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RADIOACTIVE METALS DISPOSAL AND RECYCLING IMPACT MODELLING

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ABSTRACT

Screening life cycle assessment models developed to investigate hypothetical disposal and recycling options for the Windscale Advanced Gas-cooled Reactor heat exchangers were used to generate more complex models addressing the main UK radioactive metals inventory. Both studies show there are significant environmental advantages in the metals recycling promoted by the current low level waste disposal policies, strategies and plans. Financial benefits from current metals treatment options are supported and offer even greater benefits when applied to the UK radioactive metals inventory as a whole.

1. Introduction

A significant proportion of current UK nuclear facilities are approaching or at the end of their operational lives. The volume of legacy radioactive waste [1] requiring disposal is growing as decommissioning programmes progress. Solutions must also be found for future radioactive wastes from new nuclear power stations, and materials not currently classed as waste but that may be in the future. There are a small number of UK lower activity waste (LAW) disposal repositories with limited capacity and operational life. There is no current disposal facility for higher activity wastes (HAW) hence interim storage facilities at nuclear sites are required. Treating, encapsulating and packaging waste for storage and disposal have high environmental and financial costs. Effort is needed to minimise the waste sent to these facilities. Metals constitute about 17% of low level waste (LLW) and 39% of intermediate level waste (ILW) [1] hence they offer an opportunity to reduce the waste volume through treatment and recycling.

Current UK LLW policy [2], strategy [3] and plans [e.g.4] have been developed to help minimise waste consigned for disposal. A raft of environmental, health and safety legal requirements and guidance including Strategic Environmental Assessment (SEA)[5], Environmental Impact Assessment, Best Available Techniques/Best Practicable Environmental Option (BPEO) and Best Practical Means plus As Low As Reasonable Achievable/Practicable (ALARA/ALARP) radiological assessments underpin policy requirements. They help structure the assessment of radioactive waste treatment and disposal impacts.

Metals are essentially infinitely recyclable and the LLW policy, strategy and plans promote this rather than disposal under the waste hierarchy principle. Treating and recycling as much radioactive metals as practicable offers significant savings in disposal volume and conservation of resources. LLW metals treatment and recycling BPEO studies [e.g.6] underpin current LLW policy, and life cycle assessments (LCA) of decommissioning nuclear power plants have been shown to identify significant environmental benefits from recycling metals from nuclear facilities [7] & [8]. No studies to date, have quantified the potential benefit treatment and recycling for the entire UK radioactive metals inventory.

This research compares the non-radiological environmental impacts of both disposal and recycling of surface contaminated LLW metals, including consideration of the treatment, packaging and transport through LCA models and preliminary economic cost analysis. We consider a number of plausible UK recycling options. Our analysis considers the location of treatment facilities and the allocation of high volume very low level waste (VLLW) to specified landfill disposal. We also consider the melting of volumetrically contaminated LLW metals [9] and intermediate level waste (ILW) metals decayed or decontaminated to LLW for waste volume reduction.

2. Life Cycle Assessment of radioactive metals treatment and packaging

2.1 Goal and Scope

The goal was to develop screening LCA models to compare the environmental impacts of radioactive metal disposal and treatment for recycling within the nuclear industry or metal markets. Scenarios for analysis were selected to help inform radioactive waste management decision-making to minimise the use of scarce UK radioactive waste disposal capacity.

Our research considered two inventories. First, we investigate a case study of the Windscale Advanced Gas-cooled Reactor (WAGR) boilers to identify interesting disposal and recycling scenarios and the sensitivity to individual processes. Secondly, we explore the environmental and economic consequences of four disposal and recycling options for VLLW, LLW and ILW metals in the full UK inventory [1]. We considered steels (~71% of LLW metals and ~87% of ILW metals), aluminium, copper, lead, nickel and zinc. Low volume VLLW and radio-toxic high level waste (HLW) metals were excluded from the study. Finally we combine our environmental analysis with a simple estimate of the financial costs for each scenario.

2.2 LCA Methodology

The software package SimaPro 7.3.3 (PhD) with the embedded Ecoinvent 2.2 database was used for the LCA modelling. SimaPro was released in 1990, has been updated regularly and is widely used internationally by industry and universities. LCAs, ranging from buildings analysis to civil aircraft design [10] have been conducted. The Ecoinvent database has been available since the late 1990s and also updated regularly. Version 2 has in excess of 4000 datasets [11], is commonly used in all major LCA software packages and it can be regarded as the industry standard.

The LCA models were constructed via sub-assemblies of waste metals, container metals and production and transport. The sub-assemblies were linked to end-of-life processes for VLLW, LLW and ILW disposal and recycling to form integrated life cycles. Waste treatment processes for disposal and recycling included size reduction, decontamination, induction melting, secondary waste processing, avoided future metals (including re-melting) for recycling and the return of secondary waste to the UK for disposal. Disposal included the internal grouting of the WAGR boilers disposed whole, half height ISO (HHISO) freight containers for LLW metals and 4m boxes for ILW metals. It also included the external grouting of the WAGR boilers as required by the Environmental Safety Case for the UK LLW Repository near Drigg, in Cumbria.

The Life Cycle Impact Assessment (LCIA) method adopted throughout was the Eco-indicator 99 damage orientated method [12]. The weighted impact results calculated were Eco-indicator points (Pt) representing a dimensionless unit to compare relative differences between process and material impacts of different environmental options. "…1Pt is representative of one thousandth of the yearly environment load of one average European inhabitant" [12]. This method was selected because the impacts are broadly similar to the environmental impacts addressed in the LLW SEA [5] and can be used to make environmentally informed decisions on material choices [13]. The Eco-indicator points allowed for quantitative comparison of the disposal and recycling environmental impacts that can underpin existing radioactive waste policy and inform future decisions in areas of known gaps.

The WAGR case study data were predominantly derived from the nuclear operator [14] and the VLLW, LLW and ILW metals data from the UK inventory [1]. VLLW and LLW disposal, treatment, recycling and transport costs were taken from the LLW Repository website Waste Services Contract documents and Joint Waste Management Plans [e.g. 4] (www.llwrtsite.com). Additional cost data were derived from published metals treatment contracts [15], [16], [17] and [18]. Waste container data were taken from the Nuclear Decommissioning Authority (NDA) website documents (www.nda.gov.uk) plus container designer and manufacturer websites. For example materials and external volume data for the 4m box were based on published information from Croft Associates Ltd (www.croftltd.com). Similar data for the HHISOs were taken from Yorkshire Marine Containers (www.ymccontainersolutions.com) and from James G. Carrick & Co Ltd (sales@jamesgcarrick.com) for the 210 litre(l) mild steel drums.

Modelling Assumptions

A number of general assumptions were common to both case studies. Materials and process data for UK radioactive waste disposal facilities was not readily available. However the Eco-invent database has extensive data for Swiss low activity waste, LLW and ILW disposal facilities [19]. The low activity disposal impact includes data only for land use and energy to dig the near surface trenches for the repository. The LLW and ILW disposal impact data are for the material, energy and transport for creating and backfilling the shafts and tunnels, wastewater from mining and disposal plus land use and buildings for surface facilities. The Swiss ILW disposal impact data is broadly comparable with UK ILW disposal as both are engineered geological vaults [7]. The Swiss LLW repository is also an engineered geological facility whereas the UK LLW repository is an engineered near surface facility, hence the Swiss data are not entirely comparable. However, in the absence of other data it was assumed the Swiss LLW disposal impacts could be used for UK LLW disposal. The Swiss low activity waste disposal impact data are for a low engineered landfill facility and hence regarded as comparable with UK VLLW disposal facilities.

Data Assumptions

An average distance, weighted by the mass of radioactive metals at each site, was used for transport distance between nuclear sites and the LLW Repository (Drigg) and the distance to the Studsvik Metal Recycling Facility (MRF) in Cumbria. This approach was also used to represent the distance to a future GDF, assumed to be near Sellafield. This was a conservative assumption as the majority of the metals arise at Sellafield. However, modelling has shown that road transport impacts are significantly lower than disposal and waste container impacts hence the results are relatively insensitive to this parameter.

In addition to the MRF, and decontamination facilities at some sites, UK nuclear operators use international treatment/melting facilities in Germany, Sweden and the USA, but currently not those in France or the Russian Federation. Modelling suggests that there were small difference in the life cycle impacts between treatment facilities in the EU and in Russia. Hence it was assumed that Studsvik's facility in Sweden was representative of a European LLW metals treatment/melting facility. The transport distances from Hull to Stockholm and Stockholm to Nykoping were therefore added to the weighted average distance discussed above for treatment in Sweden.

It is conceivable that some metals might require too much effort, or not be ALARP, to treat and recycle. In the absence of data it was assumed that 5% of all VLLW and LLW metals were disposed directly to the appropriate repository. The remaining 95% of surface contaminated metals were treated for recycling but generated 5% secondary waste for disposal. It was further assumed that ~20% of the LLW metals were actually VLLW [1] and ~14% LLW metals were volumetrically contaminated [9]. ILW that could decay or be decontaminated to LLW is ~3% of the ILW inventory. This value increases to ~9% if it is assumed to apply only to ILW metals. A factor of 20 for melting volumetrically contaminated LLW and ILW metals decayed/decontaminated to LLW was assumed for waste volume reduction to match the 5% disposal. This is rather optimistic and warrants further study.

Waste container impacts were anticipated to be important hence a container production impact was added to the container metal impact. It was suggested by a highly experienced LCA practitioner that the Ecoinvent average metal working impact for metal product manufacturing would be a reasonable assumption at a modelling meeting with Intertek Ltd. As a consequence this was used for all containers in the study. It was also suggested that the impact could be used as a decontamination proxy. A 5% by mass average metal working proxy was used for LLW decontamination to match the 95% recycling assumption. Since VLLW contamination would be lower than LLW contamination a 1% by mass proxy was used for VLLW decontamination.

An induction melting electrical load impact was derived for the WAGR case study based on the mass of the 4 boilers, the anticipated size reduction for the furnace loading and melt duration. The Swedish medium voltage with imported supply was assumed for this melting. It was found that the induction impact was about 1/10th of the impact of melting the equivalent metal mass in an electric arc furnace (EAF). The 10% EAF impact was assumed for melting LLW and ILW metals for the UK metals inventory investigation.

WAGR size reduction data were estimated from cutting up the boilers to meet the induction furnace loading requirements. This was not practical for the UK metals inventory investigation hence size reduction was estimated from data in the decommissioning LCA [7].

3. Environmental impact assessment results and interpretation

3.1 Case Study -WAGR boilers

The two disposal scenarios were direct disposal of the boilers whole to the Drigg LLW Repository (which occurred in 1996) and packaged disposal in nominally 50 HHISOs. Direct disposal is allowed in the current policy under exceptional circumstances so the scenario was retained. The original estimate of HHISOs for packaged disposal of the boilers was unknown. The planning norm for disposal in the LLWR websites documents [e.g 4] is 10tonne(te)/HHISO. Magnox boiler disposal from the UK inventory [1] and BPEO study [6] show waste loadings from 11te to 22te per container. Hence an average loading of 15te/HHISO (i.e. 50 HHISOs) was assumed for packaged disposal. The two recycling scenarios were transporting the boilers whole to Sweden treatment/recycling, and cutting them up in the UK for transport to Sweden in nominally 50 trips for processing.

The results of the four scenarios are presented in Figures 1 and 2. Eco-indicator 99 results for each scenario are in weighted points as discussed in section 2.2, but the results are shown as percentages of the total direct disposal impact (TDDI) for ease of comparison between them. Positive values represent adverse environmental impacts; negative values represent an environmental benefit.

Figure 1 shows that both disposal options have significantly larger environmental impacts than either recycling option. It also shows that the packaged disposal would have resulted in a 45% increase in disposal impact compared to direct disposal. This is because the packaged disposal waste volume is ~25% higher than direct disposal and the HHISO metal and production impacts of packaged disposal were 18 to 19% of the direct disposal total impact respectively. The external grouting for direct disposal has a similar percentage impact. Container impacts are significantly larger than transport impacts and other process impacts across all four scenarios.

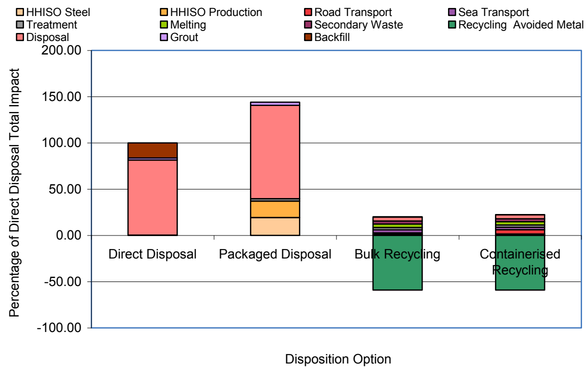

Figure 1 Environmental impacts of the four disposal and recycling options for the WAGR boilers as a percentage of the direct disposal option.

Figure 1 shows there is only a 6% difference between the recycling scenarios. In general in the recycling scenarios transport, treatment (i.e. size reduction, decontamination, melting and secondary waste production) and disposal result in adverse impacts of ~20 to ~22% of the total direct disposal total impact. Waste container impact for containerised recycling was insignificant as it was assumed that two HHISOs are reused for the return of secondary waste to the UK for disposal. Recycling the melt ingots in the Swedish iron and steel industry results in a benefit equivalent to ~60% TDDI (shown as a negative values in Figure 1) for the future avoidance of virgin materials or other scrap. Hence the net benefit of treatment with recycling was ~38 to ~40% of TDDI.

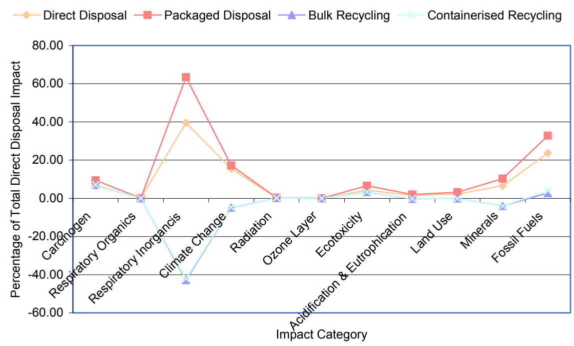

Figure 2 Percentage of TDDI by impact category for each scenario for the WAGR boilers.

Figure 2 presents the results by impact category for each of the four scenarios. As expected the packaged disposal impact category results are all higher than the direct disposal impacts, particularly for respiratory inorganic compounds, fossil fuels and mineral. Impact is dominated by the respiratory inorganic compound impacts for dust particles, NO_2 /NO_x, SO_2/SO_x and ammonia discharged to air. This is followed by fossil fuel impacts calculated from energy use and by climate change impacts from emissions of green house gases.

The minerals, carcinogen and eco-toxicity form a secondary impact group with land use and acidification/eutrophication impact as a tertiary group. Impacts from radiation, respiratory organics and ozone layer depletion are negligible.

Key to note here is that the avoidance of future virgin material and other scrap by recycling the melt ingots provides a significant net benefit for respiratory inorganic compounds (~40%), climate change (~5%) and minerals (~4%). This material avoidance also reduces the adverse impacts of the other impact categories for both recycling options, especially fossil fuels.

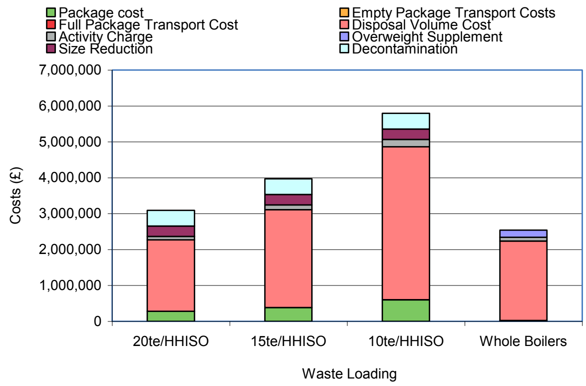

Figure 3 Comparison of estimated costs for disposal options for the 4 boilers with disposal costs for a roughly equivalent mass of Magnox metals using planning norm values [4] of 10te/HHISO (i.e. 76 HHISOs), 15te/HHISO (i.e. 50 HHISOs) as an average waste loading and 20te/HHISO (i.e. 38 HHISOs) as a maximum waste loading.

Figure 3 presents simple cost estimates for the two disposal options based on £2911/m^3 volume disposal cost (LLW Repository website) and explores the sensitivity to different packaging assumptions.

Decontamination and size reduction will depend on the level of contamination, its location, the ALARP requirements for operator dose and the desired container waste loading. For simplicity our decontamination and size reduction costs were estimated as fixed costs of £570/te and £380/te respectively from historical American studies [17] and [18]. The costs were converted from $ to £ using the average annual exchange rate for the year and inflated to 2012 values. The overweight supplement applied only to the whole boiler disposal and was estimated as 8% of the direct disposal total costs.

The cost of disposing of boilers whole is lower than any of the packaged disposal options and estimated to be about 44% of the maximum packaged disposal cost. These estimates compare with a historic estimate of a one third reduction in cost for HHISO disposal by the nuclear operator [20]. The disposal volume and package costs are half that of the 10te/HHISO planning norm if the maximum loading of 20te/HHISO is achieved. The activity charge, taken as a percentage of the disposal volume cost, also halved. This assumption may be too crude but it is a small cost component in these cases. Transport costs are <1% in all cases. Although the calculation methods are slightly different, our estimated disposal costs for the 10te/HHISO planning norm are in good agreement with industry estimates of £5.9m for the disposal of an equivalent tonnage of Magnox metals [4].

To estimate the cost of treatment and recycling we have based our estimates on the Berkeley boiler recycling contract costs of £5200 to £5800/te [15] and [16]. This gives values between £4 – 4.4m. Again, this compares well with the planning norm for a treatment cost of an equivalent tonnage of Magnox metals, which is estimated as £3.6m (£4810/te) [4]. The cost of recycling is, therefore, approximately equivalent to that for disposal with a packing density of 15te/HHISO.

The results presented in Figures 1 to 3 suggest that there are significant environmental benefits from treating and recycling radioactive metals rather than disposing of them and the economic costs are broadly similar. In this case, the cheapest option was actually to dispose of the boilers whole. This is because the thick walled boiler shells functioned as containment for the internal radioactive contamination, resulting in a smaller volume for disposal than any of the packaged disposal options. Such whole disposal of large contaminated plant without containment is not generally possible.

3.2 The Full UK Radioactive Metals Inventory

LCA models were generated for the UK VLLW, LLW and ILW metals inventory for scenarios comprising worst and best case disposal plus international treatment, with and without recycling. Studsvik's Swedish treatment facility is taken as a reference European facility.

Worst case disposal assumed that VLLW metals could not be segregated from LLW metals and that both were disposed in grouted HHISOs to the LLW Repository (Drigg) with the10te/HHISO waste loading. This reflected the UK position prior to the 2007 solid LLW policy [2]. It also assumed that all ILW metals were disposed to a future GDF based nominally at the weighted average distance from each nuclear site to Sellafield in grouted stainless steel 4m boxes with a 10te/4m box waste loading. Best case disposal assumed that 20% of LLW could be segregated as VLLW and disposed of in licensed VLLW landfill sites in ungrouted 210l drums rather than grouted HHISOs. The remaining LLW metals were disposed to the LLW Repository (Drigg) with the average waste loading of 15te/HHISO. It also assumed as before, that ~9% ILW metals decayed/decontaminated to LLW was disposed at Drigg rather than a future GDF. The nature and handling requirements of ILW metals makes a 15te/4m box unlikely, hence a target ~12te/4m box was assumed for disposal at Drigg in mild steel 4m boxes or HHISOs. The results of the disposal models are shown in Figures 4.

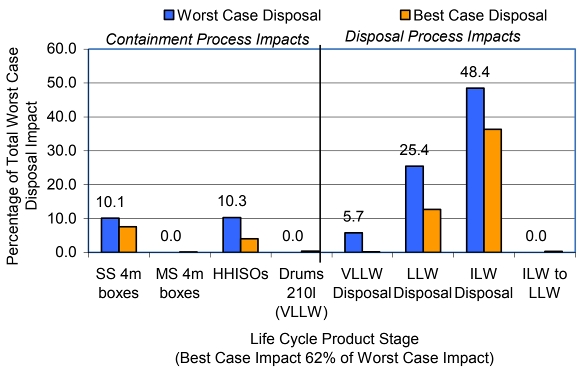

Figure 4 Comparison of worst case and best case disposal options for UK radioactive metals.

Figure 4 compares impacts of the containment process and the disposal process for both the worst and best case disposal scenarios. The impact is dominated by the ILW disposal followed by LLW disposal. Container impacts and VLLW disposal impacts are also significant (but less so) in both cases. The disposal process accounts for ~80% of the total worst-case disposal impact (WCDI) with ~20% being from container metal, production and transport. The stainless steel 4m boxes have a higher material and production impact than the HHISOs despite the much lower number of boxes.

Best case disposal gives ~38% reduction in environmental impact compared to the total WCDI: 13% from increasing the container load from 10te/HHISO to 15te/HHISO for LLW; 12% from 12te/4m box for ILW; 5% from VLLW disposal to specified landfill; 6% from changing the VLLW containers from grouted HHISOs to ungrouted 210l drums; and 2.5% for packaging the ILW decayed/decontaminated to LLW in mild steel 4m boxes or HHISOs. This is off set slightly by <1% increase in LLW disposal impact.

These results demonstrate the environmental benefits achieved by introduction of the 2007 UK LLW policy that requires the treatment and recycling of LLW and VLLW metals. Our results also imply that significant benefits could be obtained if the UK applied similar logic to ILW metals.

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Figure 5 compares the environmental impacts of disposal with those for treatment and recycling in Sweden for the whole UK inventory. The figure shows that the ILW impacts (which include container and disposal) are the same for all three scenarios, as they are not involved in the recycling process and still dominate the overall impact. International treatment without recycling (i.e. without the future metals avoidance) shows a ~47% reduction from the total WCDI. Including recycling reduces the impact further to ~38% of total WCDI. Hence, international treatment without and with recycling represent a 15 % to 24% reduction in environmental impact compared to best case disposal.

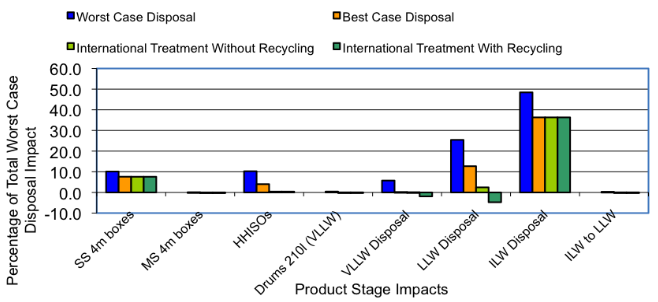

Figure 5 Comparison disposal scenarios and international treatment with and without recycling. A negative value indicates an environmental benefit.

Analysis of Figure 5 shows that the environmental benefits of treatment and recycling are principally derived from a reduction in LLW and VLLW disposal. Substantial benefits are also derived from the reduction in HHISOs and 210l drums respectively. Recycling surface contaminated LLW metals, thus avoiding future metals, gave an additional benefit of ~5% of total WCDI with ~2% for recycling VLLW metals.

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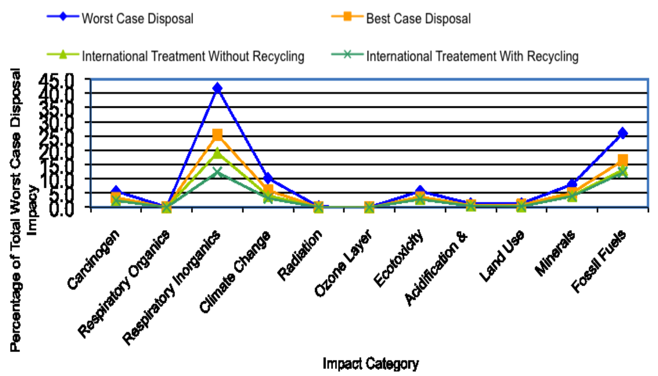

Figure 6 Eco-indicator 99 impact category results for Worst Case disposal and International Treatment, without and with recycling, as percentages of the total Worst Case disposal impact.

Figure 6 shows a progressive reduction in impacts from worst case disposal through to treatment with recycling. The addition of recycling, as opposed to just treatment. produces a further improvement to respiratory inorganic, climate change, mineral and fossil fuel impacts due to the avoidance of future metal extraction and processing, which incurs associated energy requirements.

Estimated costs for worst and best case disposal and for international treatment and recycling are presented in Table 1. The transport and activity costs are small by comparison to other costs and hence have not been included. Costs are split by components of the inventory in each case.

	ILW metals	ILW Metals decayed to LLW	LLW Metals	VLLW Metals	Total Costs
Worst Case Disposal	£3550m to £4730m	Not Applicable	£5030m	Not Applicable	£8580m to £9760m
Best Case Disposal	£2660 to £3550m	£53m	£2730m	£45m to £96m	£5488m to £6429m
Treatment And Recycling	£2660 to £3550m	£47m to £56m	£1973m to £2283	£88 to £93m	£4768m to £2432m

Table 1 Comparison of disposal and treatment/recycling costs. No allowance is made for the revenue from selling the melt ingots as scrap or for products for the nuclear industry.

The results in Table 1 show that best case disposal (i.e. assuming the increased ILW and LLW metal waste loading, the LLW disposal of 9% ILW metal decayed/decontaminated to LLW and segregating VLLW metal) saves £3 – 3.3b. The benefit of treatment for recycling of LLW and VLLW metals saves a further £0.5 - 0.7b. ILW containment and disposal are not affected by treatment and recycling hence they represent a high fixed cost. Although no cost recovery from the sale of scrap metal is included in Table 1 an American feasibility study [18] assumed that 3% of the treatment cost could be adopted, which would result in a further saving of £0.07-0.14m if recycling were included.

4. Discussion and Conclusions

We present comparisons of disposal and recycling scenarios for two inventories; the WAGR boilers case study and the current UK radioactive metals inventory. We use LCA analysis to indentify and estimate the environmental impacts and simple economic costs to compare each scenario. These results demonstrate the environmental benefits that have been achieved since 2007 through applying the current UK LLW policy, which requires the treatment and recycling of LLW and VLLW metals. Our results also imply that significant benefits could be obtained if the UK applied similar logic to the ILW metals

Overall environmental impacts and costs are dominated by ILW disposal. ILW, LLW and VLLW metal environmental impacts and costs can be reduced by increasing waste container loading for disposal and by treatment to minimise disposal. Recycling LLW and VLLW metals, thus avoiding future metals for new products for the nuclear industry or as scrap, offer further substantial improvements. The savings from these activities could be used to fund a future UK treatment/recycling facility to make the UK self sufficient in radioactive waste management enhancing the national economy and industrial knowledge and capability.

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