End-of-Life Cathode Ray Tubes:
A Life Cycle Assessment for Australia

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Introduction

Consumer electronic equipment produces large volumes of e-waste annually in Australia, most of which is currently land-filled. Figure 1 shows the dollar value of electronic equipment imported into Australia in 2004 as some indication of the likely volume of equipment which will be discarded over coming years.

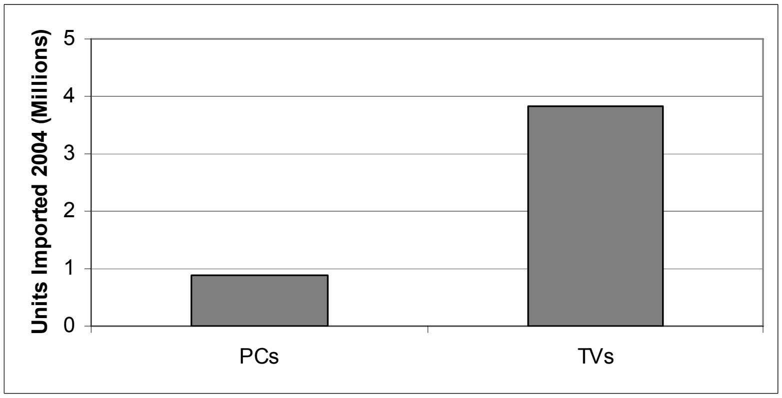


Figure 1. Imports of products likely to have contained CRTs to Australia in 2004 (Australian Bureau of Statistics 2004)

However concerns are growing that landfill may not be the preferred way of dealing with this waste and some industry bodies have commenced action to address this issue (Consumer Electronics Suppliers Association 2003, Consumer Electronics Suppliers Association 2004). One of the specific components of concern is the glass tube commonly used for computer and television displays, known as cathode ray tubes (CRTs). These tubes are of concern primarily because they contain large amounts of lead in the glass, which has the potential to escape into ground water and become a threat to human health and other life. Figure 2 gives an indication of the amount of lead contained in a range of television CRTs.

CRTs contain two different types of glass. The front or screen part is called the panel glass and is normal glass which can be recycled readily. The sides and neck portions are called funnel glass and contain lead (to reduce radiation from the sides of the tube).

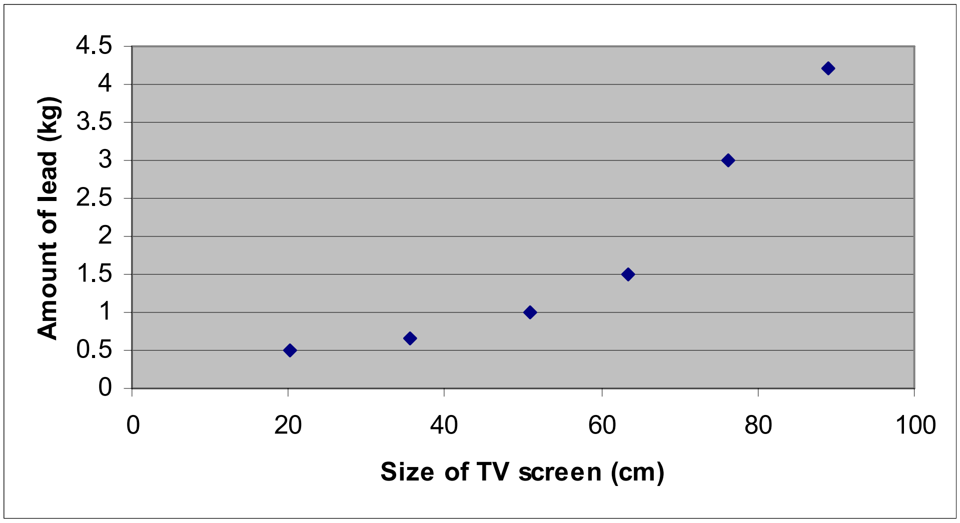


Figure 2. Typical lead content of CRTs

There is increasing international attention being paid to disposing of electronic waste with European nations and parts of the USA for example banning the disposal of CRTs in landfill (see for example Kang & Schoenung 2006 and State of California 2006). In Australia similarly, there is on-going consideration of the issue by government (Department of the Environment and Heritage 2001, Department of the Environment and Heritage 2004).

The aim of this study is to document the various processes used (or potentially used) by industry to dispose of CRTs, both in-country and overseas and thus to attempt evaluation of the relative environmental impacts of each possibility using a life cycle analysis approach.

The Potential Environmental Impacts

Lead is known to be a source of health problems for both humans and other life. The Australian government has commissioned several studies which consider this issue, for example the Computer and Peripherals Material Project (2001) and Electrical and Electronic Products Infrastructure Facilitation (2004). Imports and Exports of relevant products are controlled under the Hazardous Waste (Regulation of Exports and Imports) Act 1989. Discarded CRTs are considered hazardous waste in the USA under their Resource Conservation & Recovery Act because of the lead and other hazardous materials they contain (Jang & Townsend 2003).

However Jang & Townsend (2003) present results showing that leaching from CRTs may be less than predicted by standard US Toxicity Characteristic Leaching Procedure (TCLP) test by a factor of up to 100, so these results must be taken with caution because of local variables such as rainfall, soil types and landfill management. Further, Macauley et al. (2003) note that estimates of the leaching of lead from land-filled CRTs vary from

1.8 e-10 kg per CRT to 1.6 e-3 kg per CRT with the latter larger figure arising if the funnel glass is crushed into fine pieces and that in any case, the emissions are likely to be small and their effects on drinking water also small. They contrast this situation with that when CRTs are incinerated when about 1.2e-4 of lead is released into the air per CRT. The author is not aware of any incineration of CRTs in Australia and so this possibility has not been considered in this paper.

Recycling Possibilities

Several companies which recycle electronic equipment or are part of the processing chain were consulted in order to establish recycling or disposal options which are currently practiced. The information supplied by these companies was general in nature but was valuable in setting up realistic models for recycling of CRTs. These industries utilize the following options for recycling the glass content of CRTs:

Panel glass can be recycled as:

- normal glass;
- glass fibre batts for insulation; or
- flux in a lead smelter.

Funnel glass can be recycled as:

- glass fibre batts for insulation;
- flux in a lead smelter; or
- leaded glass such as new CRTs (which involves it being sent overseas).

Other components of the CRT assemblies such as copper and steel (as detailed below) can be recycled via normal, commonly available, recycling processes.

It is worth noting that the quantity of glass able to be processed as flux in lead smelters in Australia is much greater than that now being disposed of this way, probably partly because there is a dollar cost involved (Hainault et al. 2001, Consumer Electronics Suppliers Association 2003). It is also worth noting that Kang and Schoenung (2006) have done an economic analysis of computer recycling in California, USA and found that the cost of CRT recycling was the largest cost of all the computer parts. In fact, Macauley et al. (2003) argue that (in the USA) the cost of all methods of disposal of CRTs exceeds the value of the avoided health effects such as might occur when lead leaches from a landfill.

A significant part of the cost of disposal in every option is for transportation & labour even when CRTs are collected from a central local pick up point. There is also a lack of incentive for retailers to provide drop off facilities for old equipment (Consumer Electronics Suppliers Association 2003). Collection of used TVs and computer monitors from widely distributed domestic situations would obviously make the cost even more unattractive.

Methodology

In order to conduct the study, a detailed model of the processes involved in each of the disposal scenarios studied was conducted.

These models were implemented using SimaPro software and associated Australian and other databases. The assessment method used was Eco Indicator 99 (Pre 2006) modified for Australian substances (Centre for Design 2006), chosen because:

1. It is commonly used and gives comparisons of life cycle costs which can be related to other published life cycle assessment results if required; and
2. It gives results similar to other common methods of environmental impact assessment as evaluated by Luo et al (2001).

This method allows assessment according to the following categories of damage assessment:

- Human Health. Unit: DALY= Disability Adjusted Life Years; This means different disability levels caused by such things as pollution;
- Ecosystem Quality. Unit: PDF*m2yr; PDF= Potentially Disappeared Fraction of plant species caused for example by toxic releases; and
- Resources. Unit: MJ surplus energy. Additional energy requirement to compensate for lower future ore grade,

and according to the following more specific categories of characterization:

- Respiratory organics
- Respiratory inorganics
- Climate change
- Ecotoxicity
- Acidification/Eutrophication
- Minerals
- Fossil fuels

Models of the CRTs themselves and of their post end-of-life disposal were constructed as detailed below. For this study, the unit was taken to be a single complete CRT, disposed of in a variety of ways. (It was assumed that the material content and processing inputs and outputs of different sized CRTs were simply proportional to the mass of the CRTs and so results based on any given size of CRTs would apply to any other size).

Models of CRTs

It was assumed that recyclers would separate the CRTs from other parts of the TVs or computer monitors and that the CRTs would then become a special stream of waste (Boyce et al. 2002). This implied that all components inherently part of the CRT and difficult to separate without glass breakage should be considered as part of the CRT, namely the:

Copper yoke and associated components;

Electron gun and some associated glass;
Panel glass;
Shadow mask;
Funnel glass; and
Steel external supporting parts.

The printed circuit board and associated components was not included in the inventory.

Models of material composition were constructed for each of these parts, based largely on the data provided by Lee & Hsi (2002) for a 36 cm computer monitor, with glass composition details from Kang & Schoenung (2004) and from Atlantic Consulting and IPU (1998). The figures for lead were taken mainly from Monchamp et al. (2001) but corroborated by Peters Michaud et al. (2003) and many others. Socolof et al. (2001) also provided some guidance.

Materials in the electron gun and yoke were estimated from personal knowledge, estimations from measurements on a typical 36 cm monitor and from figures for the neck of the CRT in Monchamp et al. (2001). The resulting inventory data is given in tables 1 to 3. An attempt was also made to include the phosphor in the assessment by taking some representative data from Ekambaran (2005) and Ozawa and Itoh (2003). The inventory used is shown in table 4.

Table 1. Material content of the deflection yoke and electron gun and associated components

Component	Mass in yoke (kg)	Mass in electron gun (kg)
Copper	0.39	0.014
Plastic	0.21	0.0014
Steel	0.22	0.014
Glass	0	0.024

Table 2. Material content of the shadow mask and CRT supporting parts

Component	Mass in shadow mask (kg)	Mass in supporting parts (kg)
Steel	0.46	0.3

Table 3. Material content of the funnel and panel glass

Component	Mass in funnel glass (kg)	Mass in panel glass (kg)
SiO_2	1.25	2.89

SrO	0	0.46
Na_2O	0.15	0.36
K_2O	0.21	0.36
BaO	0.0073	0.41
PbO	0.58	0
Al_2O_3	0.090	0.094
CaO	0.097	0
ZrO_2	0	0.056
MgO	0.048	0
TiO_2/CeO_2	0	0.023
Sb_2O_3	0.0048	0.016
Fe_2O_3	0	0.0023
ZnO	0	0.023
Total	2.42	4.70
Funnel Glass Coatings	-	0
Inner iron oxide	0.064	0
Outer carbon black	0.016	0

Table 4. Material content assumed for phosphors used in CRTs.

Component	Mass (kg)
F	0.000069
Na_2O	0.00011
Al_2O_3	0.00049
SiO_2	0.00012
P_2O_5	3.56E-06
SO_3	0.0016
Cl	8.25E-07
K_2O	0.000020
CaO	0.0000074
V_2O_5	0.0000017
Cr_2O_3	1.48E-06
MnO	1.17E-06
Fe_2O_3	2.99E-06
NiO_2	0.0000028
CuO	0.0000043

ZnO	0.0012
Ga_2O_3	6.94E-07
Y_2O_3	0.00041
CdO	0.00017
I	8.25E-07
BaO	0.000022
La_2O_3	5.64E-07
Ta_2O_5	0.000015
WO_3	0.000026
PbO	0.0000056

Models for the major components such as the glass, lead, copper, steel and plastic parts of the CRTs were developed which included all material content but excluded inputs and outputs during production in order that the study include only end-of-life environmental impacts.

However, in order to identify the benefits of, for example, avoiding the production of raw lead by recovering it in a lead smelter, normal material production data which included inputs such as energy and outputs such as emissions to air were used in the waste treatment scenarios modelled.

Landfill

Details of landfill were based on typical Australian landfill figures for Australia but with specific data related to the copper and lead content of CRTs taken from Huisman (2003) as shown in table 5 and Jang & Townsend (2003). The figures used in the study are given in table 6. Note specifically that 120 g of lead released into water per ton of funnel glass was assumed for the study.

Table 5. Estimated leaching of materials in a controlled landfill. Source: Huisman (2003) table 3.3

Element	Leaching to soil	Leaching to air	Leaching to water
Cu	0.0002%	0.0000%	0.0045%
Pb	0.0001%	0.0000%	0.0016%

Table 6. Assumed releases to water from landfill of glass in Australia.
Source: Australian data included in Sima Pro database with modifications for lead according to Jang and Townsend (2003).

Element	Release from panel glass (kg/kg glass)	Release from funnel glass (kg/kg glass)

Lead	1.9e-13	0.12e-3
Cadmium	2.2e-14	2.2e-14
Zinc	3.7e-14	3.7e-14
Chromium	3.7e-12	3.7e-12
Nickel	5.1e-12	5.1e-12
Manganese	2.8e-10	2.8e-10
Asenic	0.8e-12	0.8e-12
Barium	1.4e-10	1.4e-10
Selenuim	0.8e-13	0.8e-13
Vanadium	1.5e-12	1.5e-12

Collection Scenarios Assumed

It was assumed that all product to be disposed of was collected from a collection site at a users premises, council depot, or similar. Thus any environmental costs associated with collection from small volume users have been avoided. This was done deliberately because it was assumed on the basis of other studies such as Hainault et al. (2001) that such costs would be relatively high and potentially mask other results.

Processing Data

Some guidance about details of the processes involved in handling end-of-life CRTs such as washing or crushing glass was taken from Huisman (2003) and the major ones were modelled as closely as possible, largely on the basis of energy usage. These models were constructed from data available in SimaPro using only broad knowledge of details involved. Consequently there are some uncertainties in this data, but final results suggest that these elements are not significant in the overall situation. Some consideration is given to a sensitivity analysis based on some of this data below.

Processing Models

Several different end-of-life processes were modelled so that comparisons could be made. These processes and corresponding data used are:

Entire CRT sent to landfill.

This model involves simply taking all CRTs directly to a landfill site with the following transport and processing assumptions:

Distance from source to transfer station	15 km
Distance from transfer station to landfill site	100 km
Landfill data from above.	

Yoke separated and recycled with the remainder sent to landfill.

This model involves breaking off the yoke, recycling the copper component locally and sending the remainder to landfill. The following additional assumptions were made:

Distance from source to copper smelter	700 km

Proportion of copper recovered (based on Carlier et al 2003)	98%

All CRT glass sent to lead smelter.

This model involves separating the yoke and steel parts, recycling the copper, steel and some minor plastic, land-filling the electron gun and sending both glass fractions to a lead smelter. Here glass is melted with lead ore and most of the lead is recovered with the glass fraction acting as a flux to the smelting process. The following additional assumptions were made:

Distance from source to lead smelter	900 km
Energy to melt glass	1.05 GJ/ton
Avoided lead emissions	0.24 ton/ton glass
Amount of glass effectively recovered	98%

Entire CRT glass made into glass insulation batts.

This model involves separating the yoke and steel parts, recycling the copper, steel and some minor plastic, land-filling the electron gun and sending both glass fractions to be made into glass fibre insulation batts. The following additional assumptions were made:

Distance from source to batt manufacturer	30 km
Amount glass converted to fibre batts	95%
Avoided lead emissions	Nil

Funnel glass sent to lead smelter, panel glass recycled as normal glass, both in Australia.

This model involves separating the yoke and steel parts, recycling the copper, steel and some minor plastic, land-filling the electron gun and sending the funnel glass fraction to a lead smelter and the panel glass fraction to normal glass recycling. The following additional assumption was made:

Distance from source to local glass recycler	30 km

Funnel glass recycled as leaded glass, panel glass recycled as normal glass in Europe (Holland).

This model involves separating the yoke and steel parts, recycling the copper, steel and some minor plastic, land-filling the electron gun and sending both glass fractions to Europe where the funnel glass is recycled as leaded glass (new CRTS for example) and the panel glass is recycled as normal glass. The following additional assumptions were made:

Distance from source to wharves	20 km
Distance by ship to Europe	18,000 km
Avoided lead emissions	0.24 ton/ton glass

Results

Landfill or Recycle?

A useful initial comparison is simply between land-fill with and without recycling of the copper yoke and sending all glass components to Europe for recycling, tentatively assumed to be a worst case recycling scenario. The comparison is given in figure 3 where negative values indicate net environmental benefit (and all scales have been normalized to -1 for the largest score).

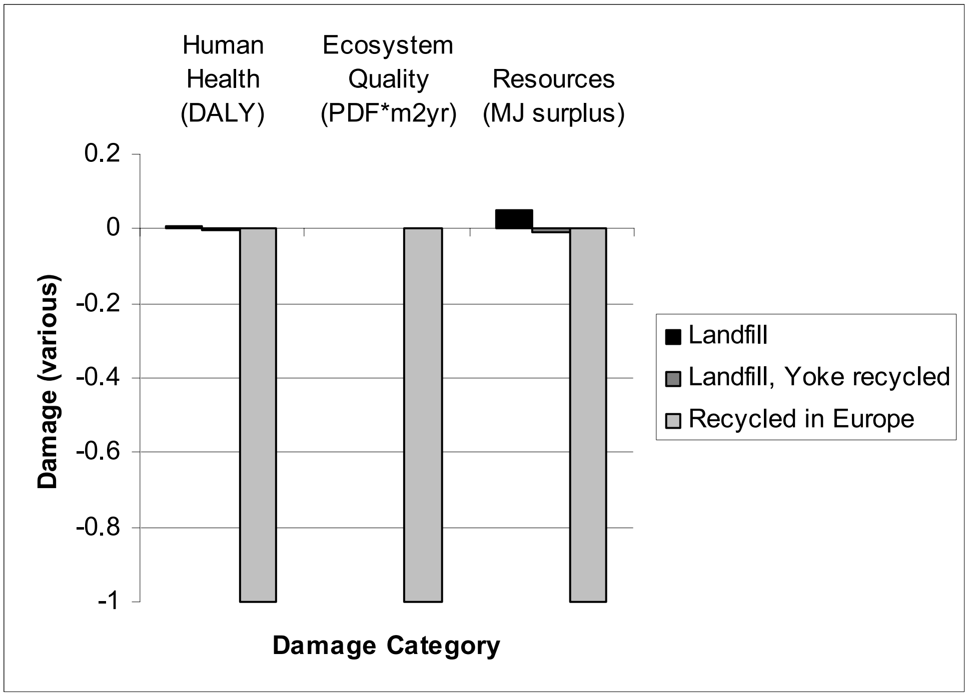


Figure 3. Comparison of two land-fill scenarios and a glass recycling scenario

These data suggest that sending the glass components to Europe for recycling is overwhelmingly beneficial compared to the small benefit or disadvantage due to either of the land-fill scenarios. As might be expected, simple and complete disposal to landfill causes net damage in all three categories of figure 3 and separating and recycling just the copper yoke results in a net but small benefit. In the latter case this means that the value of recycling the yoke just outweighs the costs of landfill of the remainder.

On the basis of the above, it was then taken as fact that land-filling was likely to be a less attractive possibility for CRTs and that some form of recycling was desirable. The remainder of this paper therefore concentrates on which recycling details might be preferred.

Recycling Possibilities

To establish a broad overall comparison of the different recycling possibilities practiced by industry, a broad comparison of the four different recycling scenarios described above was done for a particular CRT. Each of these scenarios include identical recycling of the copper yoke, steel parts and some minor plastic parts, with the only differences being in the treatment of the glass fractions. Some results are shown in figure 4.

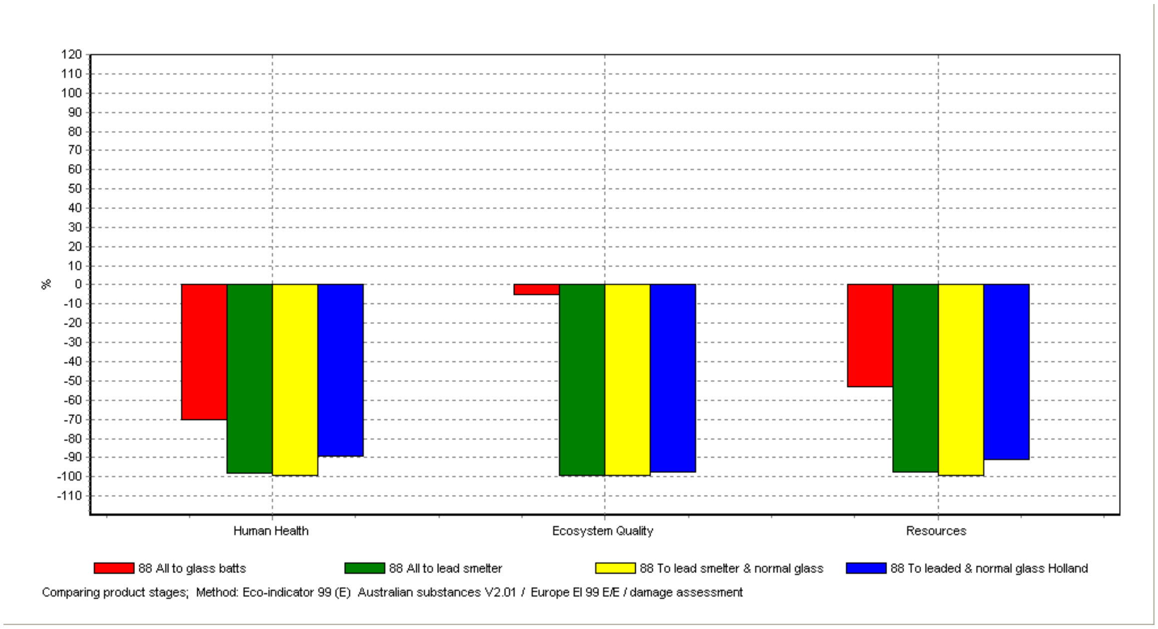
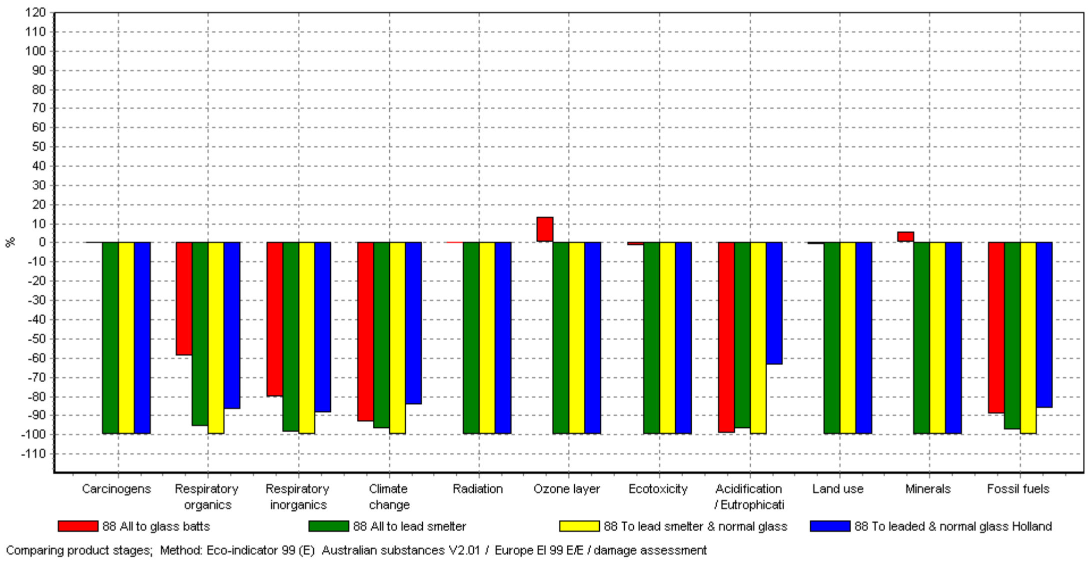


Figure 4. Comparison of four different recycling scenarios and their net impact on human health, ecosystem quality and resource use. Recycling scenarios in order from left to right are: Both glasses to insulation batts, Both glasses to lead smelter in Australia, Funnel glass to lead smelter and panel glass to normal recycling, Funnel glass to leaded glass in Holland and Panel glass to normal glass in Holland.

These data suggest that:

1. All recycling alternatives produce a net environmental benefit in all three damage categories;
2. Recycling of both panel glass and funnel glass to a lead smelter or recycling panel glass to normal recycling and funnel glass to a lead smelter are approximately equal in their impacts;
3. Both the recycling scenarios involving recycling the leaded glass through a lead smelter are slightly more beneficial on all criteria than the two other recycling scenarios; and
4. Transporting the glass fractions to Europe for recycling the funnel glass as leaded glass and the panel glass as normal glass is only slightly less beneficial (of the order of 10% or less) than local recycling;

In order to determine the actual causes of these impacts and the differences between the recycling scenarios, analysis on the finer scale of characterization was done with results shown graphically in figure 5.



Comparing product stages; Method: Eco-indicator 99 (E) Australian substances V2.01 / Europe EI 99 E/E / damage assessment

Figure 5. Characterisation of the four different recycling options. Recycling scenarios in order from left to right are: Both glasses to insulation batts, Both glasses to lead smelter in Australia, Funnel glass to lead smelter and panel glass to normal recycling, Funnel glass to leaded glass in Holland and Panel glass to normal glass in Holland.

As mentioned above, the categories carcinogens, radiation, ozone layer and land use will be neglected in this paper because their importance relative to the other categories is judged to be minor.

The reasons for the differences indicated in figure 5 are outlined in table 7.

One factor emerging from this analysis is that in general the environmental costs of transportation in the two cases involving transport over long distances - transport 900 km by road to Port Pirie and 18,000 km by ship to Europe - were similar in magnitude and both had only a small effect on the results.

Table 7. Principal reasons for differences between the four alternative recycling scenarios

Characterisation	**Factors in the Result**
Respiratory Organics	Recycling to batts is less beneficial than other options because of the use of fuel oil used in transport etc for new lead production to replace that lead tied up in batts. The transport fuel used for long distance transport to the lead smelter or to Europe are also major elements of the impact.
Respiratory Inorganics	All recycling options cause net environmental benefit mainly by avoiding virgin lead production with the batt option having less benefit because it does not avoid lead production. Transporting the product internationally by ship with consequent releases of several gases, causes the recycling in Europe to have less beneficial impact.
Climate change	All recycling options cause net environmental benefit mainly by avoiding virgin lead production with the batt option having less benefit because it does not avoid lead production. Transporting the product internationally by ship with consequent releases of several gases, causes the recycling in Europe to have less beneficial impact.
Ecotoxicity	Recycling to glass batts is much less beneficial because the lead content remains in the glass and is not recovered.
Acidification Eutrophication	The European recycling option is significantly less beneficial because shipping internationally releases nitrogen oxides and sulphur oxides from use of fossil fuels.
Minerals	Recycling to glass batts causes some net damage because the lead content remains in the glass and is not recovered and so virgin lead is required to replace it. The other three options recover useful lead.
Fossil Fuels	The net results here are similar for all four recycling options and are the result of several fuel use factors in each case. The energy avoided in the production of lead is a significant factor in the non-batt options.

Sensitivity Analyses

Since the data used in the study was subject to doubt in various ways, some sensitivity analyses were conducted to determine the impact variations in numerical values had on the outcomes. Variables which were tested in this way and the extent of their variations were:

> Energy required to melt glass for recycling – value doubled;
> Impact of production of virgin lead avoided – three alternative models of lead production for Europe;
> Location in Australia from which end-of-life CRTs were collected – Melbourne, Sydney and Brisbane;
> Variation in the amount of non-glass fractions of CRTs included in the analyses – all non-glass fractions (including the copper yoke) were excluded; and
> Lead leaching value – varied from 20 g, to 120 g to 240 g per tonne.

Sensitivity Analyses for Glass Melting Energy

The energy data for the recycling component was judged to be somewhat uncertain and so a sensitivity analyses on this data was conducted. The glass processing energy in the smelter or recycling facility where glass is melted was doubled from 1.05 GJ per ton to 2.0 GJ/ton, resulting in only about 2% change in damage in the worst case. This means that the sensitivity of the results to the somewhat uncertain energy data for melting the glass fractions is low and this uncertainty can be neglected.

Sensitivity Analysis for Avoided Lead Model

Several models were available in the database for production of virgin lead, all for use in Europe. Three of these were modified simply by deleting energy use in international shipping since a significant proportion of European lead is mined in Australia. The characterization analysis was repeated with each of the three models. Most characteristics showed little change from one model of lead production to another. However the amount of mineral use varied significantly over the range +2 to – 35 MJ surplus for the unit of study. The Impact on ecotoxicity (due to the toxic nature of lead) also varied over the range of -11 to -28 PDF*m2Yr. These results indicate that a valid analysis of the impact of CRT recycling would depend heavily on understanding of the particular process used to produce the lead actually being avoided by recycling the CRTs.

Sensitivity to Location in Australia

The analysis above was modelled on the CRTs' place of collection being Melbourne. However since transport is significant in some results, the transport assumptions were changed to suit other eastern Australian cities as shown in table 8. All transport in this case was assumed to be by road.

Table 8. Transport assumptions for recycling end-of-life CRTs

Transport	Melbourne Distances (km)	Sydney Distances (km)	Brisbane Distances (km)
From pick up to disassembly plant	30	30	30
From disassembly plant to lead smelter (Port Pirie)	900	1600	1900
From disassembly plant to local glass recycler	30	30	30
From disassembly plant to copper smelter (Port Kembla)	700	120	1300

Variations were mainly due to the differing distances from the Port Pirie lead smelter but were relatively small for most categories except perhaps for the following which are however mostly small enough to be neglected:

Respiratory organics	6%
Acidification-Eutrophication	5%
Use of fossil fuels	2.5%

This result reinforces the finding that the benefits of recycling CRTs apply even though the glass fractions may need to be transported large distances by truck.

Sensitivity to Excluding Non-glass Fractions

An identical analysis was conducted on just the glass fractions of the same CRT as above but also including the land-fill scenario. The relative performance of the different recycling possibilities are very similar in both scenarios, suggesting that most of the impacts are due to the glass fractions and that the non-glass fractions, including the copper yoke, make little overall difference. The data also suggests that the damage done by land-filling is small compared to the benefit due to recycling and that recycling should perhaps be thought of as being for the purpose of achieving benefits, rather than of avoiding damage.

Sensitivity to Lead Leaching Value

As described above, a figure of 120 g per ton of lead leached to water from land-fill of funnel glass was assumed for all the analyses above. Since this figure is contested by Jang & Townsend (2003) and Maccaulay et al. (2003) (as described above), it was varied in the model to determine any variations in outcome. The comparison between the two landfill scenarios and the same recycling scenario as shown in figure 3 was used to locate any differences. The only differences were in the area of ecotoxicity, some indication of which is given in figure 6.

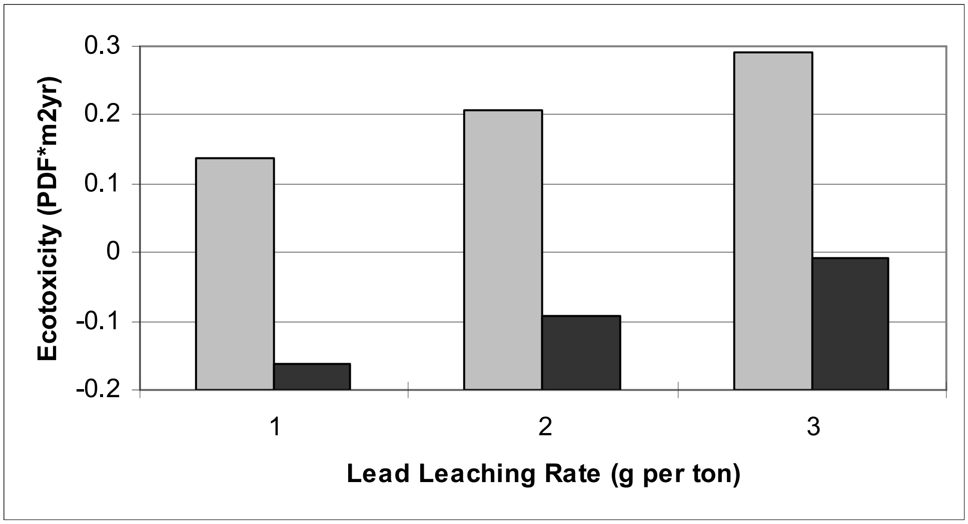


Figure 6. Variation in ecotoxicity due to land-filling the 88 cm CRT as lead leaching rate is varied from 20 g (1), to 120 g (2) and to 240 g (3) per ton. The positive figures are for land-filling the entire CRT (net damage) and the negative figures (net benefit) are for recycling the copper yoke and land-filling the remainder.

Figure 6 indicates simply that as the amount of lead leaching is increased, net damage to ecotoxicity from land-fill is increased. The quantities here however are relatively small being at most only about 0.1% of the corresponding benefit of recycling. It can thus be concluded that the figure for lead leaching assumed in all the earlier analyses does not have a major impact on overall results.

Conclusions

Conclusions about the alternatives ways in which CRTs could be disposed of at the end of their lives are:

1. Disposal of CRTs to landfill causes net damage in all three categories; human health, ecosystem quality and use of resources;
2. Separating and recycling just the copper yoke and land-filling the remainder results in a net but small benefit in the same three categories;
3. All glass recycling alternatives produce a net environmental benefit in all three damage categories;
4. The damage done by land-filling the glass is small compared to the benefit due to recycling;
5. Recycling of both panel glass and funnel glass to a lead smelter or recycling panel glass to normal recycling and funnel glass to a lead smelter are approximately equal in their impacts;
6. Transporting the glass fractions even as far as to Europe for recycling the funnel glass as leaded glass and the panel glass as normal glass is only somewhat less beneficial (of the order of 10% or less) than local recycling;

7. The benefits of recycling CRTs apply to all Eastern Australian major centres because the differences in road transport impacts are not major.

Consequently, even though there is no economic incentive to do so at the moment, recycling of CRTs should be encouraged because it clearly produces overall environmental benefits. Further, the possibility exists, through the lead smelter at Port Pirie at least, to handle many more CRTs than at present and the costs created by the related transport to that location are outweighed by the benefits.

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