in Australia for destruction. Exports were approx. 15 tonnes of HFC in March year 2011/12. Also exported similar amount HCFC and approx. 1 tonne CFC. (www.refrigerantrecovery.co.nz) Otherwise refrigerants remain in products and when products go to landfills or are recycled the refrigerants can be discharged into the atmosphere.
Cost of disposal	RRNZ accepts gases for free within approved containers at depots around NZ. Disposal levy on imported bulk refrigerants of $1.50/kg HFC or HCFC in 2011, paid to Trust (Hennessy, 2011) Levy from Jul-13 on importation of goods containing SGG of around $5-15/kg; vehicles paid to NZTA, equates to around $3.50/car, $4.70/truck and $23.50/bus (www.nzta.govt.nz); others paid to Customs (www.customs.govt.nz) Cost to dump fridge or chiller at landfill at typical landfill costs of $0-400/tonne The Emissions Trading Scheme (ETS) provides an incentive for export or destruction of synthetic greenhouse gases through tradable ETS units.
Cost to end-of-life holder to recycle	To be determined
Extent of recycling	A relatively low proportion of used refrigerants is collected for recycling or destruction in New Zealand compared with Australia. (CRL, 2008)
Extent of 'dumping'	100% of HFC assumed emitted at end-of-life for household fridges for ETS/IPCC accounting (fridges have half-life of 20 years) 27% of HFC emitted due to end-of-life vehicles (CRL, 2008)
International commitments	Kyoto Protocol requires control of Synthethic Greenhouse Gases (SGG) emissions, viz. HFCs, PFCs, SF6. • SGG are included in New Zealand's ETS • The Climate Change Response Act (CCRA) 2002 bans the wilful release of SGG, with fines up to $50,000. (www.climatechange.govt.nz) Montreal Protocol requires phase out of Ozone Depleting Substances (ODS) • Ozone Layer Protection Act (OLPA) bans bulk imports, but not ODS imports within goods (www.climatechange.govt.nz). The OLPA makes it an offence to release controlled substances knowingly during installation, servicing, operating or dismantling equipment.
Sufficiency	Risk exists that Trust funds (Trustee reported accumulated funds of $5.3m as at Mar-12) may be insufficient to collect/export end-of-life gases
Efficiency	Greater efficiency achievable through design changes, and higher reuse of existing refrigerant
Fairness	Bulk importers of HFC/PFC disadvantaged as non-bulk importers (317 tonnes) do not pay disposal levy

Source: NZIER

Refrigerants such as CFC are a well-known ozone-depleting substance (ODS), as well as having high global warming potential (GWP). While bulk CFCs were no longer imported into the country after 1996, and bulk HCFC will no longer be imported after 2015, these gases remain in existing products and can be imported within products.

Some other refrigerants still in use also have a greenhouse impact e.g. HFC-134a as used in mobile air conditioners still has about one-eighth the greenhouse effect of CFC-12.

Alternative natural refrigerants such as ammonia, carbon dioxide or mixtures of hydrocarbons have no or negligible contribution to the greenhouse effect.

There are likely to be releases of damaging refrigerant gases to the atmosphere in New Zealand by holders of gas-containing products although the majority of release is believed to be by accident (Henessy, 2011).

The other local issue is that the voluntary disposal scheme does not cover all importers.

From an end-of-life perspective, the challenge is to

- encourage safe reuse of gases
- safely and efficiently collect and dispose of gases eventually,
- set pricing and provide information that encourages efficient and safe disposal (and use), and
- fairly share the costs of disposal.

From a whole-of-life perspective, the challenge is to

- reduce inefficient use of the gases, both at the production stage and during use
- reduce the external impact of any gas use
- encourage safe handling.

7.3. WTP

We were unable to find any studies that estimated the WTP of consumers, households or others to recycle refrigerants. There are broad studies that imply that people in general are willing to pay to reduce the damage that refrigerants can cause but there is also widespread evidence of avoiding payment, for reasons unknown but likely to relate to uncertainty and unwillingness to pay for external effects.

Table 21 Evidence of WTP for end-of-life refrigerants

Country	Key findings	Limitations	WTP
NZ	No survey found of WTP by importers or households. Bulk importers volunteering to pay $1.50/kg at present.	Unknown why some bulk importers do not volunteer for scheme.	-
AUS	No survey found of WTP by importers or households. RRA recovering around 50% of annual ODS from end-of-life refrigeration and air cons.	Unknown what is WTP as import license holders are required to be part of approved product stewardship scheme, currently paying $2/kg	-
US	No survey found of WTP by importers/manufacturers.		-
EU	No survey found of WTP by importers/manufacturers.		-
Surveys addressing emissions in general (rather than in end-of-life product)			
AUS **US**	Householder WTP in various countries to reduce GHG emissions	Figures inferred from equivalence of well-being due to income and well-being due to GHG reductions, rather than an explicit payment indicated or made.	AUS: US$ 54 / annum / ton GHG reduced emission US: US$ 1301 / annum / ton
US	US householders WTP extra for fridge to reduce GHG emissions by 17% by 2020		US$ 60 / annum
US	US householders WTP extra for fridge that reduces environmental impacts	It is unclear from study just what the extra WTP actually related to	US$ 96 / fridge
US **Korea**	Households WTP extra taxes to prevent climate change	The success or failure of 'prevention' was not considered	US: US$ 13.70 / month Korea: US$ 1.60 / month

Source: NZIER

7.3.1. General WTP studies

Whilst there were no specific studies found that consider the WTP to pay to 'recycle', there are studies in the household WTP to reduce emissions, including by inference refrigerants in one case:

- Beja (2012) in a Subjective Well-Being study put the US WTP at US$1301 per ton per capita per annum, and at US$54 in Australia
- Kotchen et al (2012) calculated a US WTP greater than US$60 per annum to reduce greenhouse gases by 17% by 2020
- Ward et al (2010) found the WTP for an ENERGY STAR fridge was US$96 per fridge greater than the energy savings described to respondents, implying a benefit up to this value derived from the reduced environmental impact of refrigerants.

Studies also exist that estimate the household WTP to prevent climate change:

- Lee (2010) put the Korean WTP at GBP 1.60 per month
- Berk (1999) put the US WTP at US$ 13.70 per month.

There are within the aforementioned studies significant proportions of respondents who were unwilling to pay.

7.3.2. Programme details

There are various schemes operating internationally to recycle and dispose of refrigerant gases. Only some schemes, though, encourage reduced use of dangerous refrigerants.

Table 22 Schemes for managing end-of-life refrigerant gases

Jurisdiction	Responsibility
Europe	2002 ELV Directive requires manufacturers to take responsibility for ELV (has not provided full coverage) EPR for fridges under WEEE Directive – high recycling rates but high costs
Denmark	EU Plus a deposit refund scheme
Norway	EU Plus tax importer/producer on CO2-equivalent then refund on destruction
United States	US varies by State for air conditioners, fridges, vehicles Generally low recycling rates with most gas destroyed
Japan	2005 ELV Recycling Law mandates vehicle recycling and appropriate disposal funded by an at-registration fee 2001 Home Appliance Recycling Law mandate retailer collect end-of-life product on delivery of new product, with fee charged to consumer at time
Australia	Mandatory to manage end-of-life refrigerants (Refrigerant Reclaim Aus.). Plus, the Federal govt pays $1.5/kg for ODS destroyed from 1 Jul 2013

Source: NZIER

Nicol (2007) states that extended producer responsibility (EPR) schemes are superior:

> *Mandatory EPR programmes that target specific recovery and recycling rates are effective in reducing waste and driving DfE changes for consumer products, such as switching from HFCs to hydrocarbons as refrigerant. In contrast, product stewardship externalizes end-of-life costs and provides no incentive to prevent the generation of waste during the design stage and no regulation to reduce emissions or increase recycling rates.*

7.4. Implications

Table 23 Potential benefits not captured for end-of-life refrigerants

Descriptor	
Heterogeneous preferences	Not noted.
Non-market values	Potentially large as the future generation cannot price damage that they might incur, including potentially irreversible effects.
Externalities	WTP surveys show externalities exist, although issue relates to use of products (leakage) as well as end-of-life.
Prisoner's Dilemma	Free-riding is evident but not clear whether due to strategic behaviour or lack of perceived value.
Economies of Scale	Having access to Australia destruction plant enables economies of scale. Some economies for centralised collection points but limited for most of NZ.
Information	There is likely to be incomplete information about harm to the user and the user's immediate circle (people and environment), leading to the under-valuation by individuals of WTP for safe use and disposal.
Transaction costs	There are significant costs to transport gases to collection sites (either by the end holder or the collector). There are also significant costs to extract gas from end-of-life products.

Source: NZIER

As with agricultural chemicals, there are potential benefits in the product cycle that are potentially not being internalised:

- There is likely to be incomplete information about harm to the user and the user's immediate circle (people and environment), leading to the under-valuation by individual of WTP for safe use and disposal;
- And there is the externality on current and future generations of the release of harmful gases into the wider environment, including irreversible effects.

Similar, there are potential economies of scale around the collection, reclamation and/or disposal that may not be realised without coordination.

8. Benefit transfer

The building blocks of modelling the value of waste reduction revolve around the policy questions, the approaches taken, and the data.[25] The key problem highlighted by this report is the lack of data to support our national understanding of the values consumers place on recycling the various waste streams. One way of avoiding this problem is to use benefit transfer.

8.1. What is benefit transfer?

Benefit transfer is a technique in non-market valuation of identifying WTP estimates derived in one situation (the source study), then transferring those estimates to a new situation (the destination study). For example, a piece of research might estimate that people in one area are willing to pay $10 each for a clearer lake. That estimate might then be used as an indication for how much people somewhere else are willing to pay for a cleaner stream. In one sense, this is an unremarkable process: economist and other researchers rely on existing data all the time to make judgements about other contexts (Morrison, et al. 2002). The difference is that non-market valuation is sensitive to the things being valued and the people doing the valuing, so there are questions about sensitivity of benefit transfer estimates (Morrison, et al. 2002). This is particularly so in waste streams since different jurisdictions face different waste issues and these need to be fully investigated (see section 4.2).

Because of the sensitivity, the validity of benefit transfer has been studied. Morrison, et al. (2002), for example, investigated different ways of assessing the validity of benefit transfer across several different sites. They found that prices implied by the different models could generally be transferred across sites. However, the total values placed on environmental impacts in two different sites were not equivalent because of differences beyond the environmental attributes measured. Rolfe, Loch and Bennett (2006) conducted a similar test, and again found that some prices could be validly transferred. They also found that total values could the transferred in some circumstances. Rolfe et al. (2006) suggested that two things are required for benefit transfer to be valid: the source study needs to be free of major biases, and the transfer process itself needs to avoid introducing biases e.g. to avoid biases New Zealand policy makers are more likely to rely on waste recycling estimates in countries with similar situations to New Zealand.

Much of the work on benefit transfer has been undertaken in the context of environmental economics. Some effort has been made to transfer the learnings to health economics (Hanley, Ryan and Wright, 2003). There does not appear to be any such work in the context of waste management.

25 For a fuller discussion see Nixon and Yeabsley (2013) p13.

8.2. What are the issues?

8.2.1. Sites/products

The first main issue with benefit transfer is ensuring that the source study and the destination study are measuring the same thing, that we are comparing apples with apples (Rolfe, et al. 2006). With environmental studies, it is important that the scale of the impacts and the location being studied have similar characteristics (or can be made similar). In the waste management context, studies should focus on the same products and the same alternative treatment of end-of-life products. More generally, the situations should have the same context. Crucially for end-of-life products is the availability of landfill space or areas for new landfills. Some jurisdictions are constrained in the availability of land and/or landfills. This constraint has been mentioned in the context of Hong Kong waste minimisation efforts, for example. Other jurisdictions, such as Australia and parts of the United States, have much lower population densities and thus face lower land constraints.

Therefore the Australian URS (2009) study on e-waste which suggests that consumers are willing to pay up to A 50 cents per unit for a 1% increase in e-waste diversion is probably more representative than estimates from California and Macau where landfill pressures are more intense.

8.2.2. Populations

The second major issue with benefit transfer is ensuring that the populations studied are similar. Populations are expected to have differences in their preferences, and indeed subsamples are shown in choice modelling exercises to have heterogeneous preferences. It is important to consider the impact of respondent characteristics on the WTP values estimated in the source study, and then adjust for the destination population.

Populations also change preferences over time and react to various events (such as the TV and other media advertising campaign on the digital switch over in New Zealand). Potentially such campaigns can further heighten awareness of (in this case) e-waste. Combine this the general impact that consumers as they become richer are more concerned with environmental issues means that it is possible that:

- The studies conducted two to five years ago underestimate the current willingness to pay
- The general concern for environmental issues is increasing and these willingness to pay figures will increase since the problem is unlikely to go away.

8.2.3. Framing effects

Rolfe et al. (2006) also point the importance of framing effects. Research by such people as Paul Slovic and Daniel Kahneman has established that the way questions are framed – the words used and the way effects are described – can have a statistically significant effect on results. This effect has also been identified as an issue with non-market valuation (Bateman, et al. 2002).

With end-of-life products, framing effects are likely to be important. In particular, the way in which the 'waste problem' is presented is likely to affect the values that individuals place on alternative to landfills.

8.3. Existing techniques

There are essentially two techniques for benefit transfer. The first technique, seen in the Morrison et al (2002) and Rolfe et al (2006) studies already discussed, is to make the source and destination studies as similar as possible. They should investigate similar effects, include samples with similar socio-demographic characteristics, and strive for similar framing of the issues being studied. A second approach is meta-analysis (Abdullah & Rosenberger, 2012; Bergstrom & Taylor, 2006; Brander, Van Beukering, & Cesar, 2007; Brons, Pels, Nijkamp, & Rietveld, 2002). Meta-analysis is a technique used for many purpose across several disciplines (Poot 2012). It combines the data and estimated variables from a number of studies and finds an overall analysis that accounts for all the studies. It is in a sense an estimate of estimates. The strength of meta-analysis is that it is a systematic way to find the common values estimated across a number of studies.

8.4. Benefit transfer for end-of-life products

Table 24 Products

End-of-life product	WTP data
E-waste	URS (2009) A$0.50 cents per item for a 1% increase in e-waste diversion Nixon and Saphores (2007) in California extra 1% on the purchase price for improved e-waste disposal Wisconsin (2006, 2010, 2011) up to $US5 to have electronic devices recycled Macau $US 2.50 per household for e-waste
Batteries	None
Mercury-containing lamps	None
Tyres	New Zealand $2.22 per tyre (survey) California US$1.75 per tyre (actual)
Agricultural chemicals	No specific studies
Refrigerant gases	No specific studies AUS: US$ 54 / annum / tonne GHG reduced emission US: US$ 1301 / annum / tonne GHG reduced emission

Source: NZIER

The first thing required for a benefit transfer exercise is data. Of the six end-of-life products, three have no studies available in the international literature. Two additional products have only one or two values that can be derived from the literature. The only product with multiple estimated values is e-waste, for which we were able to find several estimated. Unfortunately, the studies measure different things. The Wisconsin studies, for example, focused on the recycling of specific products. The URS studied valued a change in the level of e-waste, while the California study considered a general ‘improvement’ in e-waste disposal. These differences are a combination of product differences – what end-of-life product/treatment is being considered – and framing differences – how the e-waste treatment is presented.

We discussed two methods of benefit transfer: simply moving values from a source estimate to a similar destination estimate, and meta-analysis. The data available do not appear to be robust enough for either approach. First, there are differences between the source populations and destination population that will have unknown impacts on the results. Secondly, the specific end-of-life products being valued – particularly in the case of e-waste – are too variable to attempt a meta-analysis.

We are therefore left with the following:

- For tyres, there is an existing New Zealand WTP estimate that is of a similar magnitude to disposal fees in New Zealand and elsewhere
- For e-waste, there is the suggestion that consumers will pay some small amount for improvements in end-of-life diversion options, but the exact amount will depend on the product and the diversion
- For all other products, we are unable to estimate WTP for end-of-life options.

8.4.1. Potential biases from benefit transfer

The benefit transfer approach can lead to biased estimates of willingness to pay. Two main sources of bias are bias in the source studies, and bias introduced by transferring the values.

Bias from sources studies needs to be evaluated on a case-by-case basis. Examining the key estimates identified in this report as good sources of WTP data, we can make the following observations about bias:

- The URS (2009) study is very good and should be relatively free from bias. The survey was conducted using quotas for different demographic targets, which would contribute to making the survey representative of the target populations (Adelaide, Brisbane and Melbourne). The analysis conducted using a sensitive model that controlled for various characteristics of the population and the decision situations[26]
- Nixon and Saphores (2007) is likely to represent a biased estimate. The mail survey clearly produced a sample that was not representative of the California population. As they note, ‘Almost all of our respondents are college-educated with household incomes greater than $40,000’ (p. 15), which they attributed to the populations of the specific counties where they surveyed. In addition, the estimation method (ordered logit) was appropriate for the design of the questionnaire, but better overall design would have produced a more accurate estimate of WTP

[26] The study used a discrete choice survey. The data were modelled using a random parameters logit, a sophisticated econometric technique for analysing discrete choice data that allows researchers to investigate the diversity of values expressed by people in the sample.

- The Wisconsin studies (2006, 2010, 2011) are difficult to interpret, because they surveyed residents about WTP for e-cycling, a generic term. It is not clear whether respondents were focused on televisions, computers, mobile phones or other electronics, and whether WTP varies across products. A further weakness is that the studies used a rudimentary method to capture data on WTP, and the analysis was similarly basic. Bringing the results across to New Zealand is not likely to produce any special bias (beyond the general question of comparability): results were disaggregated by demographic variables and could be adjusted to account for differences in age, income and homeowners between Wisconsin and New Zealand
- The Macau study (2012) was similarly generic, asking about WTP to recycle e-waste, which could produce biased estimates if applied to specific products. In addition, Macau has a wealthier population and a much higher population density, two factors that are likely to increase the WTP for alternatives to landfills. Transferring the WTP figures to the New Zealand context is likely to create biased results
- For tyres, the data from California substantiate the results from survey research in New Zealand. Although there are differences across the two populations, the similarity of the WTP figures provides some comfort that the New Zealand figure is realistic
- The refrigerant gases estimates are likely to lead to biased estimates. First, the calculation relies on several steps and a causal link between WTP for greenhouse gas reduction and WTP for end-of-life treatment of refrigerant gases. This calculation is likely to introduce some unknown bias. Secondly, the two values identified from Australia and the United States are very different from each other, making it difficult to arrive at a value for New Zealand.

Overall, the URS (2009) study on e-waste and the New Zealand estimate for tyres are likely to be appropriate for estimating the value of end-of-life programmes in New Zealand. The other studies either have weaknesses in their methods that are likely to lead to biased estimates, or raise questions about the transferability of results to the New Zealand context.

9. Conclusion

This study explored the international literature to determine where there might be unrealised benefit in current treatment of six end-of-life products, with a focus on willingness to pay studies. We examined the products in the context of the New Zealand Waste Strategy and the Waste Minimisation Act 2008. In theory there are several ways in which markets may not capture all benefits, as discussed in Appendix B; evidence was sought to show whether these benefits might exist in New Zealand.

In general the international literature is incomplete with regard to the specific willingness to pay for the end-of-life products of this study.

One product where benefits in New Zealand can be reasonably inferred is e-waste. In particular an Australian conclusion that consumers were willing to pay extra for reduced waste is likely to be similar here in New Zealand. Further support for this inference comes amongst the many studies that sections of the general public are willing to pay extra for products and practices that are less harmful to people and the environment, including for refrigerant gases and agricultural chemicals.

Another product with information on WTP is tyres. Estimates of WTP for tyre recycling in New Zealand and California are available.

It is also important to understand the value of resource recovery for end-of-life products. WTP studies offer little insight into this value extraction, which is more related to technological issues. However reports pointing to the difficulties and costs of extracting resources from e-waste, batteries, mercury, tyres and refrigerants highlight not only the challenge with resource recovery but the informational challenges that consumers face when 'buying' recycling services. People may value less harm to others and the environment, they may be uncertain as to whether the recycling system will achieve this aim.

International practice tends to suggest that there are advantages to imposing the responsibility for recycling on the original producers, which is the product stewardship approach. This approach both serves to internalise part of the externality, plus provides an incentive for product design to take better account of disposal. However, New Zealand is a small market with less leverage, as discussed in section 10.4.

In sum, there is a general willingness to pay to reduce harm but information costs and technological constraints appear behind generally observed low levels of recycling.

10. Recommendations

10.1. The importance of data

Recommendation 1: Well-structured willingness to pay research should be conducted for selected products to inform durable waste policy in New Zealand.

To inform durable policy approaches, the economic advice needs to be balanced. Balance requires asking the right questions, using robust approaches (theory) and having suitable data. The key issue for environmental policy (as well as waste policy) is the lack of data in the New Zealand context. As Leamer (1996, p175) suggests, if we forget data and rely on policy questions and theory, we are replacing potential evidence with 'hunches' or opinions. Without reliable data the chances of producing durable public policy are greatly reduced.

Further, if economics is not factored into waste solutions alongside the political and institutional processes/drivers (see Figure 7), then solutions will be found but they are likely to be suboptimal and at some stage will have to be revisited.

WTP is useful if it can provide the data to inform the economic value of different types of waste disposal.

We would tend to recommend survey-based ('stated preference') methods for this research. Such methods allow testing of a number of potential waste-reduction scenarios, exploring the attitudes and beliefs that support willingness to pay findings and investigating demographic effects.

Potentially, other methods could be used to determine willingness to pay. These include:

- market value approaches that infer value using market-based prices, such as the value of the next best way of achieving a given outcome (e.g. the costs of running a waste disposal service and landfill operation)
- revealed preference approaches that infer value of a non-market resource from observed behaviour with regard to related activities, for instance inferring the value of waste diversion by investigating how people store end-of-life products at home.

These methods focus on what we might observe in the market. In many cases, these approaches provide good proxies on what individuals and society are prepared to pay for goods such as flood protection. However, they may provide limited understanding of non-use values in the waste area (see Appendix B.2).

10.2. Few studies of use

Recommendation 2: Overseas estimates for willingness to pay can be used in New Zealand for some products, such as tyres and (initially) e-waste. However further New Zealand based work is required to confirm WTP and understand the underlying dynamics (i.e. is the willingness to pay by consumers increasing or static over time?)

In many of the waste streams covered there is little international work that reports reliable WTP estimates. However, in two cases (tyres and e-waste) WTP estimates are available.

More information would clearly assist policymakers. Some specific areas include:

- e-waste is a large, differentiated category with lots of potential for hazardous waste and resource recovery. We suggest further research on WTP to confirm estimates and further understand how these estimates change over time. Given preliminary work in Australia, an e-waste point of sale levy of between 0.5% and 2% per item is appropriate
- batteries: it is possible that landfilling of household batteries represents an efficient method of handling end-of-life batteries. Further study would determine the extent to which people would like an alternative to the general waste stream
- for mercury-containing lamps and other mercury-containing end-of-life products (e.g., batteries) we suggest that policymakers build on the mercury inventory (Chrystall et al., 2009) and a mercury life cycle assessment (Garrett & Collins, 2009) to develop a WTP survey that would provide estimates of public concern and appropriate levels of intervention.

Areas of research that might be less valuable:

- tyres: indications of WTP are already available and considerable work is already being done. Technical issues still need to be resolved around resource recovery and design of product stewardship. Thus, further research would tend to be valuable if improved estimates of WTP are required to demonstrate the value of planned interventions. Current estimates are $2.22 per tyre in New Zealand
- agricultural chemicals: they are also subject to existing programmes, and a highly differentiated category. Likely to benefit from product stewardship for new chemicals, and subsidised disposal for legacy chemicals. Individual WTP is less an issue for this end-of-life product
- refrigerant gases: effects are global, so these products are best handled with appeal to international obligations.

10.3. Further work on programme design

Recommendation 3: Further work should be conducted on implementation design.

Programme design of product stewardship arrangements is an area where further research is potentially useful. This is because there is a disconnect between willingness and action. There are many volunteer stewardship schemes reported but also a general low level of recycling within the same jurisdictions. Even legally imposed bans on dumping can have poor outcomes, such as the 90% of mercury still dumped in Californian landfills. The Australian solution for e-waste was to adopt mandatory membership of recycling schemes by manufacturers and importers.

The WTP literature does not provide a definitive answer as to why volunteer schemes have low effectiveness. One factor is likely to be information. There is evidence that people will respond to explicit warnings about products, suggestive that they were not fully aware beforehand of risk to themselves. There is evidence of peer pressure, suggesting that information is either difficult to acquire and/or that impacts are uncertain and hence people resort to popular opinion.

There are several information problems with the products investigated. Many products contain newer technology – e.g. cell phones, cadmium batteries, mercury lamps – where an element of learning is still occurring. In addition, product categories, such as 'chemicals', can be heterogeneous, which makes it difficult for people to differentiate risks. Some products, e-waste in particular, can include many components, making them complex to recycle. Finally, some harms are typically not immediately apparent but are only perceived in the future.

While some people do express a willingness to pay to reduce waste, others do not. The drivers of this unwillingness to pay are not clear, especially whether they are driven by general attitudes about waste or specific understanding of particular risks. That is, it is unclear whether people differentiate by end-of-life product. Regardless, whether products contribute to landfills or have greater environmental harm, those who are disposing of end-of-life products are imposing costs on others. Therefore, they should pay for imposing those costs. This conforms to the stance taken in Australia over the mandatory e-waste product stewardship scheme.

Another aspect of low response to recycling is convenience. Studies show people value convenient collection and/or drop off points. That is, the cost of disposing of an end-of-life product includes both any charge for disposal plus the costs incurred by the product holder in presenting the product for collection (time).

An associated point is that people appear to place an option value on owning some products. The hoarding of phones and computers and chemicals may not simply reflect high disposal costs and uncertainty about recycling but also a value in having a product or material available, just in case it is needed.

Further understanding by improving the efficiency and durability of waste programmes could assist by:

- clearly setting out how to manage those unwilling to pay. Our view is that if they are imposing costs on others in particular waste streams then they should bear some of the costs through mandatory schemes
- further understanding the main drivers that influence whether more waste is recovered or landfilled i.e. the importance of convenience and time taken to separate various waste streams. This will assist in determining the extent of waste reduction programmes
- demonstrating what the general public's concerns are around the various waste streams. This will ensure that policymakers fully understand what consumers/general public value and what they do not value and how this might change over time
- targeting the waste reduction in the most important products
- developing ways (incentives) aimed at maximising the waste reduction.

10.4. Point of sale is the preferred option

Recommendation 4: Point of sale levies should be preferred to end-of-life levies in New Zealand for these products.

The final area that requires more investigation is the point at which end-of-life charges are paid. We have no primary research that shows us at what point consumers or producers should pay for diversion in the six product category areas. Overseas regulatory regimes do not seem to be based on any particular economic logic and occur over the whole product life cycle spectrum, and they vary from jurisdiction to jurisdiction.

After reviewing the studies and the economics, we suggest that the point of initial sale in New Zealand is the point in the product life cycle to apply a fee, for the following reasons:

- New Zealand has little direct influence over product design and manufacture. Most of the products we investigated are imported, so production happens overseas
- buying products is a critical point in the life cycle, and leads to the eventual end-of-life situation. Putting a cost on diversion or disposal at that point will have flow-on effects for the rest of the product life cycle
- disposal fees – pay as you throw (PAYT) – have the potential to create unwanted or unintended consequences: free-riding, fly-tipping, hoarding and illegal dumping.

We do recognise that there are arguments in favour of PAYT. In particular, by making disposal more expensive, PAYT encourages consumers to get the most out of products. If they have to pay to dispose of a television or computer, consumers are likely to use them for a little longer. However, the decision to dispose of a product is affected by many factors. For example, tyres are likely to be changed during regularly scheduled maintenance rather than at some point of optimal usage. The impact of the disposal fee will be diluted by the other factors affecting disposal decisions.

The design of end-of-life charges is likely to be important to the success of programmes for waste reduction. There is a lack of research on the effectiveness of different designs. Focused primary and secondary research are would be useful to understand these issues further in a New Zealand context.

10.5. Summary

Table 25 summarises the recommendations, policy questions and gaps in information required.

Table 25 Summary of policy questions and current information

	Does the New Zealand public want to recycle specific end-of-life-products enough to pay for it?	If yes, would they prefer to pay for it at the end of life or point of sale?	Is more information required?
E-waste	Yes, given increased awareness of environmental issue and Australian e-waste study (URS, 2009), we would expect between 0.5% and 2% of the purchase price	No information. However, from an equity[1] perspective point of sale is our preferred option. It also provides incentives to producers for better disposal design	Information required on intervention design and the willingness to pay (and its dynamics)[2]
Batteries	Unsure, landfill may be the most efficient disposal method	Will depend on the results of further investigation	Further consumer study required to understand whether a diversion scheme is required
Mercury-containing lamps	Unsure, we have little understanding of consumer preferences	Will depend on the results of further investigation	Use mercury inventory (Chrystall et al, 2009) and mercury life assessment (Garrett & Collins, 2009) to develop a WTP survey
Tyres	Yes, further work required on the design of product stewardship approach and technical issues associated with resource recovery	No information but our view is point of sale (see e-waste)	WTP estimates already available (see Table 24)
Agricultural chemicals	Subject to existing product stewardship.	N/A	Less of an issue for end-of-life product
Refrigerant gases	Better dealt with under international obligations	N/A	N/A
Notes: (1) The NZIER has a strong preference for point of sale charges because it is likely computers will be passed on/sold to other users who have less ability to pay for disposal. (2) We have little information on willingness to pay in New Zealand or how that willingness to pay will move over time.			

Source: NZIER

11. References

Abdullah, S., & Rosenberger, R. S. (2012). *Controlling for Biases in Primary Valuation Studies: A Meta-analysis of International Coral Reef Values* (Working Paper No. 2012.72). Fondazione Eni Enrico Mattei. Retrieved from http://ideas.repec.org/p/fem/femwpa/2012.72.html

Achillas C, Vlachokostas C, Karagiannidis A, Sartzetakis E, and Moussiopoulos N (2012), Willingness to pay for more sustainable e-waste management: a contingent valuation study in Greece. *Economics and Policy of Energy and the Environment Vol 2012/3, Issue 3, pp 19-35*.

Agrecovery - Rural Recycling Programme. (n.d.). Retrieved 12 July 2013, from http://www.agrecovery.co.nz/

Atasu, A., Lifset, R., Linnell, J., Perry, J., Sundberg, V., Mayers, C. K., … Gregory, J. (2010). *Individual Producer Responsibility: A Review of Practical Approaches to Implementing Individual Producer Responsibility for the WEEE Directive* (INSEAD Working Paper no 2010/71). Social Innovation Centre. Retrieved from http://papers.ssrn.com/sol3/Delivery.cfm?abstractid=1698695

Bateman, I. J., Carson, R. T., Day, B., Hanemann, M., Hanley, N., Hett, T., Swanson, J. (2002). *Economic valuation with stated preference techniques: a manual.* Cheltenham, UK: Edward Elgar Publishing Ltd.

Beja, E. L. (2012). Subjective Well-Being Approach to Environmental Valuation: Evidence for Greenhouse Gas Emissions. *Social Indicators Research*, *109*(2), 243–266. doi:10.1007/s11205-011-9899-1

Bergstrom, J. C., & Taylor, L. O. (2006). Using meta-analysis for benefits transfer: Theory and practice. *Ecological Economics*, *60*(2), 351–360. doi:10.1016/j.ecolecon.2006.06.015

Berk, R. A., & Fovell, R. G. (1999). Public Perceptions of Climate Change: A 'Willingness to Pay' Assessment. *Climatic Change*, *41*(3-4), 413–446. doi:10.1023/A:1005489221644

Brander, L. M., Van Beukering, P., & Cesar, H. S. J. (2007). The recreational value of coral reefs: A meta-analysis. *Ecological Economics*, *63*(1), 209–218. doi:10.1016/j.ecolecon.2006.11.002

Brindley, F., Mountjoy, E., Mountjoy, G., Council of Australian Governments, & Standing Council on Environment and Water. (2012). *Study into domestic and international fate of end-of-life tyres*. Melbourne: Hyder Consulting.

Brons, M., Pels, E., Nijkamp, P., & Rietveld, P. (2002). Price elasticities of demand for passenger air travel: a meta-analysis. *Journal of Air Transport Management*, *8*(3), 165–175.

Campbell, A., Crane, P., Schaaf, C., Tracy, S., & Vo, B. D. (2008). *Determining a Waste Tyre Management System for Hong Kong*.

Chrystall, L., Rumsby, A., & Pattle Delamore Partners Ltd. (2009). *Mercury Inventory for New Zealand 2008* (A report for the Ministry for the Environment).

CM Consulting. (2012). *Managing Canada's waste batteries*.

Covec (2012), Economic Factors of Waste Minimisation in New Zealand. *Prepared for the Ministry for the Environment*, November 2012.

CRL Energy Ltd. (2008). *Synthetic Greenhouse Gases and the Emissions Trading Scheme* (A report for the Ministry for the Environment).

Crutchfield, S. R., Feather, P. M., & Hellerstein, D. R. (1995). The benefits of protecting rural water quality. *USDA Agricultural Economic Report*, *701*. Retrieved from http://ddr.nal.usda.gov/bitstream/10113/33022/1/CAT10714002.pdf

Denne, T., Atreya, N., & Robinson, M. (2007). *Recycling : cost benefit analysis. Prepared for Ministry for the Environment* (Final report). covec.

Department of Sustainability, Environment, Water, Population and Communities. (2012). *Destruction of Waste Ozone Depleting Substances and Synthetic Greenhouse Gases Program Consultation Paper*. Retrieved from http://www.environment.gov.au/atmosphere/ozone/destruction-program/pubs/destruction-program-consultation-paper.pdf

eDay NZ Trust (2010), eDay 2010 Report 2010.

European Commission. (n.d.). Waste Electrical and Electronic Equipment. Retrieved 9 July 2013, from http://ec.europa.eu/environment/waste/weee/studies_weee_en.htm

European Commission. (2008). Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on waste and repealing certain Directives.

Foster V and Mourato S (2000), Measuring the Impacts of Pesticides Use in the UK: A Contingent Ranking Approach. *Journal of Agricultural Economics*, 51. 1-21

Freedonia. (2012). *World Commercial Refrigeration Equipment*. Retrieved from http://www.freedoniagroup.com/industry-study/2953/world-commercial-refrigeration-equipment.htm

Garrett, P., & Collins, M. (2009). Life Cycle Assessment of product stewardship options for mercury-containing lamps in New Zealand (Report for the Ministry for the Environment). Environmental Resources Management. Retrieved from http://www.mfe.govt.nz/publications/waste/product-stewardship-options-mercury-containing-lamps/lca-product-steward-options-mercury-containing-lamps.pdf.

Gilbert G (2001), Establishing Permanent Collection Sites for Container Recycling at business locations- How to convince a pesticide dealer to collect containers for you. National Pesticide Stewardship Alliance conference November 27-30 2001 http://www.npsalliance.org/2002 conf.html

GHD Limited. (2012). Canterbury non natural rural waste regional assessment and guidance note development (Report prepared for Environment Canterbury No. R12/74). Christchurch, N.Z.: Environment Canterbury.Hanley, N., Ryan, M., & Wright, R. (2003). Estimating the monetary value of health care: lessons from environmental economics. *Health economics*, *12*(1), 3–16.

Handley N, Ryan M, Wright R (2003) Estimating the Monetary value of Health care: Lessons from environmental economics. *Health Econ, 2003 Jan(1): 3-16*.

Hansmann, R., Bernasconi, P., Smieszek, T., Loukopoulos, P., & Scholz, R. W. (2006). Justifications and self-organization as determinants of recycling behavior: The case of used batteries. *Resources, Conservation and Recycling*, *47*(2), 133–159. doi:10.1016/j.resconrec.2005.10.006

Hasing, T. (2010). *The Effect of Label Information on Farmers' Pesticide Choice*. Department of Agricultural and Resource Economics, North Carolina State University.

Retrieved from http://ageconsearch.umn.edu/bitstream/61131/2/Tomas_paperAAEAmeetings.pdf

Hennessy, W. (2010). *Methods for synthetic greenhouse gas regulations* (A report for the Ministry for the Environment).

Hennessy, W. (2011). *Evidence on Alternative S ynthetic Greenhouse Gas Policies* (A report for the Ministry for the Environment).

Houghton, N., Preski, K., Rockliffe, N., & Tsolakis, D. (2004). *Economics of Tyre Recycling* (A report prepsred for the Department of Environment and Heritage).

Kotchen M, Boyle K, and Leierowitz A (2012) Willingness to Pay and Policy-Instrument Choice for Climate Change Policy in the United States. (Forthcoming in Energy Policy) environment.yale.edu/kitchen/pubs/pinstchc.pdf

Kwak, S.-J., Yoo, S.-H., & Kim, C.-J. (2004). Measuring the economic benefits of recycling: the case of the waste agricultural film in Korea. *Applied Economics*, *36*(13), 1445–1453. doi:10.1080/0003684042000206988

Lee J, Yoo S, and Kwak S (2010) Public's willingness to pay for preventing climate change. EconPapers 2010 vol 17, Issue 6, pp 619-622

Leamer Edward (1996), Questions, Theory and Data. In Foundations of Research in Economics: How do economists do economics eds Medema and Samuels. Edward Elgar.

Levitan, L. C., Cox, D. G., & Clarvoe, M. B. (2005). *Agricultural Plastic Film Recycling: Feasibility and Options in the Central Leatherstocking-Upper Catskill Region of New York State* (Technical Report). Recycling Agricultural Plastics Project (RAPP), Cornell University. Retrieved from http://ecommons.library.cornell.edu/handle/1813/33176

Lifset, R., Atasu, A., & Tojo, N. (2013). Extended Producer Responsibility. *Journal of Industrial Ecology*, *17*(2), 162–166. doi:10.1111/jiec.12022

Lundgren, K., International Labour Office, Programme on Safety and Health at Work and the Environment, International Labour Office, & Lundgren, K. (2012). *The global impact of e-waste: addressing the challenge*. Geneva: ILO.

MacGibbon J (2011) The ewaste problem – size and trends. . In *Ewaste in New Zealand: Five years on.* eDay.

Milovantseva, N., & Saphores, J.-D. (2013). E-waste bans and U.S. households' preferences for disposing of their e-waste. *Journal of Environmental Management*, *124*, 8–16. doi:10.1016/j.jenvman.2013.03.019

Ministry for the Environment. (n.d.). Safe disposal of batteries. Retrieved 12 July 2013, from http://www.mfe.govt.nz/publications/waste/safe-disposal-of-batteries/html/

Montrose, P. (2011). Think Before You Shuffle. *News to Use*. Retrieved from http://www.wellcorps.com/Think-Before-You-Shuffle.html

Morrison, M., Bennett, J., Blamey, R., & Louviere, J. (2002). Choice Modeling and Tests of Benefit Transfer. *American Journal of Agricultural Economics*, *84*(1), 161–170.

Negra C & Rogers G (1998), Agricultural Plastic Recycling: Investigation of Opportunities and Obstacles in Vermont. Prepared for the USDA.

Nicol, S., & Thompson, S. (2007). Policy options to reduce consumer waste to zero: comparing product stewardship and extended producer responsibility for refrigerator waste. *Waste Management & Research*, *25*(3), 227–233. doi:10.1177/0734242X07079152

Nixon C (2011), The economics of ewaste. *In ewaste in New Zealand: Five years on*. eday and Ministry for the Environment.

Nixon C and Yeabsley J (2013), Voyage of discovery: How do we bring analytical techniques to state driven behaviour change? http://www.regulatorytoolkit.ac.nz/improving-regulation/quantitive-tools

Nixon, Hilary, and Jean-Daniel M. Saphores. (2007). Financing Electronic Waste Recycling Californian Households' Willingness to Pay Advanced Recycling Fees. Journal of Environmental Management 84 (4), September.

Nixon Hilary, Saphores Jean-Daniel, Ogunseitan Oladele, and Shapiro Andrew (2008), Understanding Preferences for Recycling Electronic Waste in California: The Influence of Environmental Attitudes and Beliefs on Willingness to Pay. Environment and Behaviour 2009 41 101 http://www.sagepub.com/dicken6/Additional%20Resources%20for%20International%20Business/Online%20journal%20readings/nixon%20et%20al%202009.pdf

Ongondo, F. O., & Williams, I. D. (2011). Greening academia: Use and disposal of mobile phones among university students. *Waste Management*, *31*(7), 1617–1634. doi:10.1016/j.wasman.2011.01.031

Planet Ark (2010) Battery Use, Disposal & Recycling in Australia. http://recyclingweek.planetark.org/documents/doc-513-battery-research-report-final.pdf

Poot, J. (2012). *Learning from the Flood of Findings: Meta - Analysis in Economics*. Presented at the Presentation to the New Zealand Association of Economists conference, Palmerston North.

Rolfe, J., Loch, A. J., & Bennett, J. (2006). Testing benefit transfer with water resources in Central Queensland, Australia. *Choice Modelling and the Transfer of Environmental Values*, 112–135.

Saphores J, Ogunseiten O, Shapiro A (2012), Willingness to Engage in Pro environmental behaviour. An analysis of e-waste recycling based on a national survey of US Households. Resources, Conservation, and Recycling, 60 (March): 49-63

Skevas T, Stefanou S, Lansink A (2010), Do Farmers Internalise External Impacts of Pesticides in Production? Paper prepared for presentation at the 120th EAAE Seminar "External Cost of Farming Activities: Economic Evaluation, Environmental Repercussions and Regulatory Framework. Chania, Crete, Greece, date as in: September 2-4, 2010.

Smith, T., Sonnenfeld, D. A., & Pellow, N. (Eds.). (2006). *Challenging the Chip: Labor rights and Environmental Justice in the Global Electronics Industry*. Philadelphia, PA: Temple University Press. Retrieved from http://www.temple.edu/tempress/titles/1788_reg_print.html

Song Q, Wang Z, and Li J (2012), Residents' Behaviours, Attitudes, and Willingness to Pay for recycling e-waste in Macau. *Journal Environmental Management* 2012, Sept 15th 106: pp 8-16

Tanskanen, P., & Butler, E. (2007). Mobile phone take back learning's from various initiatives. In *Proceedings of the 2007 IEEE International Symposium on Electronics the Environment* (pp. 206–209). Presented at the Proceedings of the 2007 IEEE

International Symposium on Electronics the Environment. doi:10.1109/ISEE.2007.369395

Travisi, C. M., & Nijkamp, P. (2004). *Willingness to pay for agricultural environmental safety*. Tinbergen Institute. Retrieved from http://papers.tinbergen.nl/04070.pdf

Tripathi, N. (2011). International approaches to ewaste recycling. In *Ewaste in New Zealand: Five years on.* eDay.

Tsydenova, O., & Bengtsson, M. (2011). Chemical hazards associated with treatment of waste electrical and electronic equipment. *Waste management (New York, N.Y.)*, *31*(1), 45–58. doi:10.1016/j.wasman.2010.08.014

Tunsu C, Retegan T, and Ekberg C (2011), Sustainable processes development for recycling of fluorescent phosphorous powders – rare earths and mercury separation: A literature report. *University of Technology Gothenburg* Sweden, 2011.

Tyrewise Working Group. (2012, June 7). Scoping Report 1: Investigation into the collection and disposal of used tyres in New Zealand and internationally. Retrieved from http://www.tyrewise.co.nz/wp-content/uploads/2012/06/Tyrewise-Scoping-Report-1-Current-ELT-Situation-V3.pdf

UNEP. (2013a). *Costs of Inaction on the Sound Management of Chemicals*. UNEP.

UNEP. (2013b). *Global Chemicals Outlook*.

United Nations Environmental Programme Division of Technology, Industry and Economics. (2007). *E-waste: Volume 1: Inventory assessment manual*. Osaka/ Shiga: International Environmental Technology Centre. Retrieved from http://www.unep.or.jp/ietc/Publications/spc/EWasteManual_Vol1.pdf

URS. (2009). Willingness to Pay for E-waste Recycling. Report prepared for Environment Protection and Heritage Council.

URS. (2006a). *Product Stewardship Case Study - Cell Phones* (A report for the Ministry for the Environment). Retrieved from http://www.mfe.govt.nz/publications/waste/product-stewardship-cell-phones-may06/index.html

URS. (2006b). *Product stewardship case study for end-of-life tyres* (A report for the Ministry for the Environment). Retrieved from http://www.mfe.govt.nz/publications/waste/product-stewardship-endoflife-tyres-may06/index.html

Ward, D. O., Clark, C. D., Jensen, K. L., Yen, S. T., & Russell, C. S. (2011). Factors influencing willingness-to-pay for the ENERGY STAR label. *Energy Policy*, *39*(3), 1450–1458.

Warnken Industrial and Social Ecology Pty Ltd. (2010). *Analysis of battery consumption, recycling and disposal in Australia* (Report for the Australian Battery Recycling Initiative).

Watkins, T. (2011, August 8). Exide suing Government over used battery imports. *Business Day*. Retrieved from http://www.stuff.co.nz/business/5404900/Exide-suing-Government-over-used-battery-imports

Wilhelm, W., Yankov, A., & Magee, P. (2011). Mobile Phone Consumption Behavior and the Need for Sustainability Innovations. *Journal of Strategic Innovation and Sustainability*, *7*(2), 20–40.

Appendix A Government policy and legislation

A.1 Domestic

Government policy is presented in the New Zealand Waste Strategy. Key legislation comprises the Waste Minimisation Act 2008, the Resource Management Act 1991, Hazardous Substances and New Organisms Act 1996, Imports and Exports (Restrictions) Prohibition Order (No 2) 2004, Ozone Layer Protection Act 1996, and the Climate Change Response Act 2002.

A.1.1 New Zealand Waste Strategy

The revised New Zealand Waste Strategy, which sets out the Government's long-term priorities for waste management and minimisation, was released in 2010. The strategy provides high-level direction. Its two goals are:

- Goal 1: Reducing the harmful effects of waste
- Goal 2: Improving the efficiency of resource use.

With respect to Goal 1, the strategy states that: there is a risk waste will cause harm to the environment or human health; the level of risk can be established by using appropriate criteria to assess both the likelihood of the harm occurring and its potential consequences (ISO 31000:2009 *Risk Management* and HB 436:2004 *Australia New Zealand Handbook on Risk Management Guidelines*); and reducing the risk of harm means taking steps to reduce the likelihood of it occurring or, if unavoidable, managing its consequences.

With respect to Goal 2, the strategy provides examples as follows.

- Resource efficiency in production could mean: improving the ratio of outputs to inputs; reducing and reusing end-of-life products, and minimising what needs to be disposed of at the end of a product's life.
- Resource efficiency in consumption could mean: choosing products that are reusable, durable and able to be repaired rather than discarded; choosing products with less packaging, choosing recyclable products and packaging.

The strategy states that, to support New Zealand's moves towards improving efficiency, a change in the way we all buy and dispose of goods and services is needed and pricing policies that reflect the full costs associated with waste are one way to instigate this behaviour change.

The strategy documents notes that the lack of data about waste hampers our ability to plan appropriate activities to improve waste management and minimisation. This situation continues.

A.1.2 Waste Minimisation Act

The purpose of the Act is to encourage waste minimisation and a decrease in waste disposal in order to: protect the environment from harm; and provide environmental, social, economic, and cultural benefits.

Significantly the Act differentiates between *waste* (effectively that material destined for a landfill) and *diverted material* (anything no longer required for its original purpose and, but for commercial or other waste minimisation activities, would be destined for a landfill).

Also, significantly, a *disposal facility* is a specific type of facility – one which receives waste, including household waste and is run, at least in part, as a waste disposal business. Plus a disposal facility means any other facility at which waste is disposed of that is prescribed as a disposal facility.

The Act provides for product stewardship through the accreditation of voluntary product stewardship schemes and the declaration of priority products, with a purpose to encourage (and, in certain circumstances, require) the people and organisations involved in the life of a product to share responsibility for waste minimisation and managing any environmental harm arising from the product when it becomes waste. A priority product may be declared if a product will or may cause significant harm when it becomes waste or there are significant benefits from minimising the waste product.

Currently there are nine accredited product stewardship schemes and, significantly, there are no priority products or regulations for end-of-life products.

The Act provides for the imposition of a levy on all waste to a disposal facility to raise revenue for promoting and achieving waste minimisation; and to increase the cost of waste disposal to recognise that disposal imposes costs on the environment, society, and the economy. Currently the levy is $10 per tonne of waste. Half the income is distributed to territorial authorities for waste minimisation activities and the rest, less administration costs is used to fund approved projects through the Waste Minimisation Fund.

Under the Act, territorial authorities must promote effective and efficient waste management and minimisation within their districts and adopt a plan for this purpose that address matters stipulated in the Act.

A.1.3 Resource Management Act

The purpose of this Act is to promote the sustainable management of natural and physical resources. Under the Act, the environmental effects of facilities at which waste is or has been disposed, waste transfer stations and resource recovery facilities are controlled.

A.1.4 Hazardous Substances and New Organisms Act 1996

Under this Act and its regulations the import, manufacture, use and disposal of hazardous substances and the introduction of new organisms into New Zealand are controlled.

A.1.5 Climate Change Response Act 2002

Under this Act, regulations apply to the disposal of waste at a disposal facility, as defined in the Waste Minimisation Act, with a purpose of controlled greenhouse gas emissions (landfill gas). Operators are required to surrender emission units in proportion to one of two prescribed methods of estimating landfill gas emissions. The cost of emission units currently are much lower than projected at the time of the last levy review (NZU approximately $2 and ERU $0.2 to $0.3). The low unit price has had little effect on landfill gates fees and, consequently, has not contributed as forecast as a disincentive for the disposal of waste to landfill.

A.1.6 Other relevant legislation

- Imports and Exports (Restrictions) Act 1988
- Imports and Exports (Restrictions) Prohibition Order (No 2) 2004
- Agricultural Compounds and Veterinary Medicines (ACVM) Act 1997
- Ozone Layer Protection Act 1996 and Regulations

A.2 International Obligations

The **Stockholm Convention** on Persistent Organic Pollutants commits signatories to protect human health and the environment by reducing and, where feasible, eliminating the production and environmental releases of listed persistent organic pollutants (POPs), such as polychlorinated biphenyls (PCBs). The Convention requires that New Zealand collects and arranges for the destruction of specific waste chemicals, and that we properly manage material and sites that are contaminated with POPs.

Basel Convention on Control of Transboundary Movements of Hazardous Wastes and their Disposal aims to achieve the environmentally sound management of hazardous wastes through the appropriate regulation of hazardous waste movement across national boundaries. The Convention applies a prior informed consent process to transboundary movements of hazardous wastes. To implement the Convention, New Zealand requires those exporting hazardous waste to get a permit to ensure the waste is only shipped to countries or facilities where environmentally sound management of the waste can be ensured.

Montreal Protocol on Substances that Deplete the Ozone Layer was designed to reduce the production and consumption of ozone depleting substances in order to reduce their abundance in the atmosphere, and protect the earth's fragile ozone Layer. The original Montreal Protocol was agreed on 16 September 1987 and entered into force on 1 January 1989.

Waigani Convention is a Convention to Ban the Importation into Forum Island Countries of Hazardous and Radioactive Wastes and to Control the Transboundary Movement and Management of Hazardous Wastes within the South Pacific Region, provides additional provisions concerning problems with hazardous wastes in Pacific Island states.

Minamata Convention on Mercury. The Minamata Convention on the control of mercury is open for signature later this year, and will come into force when 50 countries have ratified it.

OECD Decision C/2001 107/Final is a decision of the OECD governing the transboundary movement of hazardous waste destined for recovery operations, between OECD member countries.

Appendix B Framework for analysis

B.1 Supply and demand

A market for anything – a product, resources or services – can be divided into two parts: supply and demand. The waste area is no different. Waste minimisation or end-of-life products can be analysed as a market for diversion – a market for all the things that can happen to waste products other than going to landfill. The perception that there is a waste ‘problem’ is the same as saying that the market outcome – the current intersection of supply and demand – is not producing the desired result with respect to waste diversion. We set up a theoretical analysis of the waste diversion market, describe the factors affecting supply and demand, and then discuss what ‘problems’ could be occurring.

We start with a standard supply-demand diagram, similar to the approach in other waste and recover literature (Denne et al., 2007; Houghton, Preski, Rockliffe, & Tsolakis, 2004). The quantity of diversion is given on the x-axis, and can be measured as percentage diversion on a scale from 0% to 100%. The dollar measure of diversion, which indicates the cost, price or willingness to pay, is given on the y-axis. Demand for diversion is driven by a number of factors:

- personal characteristics, such as income and gender as well as attitudes and beliefs
- the product and its characteristics
- the type of diversion (and its reliability).

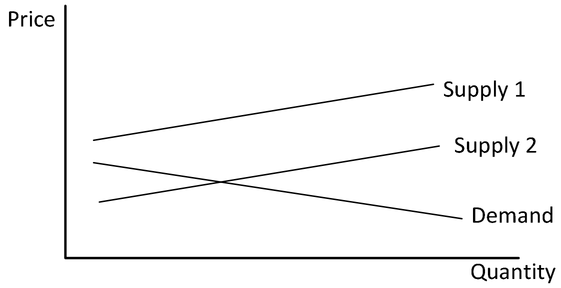
Supply of diversion is also driven by several factors:

- technology of diversion, including impacts of economies of scale
- value of feedstock and diverted material
- cost of alternatives, such as landfills.

Taken together, the supply and demand describe the market for diversion, as shown in Figure 4. The figure shows one demand line and two possible supply lines. Supply 1 indicates that the cost of supplying an end-of-life product is always greater than people are willing to pay; in this situation, no diversion occurs. Supply 2 has a lower cost of diversion, so it starts below the demand line and then rises above it. The point where the two lines cross indicates the amount of diversion that would occur. It is some positive amount, but much of the quantity is not diverted and goes to landfill.

This diagram can be modified to deal with different waste situations.

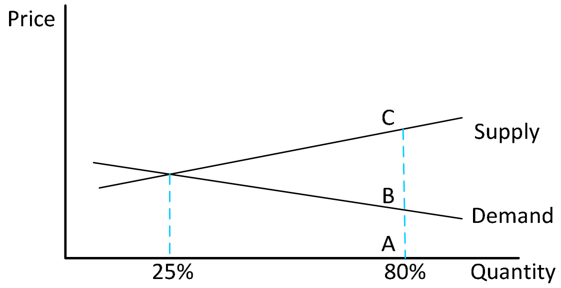
Figure 4 Market for end-of-life products



Source: NZIER

One possible situation is shown in Figure 5. In the figure, demand and supply of an end-of-life product is such that 25% of possible diversion actually occurs without specific intervention. An 80% diversion target is also shown in the figure. At 80%, the marginal demand for diversion is lower than the cost of supplying the diversion. The demand 'price' is shown by B, while the higher supply 'price' is shown by C. The distance AB is willingness to pay for that level of diversion. There are two points to make. First, the WTP at that level is unobserved. We can observe what actually happens – the 25% level – but not the target level. This WTP must be estimated, which is the goal of the first and second stages of the current research programme. Secondly, the distance BC represents the amount of support or subsidy that must be provided to produce the 80% diversion target. Some WTP exists at that level (AB); the difference between the demand and supply 'prices' (BC) indicates the required level of support.

Figure 5 Raising the level of diversion



Source: NZIER

B.2 Total economic value

Non-market valuation methods measure the costs and benefits of changes in effects that are not reflected in market transactions, to estimate potential value to a community or nation. For this reason they can be useful in valuing environmental "goods" such as waste diversion. The approaches attempt to gauge value by measuring society's Willingness to Pay (WTP) to obtain a beneficial change.

If a market for a specific good, such as waste diversion does not exist, then it is not possible to assess WTP by examining market price and quantity traded. Instead, WTP must be assessed by other methods. These methods are broadly divided between:

- Market value approaches that infer value using market-based prices, such as the value of the next best way of achieving a given outcome (e.g. the costs of running a waste disposal service and landfill operation)
- Revealed preference approaches that infer value of a non-market resource from observed behaviour with regard to related activities, for instance inferring the value of a landfill operation by the number of users
- Stated preference approaches that infer value of a non-market resource from surveys of the affected population asking them how much they would pay to obtain changes in particular environmental effects.

The non-use component of WTP can only be captured through stated preference techniques i.e. techniques that are based on questionnaires given to respondents and which elicit the respondents WTP directly or indirectly from the respondents answer.

By way of example Table 26 sets out the costs and benefits of provisions to introduce waste diversion schemes. The pattern of costs and differs with and without the waste diversion schemes. The approach aims to weigh up the incremental gains and losses to identify which has the greatest benefit.

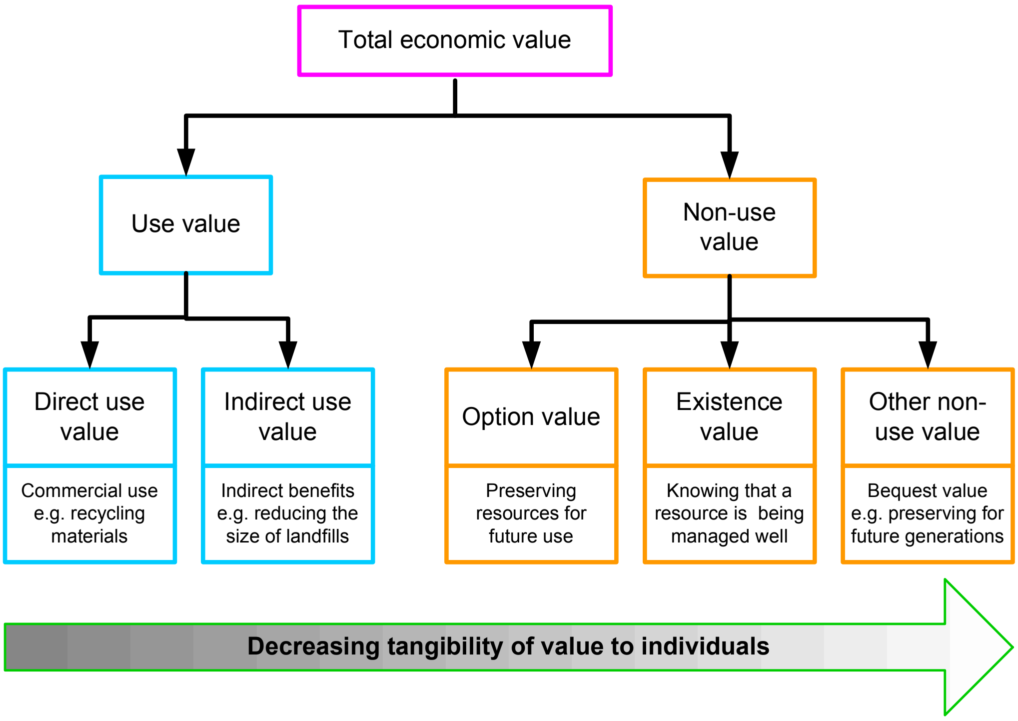
Table 26 Costs and benefits for provision of waste diversion services

	No waste diversion	Waste diversion in place
Current use value	Benefit from the use of landfill. Costs of landfill set up, operation, and products disposed of (loss of rare metals and increased toxic substances).	Costs of running diversion schemes. Costs associated with those unwilling to pay if a scheme is mandatory. Benefits for those using waste diversion schemes including image and branding of products. Benefits of better disposal design of products.
Future use value (or option)	Loss of opportunities to recycle potentially valuable materials. Costs associated with leaching from private landfills. The benefits of free riding on waste disposal.	Opportunity cost of using money from stewardship schemes on some other activity. Benefit from recycling opportunities as technology advances.
Non-use value	Low public benefit with no waste diversion	Higher public benefit with waste diversion schemes in place

Source: NZIER

Figure 6 sets out another way of examining the different costs and benefits associated with waste management. Use values are relatively easily defined, into commercial uses and indirect use values. Use values involve the direct commercial activities associated with recycling materials. An indirect use value might be the value minimisation of waste being put into landfills so that the potential waste streams are more efficiently used.

Figure 6 Components of economic value



Source: NZIER

A non-use value can be an:

- option value (i.e. to preserve it for future use, say when populations grow);
- existence value (to preserve what we have and improve on what we have). In the case of waste it may be the knowledge that the issue is being addressed in a socially acceptable way; and
- other non-use values such as preserving something for future generations i.e. ensuring that a solution is durable.

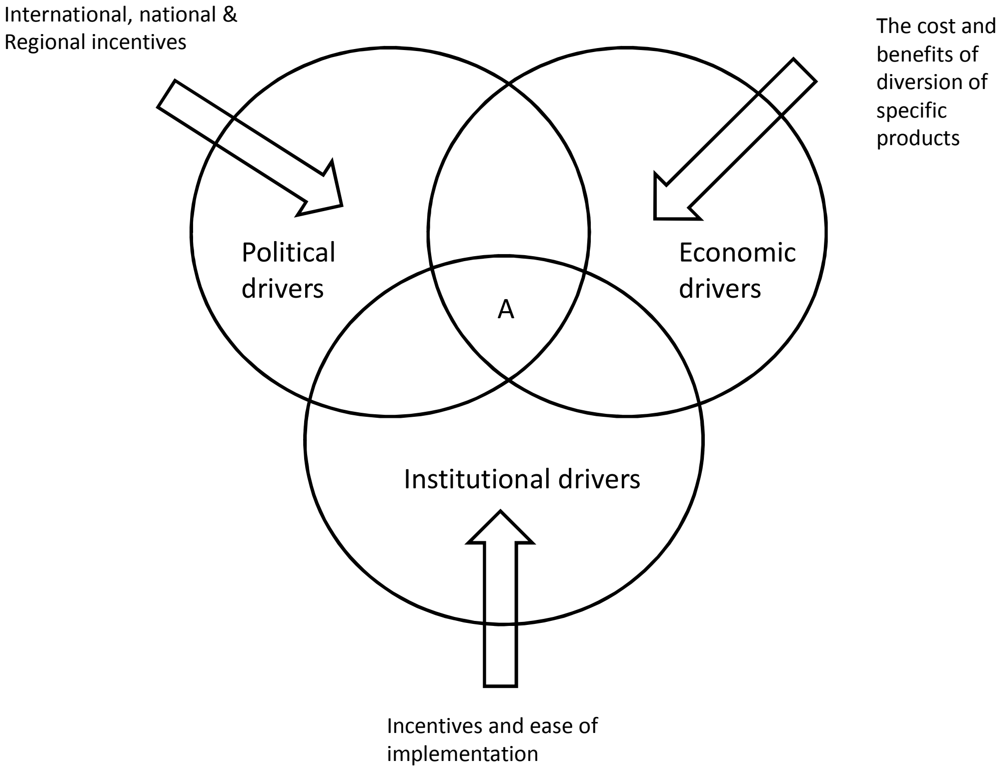
The important issue that total economic value describes is that value is more than what can be observed in markets for the buying and selling of waste products. Other issues need to be considered whose value cannot be observed in market transactions. Therefore, we are interested in whether or not the social benefit of an intervention outweighs the social cost.

B.3 Multiple drivers shape waste and recovery

To understand the current shape of waste issues in New Zealand requires us to examine the interplay of the political, economic and institutional drivers. Furthermore, it is how these three sets of drivers reinforce each other that determines and informs the public policy and regulatory approaches to management of waste and materials recovery. They also determine the extent of the efficiency, effectiveness, innovation and durability of the regime.

Figure 7 sets out one framework for understanding these drivers. It shows a Venn diagram of the political, economic, and institutional drivers that need to be taken into account in a regulatory regime.

Figure 7 Political, economic, and institutional drivers determine the shape of the regulatory approach



Source: NZIER

At point A, all three drivers overlap and so there are no barriers to a workable waste diversion regime in the short run. The three set of drivers can be explained as follows:

- political drivers
  - a country's/region's overriding concerns that motivate politicians to take action or continue the status quo, e.g. this could be driven by the effectiveness of lobby groups, general disquiet by voters about the state of the environment (as voters get richer), or one-off events that trigger a political response
  - how waste regimes affect other trading/political relationships within and between countries over time
  - the degree of local understanding of waste minimisation issues
- economic drivers
  - the economic impact of any particular waste minimisation regime – positive or negative (including use and non-use values)

  - any spillover effects (e.g. unforeseen impact on the market or other industries) from the waste regime approach
- institutional drivers
  - how the mechanisms used work over time and their effectiveness
  - the structure of rules and relationships, including the treatment of free riders
  - the impact of incentives that face institutions, including the ability to influence the rules of the game and political drivers.

A waste minimisation strategy – like any regulatory solution – works best when all three sets of drivers are aligned. This is represented by area A in the figure above: the point where the drivers overlap. In over areas of the Venn diagram, two sets of drivers may overlap but another may be working in a different direction. For example, institutional and economic drivers may be pointing towards one waste solution while political driver points elsewhere. This dynamic underlies the tension between landfilling low-value waste and finding alternatives that involve recycling and recovery.

B.4 The waste 'problem'

The supply-demand diagram provides a way to describe the waste 'problem' from and economic point of view. The problem is that the level of diversion achieved without direct intervention is too low. As in Figure 5, the *laissez-faire* level, at which the demand for end-of-life products is enough to pay for supply in a market transaction, is lower than desired.

There are several possible reasons that the market equilibrium is undesirable:

- heterogeneous preferences – some individuals may have very strong preferences for waste diversion but no way to encourage others to divert their waste. They may be willing to pay enough to reduce their own and others' waste and improve social welfare
- non-market values – part of TEV is non-market values, which are not properly priced in markets for goods and services. Including these values can change the supply and demand lines and lead to a different ideal equilibrium
- externalities – some materials, such as hazardous waste, create impacts on other people, but those impacts are not included in the prices or costs in the diagram. Examples are greenhouse gases from landfills and leachate
- Prisoner's Dilemma – this is a classic game-theory problem. Everyone might be better off if no one disposed of a particular product in landfills (tyres, e-waste, batteries). However, each individual person can improve their situation by 'defecting': by putting their waste in a landfill while everyone else pays for diversion. As a result, all the individuals end up putting their waste in landfills, even though everyone is worse off by doing do
- economies of scale – the supply line shown above assumes that higher level of diversion cost more, but economies of scale can mean that marginal costs fall as diversion increases. The problem is getting enough scale, given that the initial expense is very high

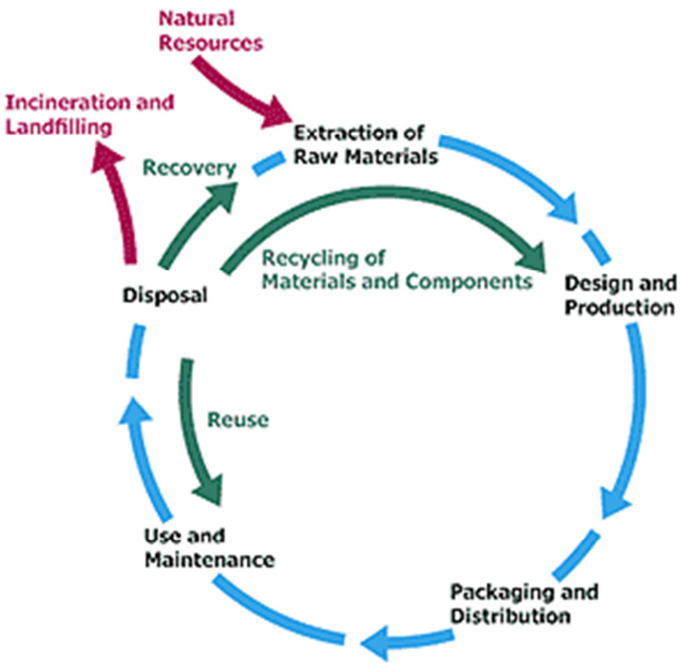
- information – information about various methods of handling waste may not be widely available. Individuals may not be aware of the impacts of certain materials in landfills, or may not know about alternatives to disposal. In addition, information asymmetry may mean that full information is being kept from people making the disposal/recycling decisions
- transactions costs – it is widely understood in economics that the costs of getting things done – of making agreements or organising processes – can have a large impact on economic decisions. For waste, the diffuse nature of the materials and the difficulty of getting the right, clean material to the right place can make it uneconomic to have resource recovery. Landfilling generally has a lower transaction cost than any other method of handling waste.

These reasons for lower rates of diversion than are socially beneficial will be part of the discussion of the individual products.

B.5 Waste within a resource cycle

Another perspective on the waste problem is to describe the life cycle of the resources that combine to make the products that we use, an approach used within the UNEP's Life Cycle Initiative.

Figure 8 Resource life cycle



Source: http://www.lifecycleinitiative.org/starting-life-cycle-thinking/what-is-life-cycle-thinking/

The focus of this paper is the benefit that can be realised by reusing, recycling or recovering products and components and/or the safe disposal of waste products, and the realisation of these benefits with end-of-life management. In many cases, these benefits could also be realised through early stage management – e.g. design – and encouragement of responsible consumer practice.

It is particularly important to take a life cycle perspective when considering policy changes. For example, there has been general disappointment with Extended Producer Responsibility (EPR) systems have not incentivised environmental improvements at the design stage (Dempsey et al., 2010); likewise there have been concerns expressed that EPR may diminish reuse (Lifset et al., 2013).

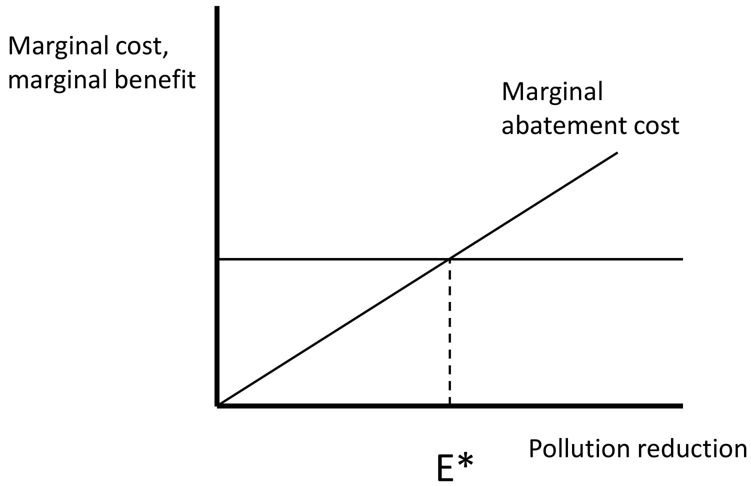
B.6 Charging for externalities

The existence and treatment of externalities is one area where an interaction exists between the life cycle and market failure.

According to the traditional Pigouvian framework, environmental taxes should equal marginal damages and be levied directly on the source of emissions.

A tax equal to marginal damages would induce the efficient level of emissions reduction, indicated by E* in Figure 9. The marginal benefit is the avoided incremental environmental damage, and this equals the marginal abatement costs, namely the tax. In the case of a GHG, the marginal damage would be the present value of future damages from the extra unit of gas emitted (IMF, 2012).

Figure 9 Welfare effects of environmental taxes in the Pigouvian Framework



Source: IMF, 2012

A principle underlying much environmental regulation is that the polluter should pay for damage caused. Applying this principle, if pollution is during the use of the good, such as spraying a pesticide, then the sprayer, and it could be argued, the original manufacturer should incur the cost. Alternatively, if pollution is during the end-of-life phase then the end-holder, and again potentially the original manufacturer, should pay. In practice, administration of charges is costly beyond the first stage so it is common to charge on production or first sale, with the expectation that this cost will ultimately pass to the polluter.

An alternative, where the pollution occurs at the end-of-life, is to impose (a) Pay-As-You-Throw charges and (b) strict environmental standards. Note, if a first-stage charge has already been applied (e.g. Agrecovery fee) and a PAYT charge is also charged then the end-of-life holder has paid twice.

In practice, there are many free-riders, for various reasons, so ‘society’ picks up the bill. In the case of the agricultural chemicals used as examples above, the cost would be incurred via local council rates for landfills or to society in general via continued pollution.