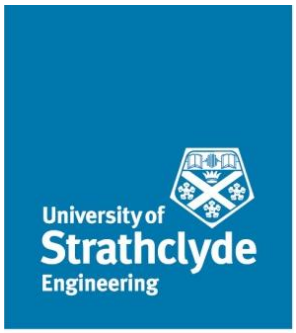
Assessing the Options: Disposal, Recycling or Re-Use of UK Legacy Radioactive Metals and Depleted Uranium

Neil W. Kemp



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Date:

Abstract

The research presented in this thesis investigates scenarios to preserve scarce radioactive waste disposal capacity of current and future UK repositories. Simplified life cycle assessment (LCA) and cost estimates are used to evaluate end-of-life options for UK radioactive metals and depleted uranium. The thesis builds on the work of previous authors by considering a wider range of potential environmental impacts and by focussing in more detail on the processes inherent in waste management and decommissioning to support future decision making.

Credible disposal and treatment scenarios are developed for LCA of UK radioactive metals - initially as a hypothetical case study - then applying the research method to the entire UK radioactive metals inventory. These scenarios identify key processes for the decontamination and melting of radioactive metals for recycling in open markets or for products to re-use in the nuclear industry. The thesis uses the experience gained from investigating the UK metals inventory to explore limiting scenarios for the disposal of the UK's depleted uranium, assuming that it is classified as a future waste.

The metals inventory research confirms that disposal and 'single-use' steel waste container impacts dominate the overall environmental impacts and financial costs - the impacts and costs of low level waste transport are negligible. Significant reductions in impacts and costs can be achieved by treating UK radioactive metals for recycling and re-use. The depleted uranium research indicates that the proposed current baseline for disposal is not optimal – impacts are dominated by disposal volume and other packaging options offer significant volume reduction- the environmental impacts and costs of packaging depleted uranium for the long-term storage are negligible.

The thesis offers recommendations for industry and for future research for number of possible longer-term management options for UK radioactive metals and depleted uranium.

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Acronyms

AGR	Advanced Gas-cooled Reactor
BAT	Best Available Technique
BPEO	Best Practicable Environmental Option
BPM	Best Practical Means
CoRWM	Committee on Radioactive Waste Management
DALY	Disability Adjusted Life Years
DDS	Direct Disposal Scenario
DECC	Department of Energy and Climate Change
DEFRA	Department for Environment, Food and Rural Affairs
DNLEU	Depleted, natural and low enriched uranium
EI99	Eco-Indicator 99
FHISO	Full Height ISO freight container
GDF	Geological Disposal Facility
GHG	Greenhouse Gas
HAW	Higher Activity Waste
HHISO	Half Height ISO freight container
HLW	High Level Waste
HMG	Her Magesty's Government
IAEA	International Atomic Energy Agency
ILW	Intermediate Level Waste
ISO	International Standards Organisation
JWMP	Joint Waste Management Plan
LAW	Low Activity Waste
LCA	Life Cycle Assessment
LCIA	Life Cycle Impact Assessment
LLW	Low Level Waste
LLWR	Low Level Waste Repository
MoD	Ministry of Defence
MJ	Mega Joule
NDA	Nuclear Decommissioning Authority

NORM	Naturally Occurring Radioactive Material
OECD	Organisation for Economic Co-operation and Development
PAF	Potentially Affected Fraction
PDF	Potentially Disappeared Fraction
PWR	Pressurised Water Reactor
RWM	Radioactive Waste Management
SDS	Shallow Disposal Scenario
SEA	Strategic Environmental Assessment
SEPA	Scottish Environment Protection Agency
TBI	Total Baseline Impact (for DNLEU disposal)
TBDI	Total Baseline Disposal Impact (for UK radioactive metals inventory)
TDDI	Total Direct Disposal Impact (for WAGR case study)
UKRWI	United Kingdom Radioactive Waste Inventory
VLLW	Very Low Level Waste
WAGR	Windscale Advanced Gas-Cooled Reactor
YLD	Years Lived Disabled
YLL	Years of Life Lost

Glossary

Activation	"the process of making a radioisotope by bombarding a stable element with neutrons or protons" (United States Nuclear Regulatory Commission website 2010)
Activation products	radionuclides produced by the interaction of neutrons with stable nuclides, for instance cobalt-60 an isotope of cobalt.
Backfill	"The material used to fill in and close off the void areas of an underground repository, such as vaults, silos, and drift tunnels, which usually occurs after the radioactive waste has been emplaced; thus "backfilling the waste"." (Committee on Radioactive Waste Management (CoRWM) 2006)
Becquerel	"The standard international unit of radioactivity equal to one radioactive transformation per second" (Her Magesty's Government (HMG) 1995).
Best Practicable Environmental Option (BPEO)	BPEO is "A concept developed by the Royal Commission on Environmental Pollution, it implies that decisions on waste management have been based on an assessment of alternative options evaluated on the basis of factors such as the occupational and environmental risks, the environmental impacts, the costs and social implications." (HMG1995)
Best Practicable Means (BPM)	"Within a particular waste management option, the BPM is that level of management and engineering control that minimises, as far as practicable, the release of radioactivity to the environment whilst taking account of a wider range of factors, including cost-effectiveness, technological status, operational safety, and social and environmental factors. In determining whether a particular aspect of a proposal

	represents the BPM, the Inspectorates will not require the applicant to incur expenditure, whether in money, time or trouble, which is disproportionate to the benefits likely to be derived." (HMG1995)
Clearance Levels	"A set of values established by the regulatory body in a country or state, expressed in term of activity concentration and/or total activities, at or below which sources of radiation can be released from nuclear regulatory control." (International Atomic Energy Agency (IAEA) 1995)
Concentration and Containment	"The preferred way to do this, where reasonably practical, is to concentrate and contain the waste and to isolate it from the environment. This allows any releases to the environment to be restricted and subject to regulatory control". (Health and Safety Executive (HSE) et al. 2007)
Contamination	"Undesirable radiological, chemical or biological material (with a potentially harmful effect) that is either airborne, or deposited in (or on the surface of) structures, objects soil, water or living organisms in a concentration that makes the medium unfit for its intended use" (United States Nuclear Regulatory Commission website 2010)
Conditioned waste	"Radioactive waste that has been treated or processed in preparation for packaging". (CoRWM 2006)
Decommissioning	"Generic term to cover all the procedures undertaken once a nuclear installation has ceased operating. Decommissioning covers process such as defuelling reactors, clean-out and making safe an installation, dismantling and removal of structures, and waste conditioning prior to storage or disposal". (CoRWM 2006)
Decontamination	"The complete or partial removal of contamination by a deliberate physical, chemical or biological process". (CoRWM 2006)

Depleted uranium	**"**Uranium where the uranium 235 isotope content is below the naturally occurring 0.72% by mass" (Department for Environment, Food and Rural Affairs (DEFRA) et al. 2007)
Dilute and Disperse	"A term normally describing a form of management for radioactive waste where radioactivity is released from a facility as a gas or liquid and is diluted in the air or marine environment". (CoRWM 2006)
Disposal	"In the context of solid waste, disposal is the emplacement of waste in a disposal facility without the intent to retrieve it at a later time; retrieval may be possible but, if intended, the appropriate term is storage. Disposal can also refer to the release of airborne or liquid waste to the environment (i.e. emissions and discharges)." (HMG1995, DEFRA et al 2007)
Fission products	"Radioactive elements produced by nuclear fission through the spontaneous or impact-induced splitting of a heavy atomic nucleus accompanied by a release of energy" (CoRWM 2006).
Geological disposal	"Disposal refers to long-term management options where future access or future changes in management are not intended. Geological disposal usually refers to a long-term management option involving the emplacement of radioactive waste in an engineered repository at between 200 metres and one kilometre underground where the geology (rock structure) provides a barrier against the escape of radioactivity." (CoRWM 2006)
Gigabequerel (GBq)	"A unit of radioactivity equal to one thousand million becquerels. When divided by weight (e.g. GBq/te) this provides a measure of the concentration of radioactivity" (HMG1995)
Grouting	"A means of encapsulating radioactive waste by mixing it with, for example, cementatious material." (CoRWM 2006)

Higher Activity Waste	The generic name encompassing low level waste (LLW) that cannot be disposed to the LLWR repository, intermediate level waste (ILW), high level waste (HLW) and spent nuclear fuel (SNF).
High Level Waste	Radioactive waste with an activity above intermediate level waste (ILW) that requires heat generation to be taken into account for handling, storage, transport and disposal.
High Volume VLLW	The 'bulk disposal' of very low level waste (VLLW) to designated landfill facilities.
Ionising radiation	"Radiation that produces ionisation in matter, for example, alpha particles, gamma rays, x-rays and neutrons. When radiations such as these pass through the tissues of the body, they have sufficient energy to damage DNA." (CoRWM 2006)
Intermediate Level Waste	Radioactive waste with an activity above low level waste (LLW) but does not require heat generation to be taken into account in its handling, storage, transport and disposal.
Kilobecquerel (KBq)	"A unit of radioactivity equal to one thousand becquerels" (HMG1995)
Long-lived waste	"Radioactive waste that contains radionuclides that have a half-life of more than 30 years." (CoRWM 2006)
Low Activity Waste	A generic term encompasing very low level waste (VLLW) and low level waste (LLW).
Low Level Waste	Radioactive waste with a low activity but which must be consigned to a fully authorised and dedicated LLW facility/
Low Volume VLLW	The 'dust bin' sized loads of very low level waste (VLLW) to municipal, commercial or industrial landfill
Nirex	UK Nirex (Nuclear Industry Radioactive Waste Executive) Ltd was a company jointly owned by DEFRA and the Department of Trade and Industry (DTI) that advised nuclear site operators on the preparation of safety submissions to the

	regulators for the conditioning and packaging of radioactive waste (CoRWM 2006). Nirex has been part of the NDA since 2007.
NORM	"The enhancement of naturally occurring radionuclides … due to human activity" (Bayliss and Langley 2003). May also be called Technologically enhanced naturally occurring material (National Council on Radiation Protection & Measurement (NCRP) 2002).
NDA	The Nuclear Decommissioning Authority is a…….. of the Department of Energy and Climate Change (DECC).
Overpacking	"A secondary (or additional) outer container for one or more waste packages, used for handling, transport, storage and/or disposal." (CoRWM 2006)
Packaged radioactive waste	"The product of conditioning that includes the waste form and any container(s) and internal barriers, prepared in accordance with the requirements for handling, transport, storage and/or disposal." (CoRWM 2006)
Partitioning	"The separating out, by physical and chemical methods, of radioactive elements contained in a waste stream to permit their further treatment." (CoRWM 2006)
Passive safety	"Passive safety describes a situation where no intervention is required to keep the waste in a condition where it poses no threat to health or safety. The waste does not require additional work or processes to be carried out to keep it in a safe condition." (CoRWM 2006)
Radioactive Inventory	database of physical and chemical properties of high, intermediate, low and very low level radioactive waste
Radionuclide	"A nucleus (of an atom) that possesses properties of spontaneous disintegration. Nuclei are distinguished by their mass and atomic number." (CoRWM 2006)

Short-lived nuclides	"Radioactive nuclides with a half-life less than 30 years. Thus, radioactive waste described as short-lived would reduce in activity by a factor of 1000 within 300 years." (CoRWM 2006)
Sievert	"The S. I. unit of radiation dose; one millisievert (mSv) is a thousandth of a sievert and one microsievert (uSv) is one millionth of a sievert." (CoRWM 2006)
Spent Nuclear Fuel	Irradiated fuel from nuclear power stations that has not been reprocessed.
Tonne (te)	a unit of mass (10^6 or 1,000,000 grams) (CoRWM 2006)
Treatment	"Operations intended to benefit safety and/or economy by changing the characteristics of the waste. Three basic treatment objectives: 1. volume reduction 2. removal of radionuclides from the waste 3. change of composition After treatment, the waste may or may not be immobilized to achieve an appropriate waste form." (IAEA 1995)

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1 Introduction

The United Kingdom (UK) defence nuclear industry started in the mid 1940s. The civil nuclear research and energy industry developed from the 1950's until a hiatus in the 1990s. A significant proportion of the legacy UK nuclear facilities are now approaching, or at the end of, their operational lives and entering the decommissioning and dismantling phase.

Growing concerns about the energy gap, security of supply and the adverse environmental effects from burning fossil fuel for electricity has seen renewed interest in the nuclear option. A proposed fleet of eight new nuclear power stations is underway, starting with the licensing and initial preparations for Hinkley Point C in Somerset. New nuclear power stations are expected to come on-line over the next two decades to replace the shutdown of legacy power stations.

The nuclear industry has given the UK significant economic and social benefits since its inception. It has however proved contentious with the public and raises issues especially after accidents such as Three Mile Island, Chernobyl and Fukushima. The principle areas of concern are nuclear safety and the management of radioactive wastes. Nuclear safety is outside the scope of this research. Radioactive waste management, in particular radioactive metal recycling and reduced disposal, are the subject of this research.

1.1 Research Context

This section contains an explanation of wastes in general, metals and the research focus.

Radioactive and non-radioactive wastes are produced in the construction, operation, decommissioning and dismantling of all nuclear facilities. An estimated total of 4.9 million tonnes of legacy radioactive wastes is expected to arise until completion of legacy facility decommissioning, circa 2110. This is very small compared with the estimated 300 million tonnes of non-radioactive waste produced annually in the UK

(NDA and DECC 2014a). Radioactive waste production, handling, packaging and disposal are strictly regulated to protect the industry workforce, the public and the environment. However, the management of radioactive waste remains a contentious issue and is often seen as the Achilles heel of the nuclear industry.

Box 1-1 UK radioactive waste category summary

Exempt Waste – Waste with levels below the activity level subject to regulatory control and can hence be 'free released' to landfill or for recycling. This waste is not included in the research.

LAW - Low Activity Waste this is the generic name for low level (LLW) and very low level radioactive waste (VLLW).

Low Volume VLLW – Very low level waste that can safely be disposed in 'dustbin' sized loads to unspecified landfill sites for municipal, commercial or industrial waste. This waste is not included in the research.

High Volume VLLW – Very low level waste that is classified as 'bulk disposal', usually arising from nuclear facility operation or decommissioning process that must be disposed to designated landfill sites with the required authorisation. The three main designated facilities are Clifton Marsh, Lillyhall and Kingscliffe. Dounreay in Caithness is constructing its own facility for the equivalent waste at the site and the adjacent Ministry of Defence site (Vulcan).

LLW – Low level waste that has a low specified activity but which must be consigned to a fully authorised and dedicated LLW facility. The LLW repository near Drigg in Cumbria is a national facility. Dounreay is also constructing its own LLW facility for the equivalent waste from Dounreay and Vulcan.

HAW – Higher activity waste includes LLW that cannot be disposed to the LLW Repository, ILW, HLW and SNF.

ILW – Intermediate level waste has an activity level above LLW but does not need to take account of heat generation in its handling, storage, transport and disposal.

HLW – High level waste that has an activity level above ILW that does require the heat it generates in handling, storage, transport and disposal is taken into account. There is currently no disposal facility for this category of waste and it is retained in long-term interim storage at individual site and at Sellafield. This waste is not included in the research.

SNF – Spent nuclear fuel represent fuel elements that have been removed from nuclear power stations that have not be reprocessed and hence are disposed whole to a deep geological facility. This waste is not included in the research.

Note: the regulatory limits for each of these categories of waste are presented in Table 2-1.

Currently there are a small number of radioactive waste repositories in the UK for lower activity radioactive waste (LAW) (as defined in Box 1-1). These have limited capacity, currently about ¼ of the estimated total volume of waste arising (NDA and DECC 2011), and limited operational life. There are currently no disposal facilities for higher activity radioactive wastes (HAW), hence long-term interim storage facilities are required at nuclear sites. The volume of legacy radioactive waste requiring disposal is growing as decommissioning programmes progress.

Solutions must be found for the legacy radioactive wastes, future radioactive wastes from new nuclear power stations and materials not currently classed as waste but that may be in the future (e.g. depleted, natural and low enriched uranium (DNLEU)). The treatment and conditioning of radioactive waste to produce a stable waste-form, packaging the waste for secure and safe storage and final disposal have significant environmental and financial costs. Hence, effort is needed to minimise the radioactive waste sent to disposal facilities.

1.1.1 Metals

Metals are a valuable natural resource with ores of varying scarcity in the Earth's crust. For example, metals of interest in this research range in cost from ~£500/tonne for steel to ~£13,000/tonne for nickel at July 2010 prices and ores range in scarcity from 0.002% in the Earth's crust for uranium to 8.0% for aluminium. Metals are used in a multitude of ways to benefit humankind and demand will grow as the world's population continues to increase.

The general metals cycle from mining, through concentration, refining, manufacture and use, to disposal can be represented as a generic metal cycle as discussed in Chapter 5. The metal cycle includes various opportunities to re-use, re-manufacture, re-refine and re-concentrate metals. The extraction, production, use and disposal of metals results in significant impacts on the natural and human environment and recycling offers the chance to reduce these impacts.

An intrinsic property of metals is that they are theoretically infinitely reusable. In practice this is not possible because it may be uneconomic for the energy required or time consuming, impractical and expensive for industry and consumers to segregate metals from other waste materials. Historically buildings, machinery and consumer goods were not designed for materials to be easily recoverable hence they were simply disposed. However, limited metal recycling has been a feature from the beginning of their use. The benefits of recycling in terms of avoided virgin material, reduced spoil from extraction and concentration, water use and pollution, emission to air, energy savings and reductions to health risks for workers, depends on the metal and its processing. For example, typical energy savings for steels range from 70-75% and raw materials savings are about 90%. It is generally true that material and emissions are significantly reduced by metal recycling, however energy reduction is not always clear. The energy balance depends on the metal, the processing plants and the recycling techniques.

1.1.2 Research Focus

Metals constitute about 17% of the LAW and 39% of the HAW hence they offer an opportunity to reduce the waste volume through treatment and recycling (NDA and DECC 2011). The assessment of the environmental impacts of radioactive metals treatment, storage, transport, disposal and recycling to minimise material use and energy and to conserve scarce disposal capacity are the focus of this research.

1.2 Nuclear Fuel Cycle

This section gives a brief overview of the whole life cycle of nuclear fuel from uranium ore extraction to waste disposal.

The nuclear industry is a global business with global benefits and detriments. The industry relies heavily on various metals, but iron and steel in particular, for all its nuclear and non-nuclear plants and equipment. Sherry et al 2010 state that "The UK has had a self-sufficient programme for nuclear power since the 1950s which included the ability to design and build reactors, to manufacture and enrich fuel and to manage the irradiated fuel after discharge from the reactor". This is best interpreted by the nuclear fuel cycle shown in Figure 1-1.

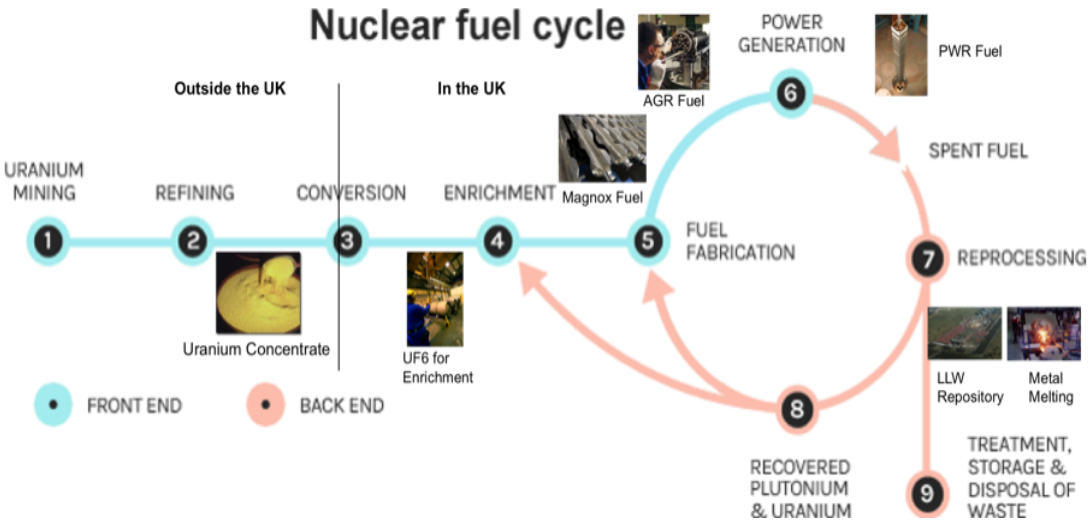

Figure 1-1 Nuclear Fuel Cycle showing the process of interest in the thesis (modified from World Nuclear Transport Institute, NDA, Westinghouse Ltd and Studsvik UK)

The uranium mining (1), refining (2) and the initial conversion (3) are done outside the UK. The UK cycle starts with final conversion (3), enrichment (4) and fuel fabrication (5). Uranium concentrate is converted into uranium metal fuel for Generation I Magnox power stations at the Westinghouse Ltd Springfields facility. Civil and defence uranium is enriched via the Urenco UK Ltd gas centrifuge plants at Capenhurst. The civil enriched uranium is returned to Springfields to produce fuel for the Generation II Advanced Gas-Cooled Reactor (AGR), Generation III Pressurised Water Reactor (PWR) power stations and research reactors. The fuel is used in civil power stations to generate electricity, for nuclear submarine propulsion and for research (6). Magnox, AGR and research reactor fuel elements are made at Springfields but the civil PWR fuel elements are made abroad. Nuclear submarine PWRs are designed and the reactor cores assembled in Derby. The submarines are built and commissioned at Barrow-in-Furness and are refuelled at operational naval bases. Nuclear deterrent warheads are made at the atomic weapons establishments at Burghfield and Aldermaston (6). Stages (1) to (6) are known as the Front End of the nuclear fuel cycle. It should be noted however that UK conversion and fuel production processes have changed, and continue to change, as no uranium conversion to UF_4 or UF_4 to UF_6 is now done in the UK (Butler 2015, pers. comm.).

The Back End of the cycle includes the reprocessing of spent nuclear fuel (7), the recovery of useable plutonium and uranium (8) and the treatment, storage and disposal of radioactive wastes (9). Reprocessing and recovery are done at the nuclear chemical processing facilities at Sellafield. The recovered materials can be re-enriched and recycled into new fuel via stages (4) and (5). Wastes are treated at individual sites or at national and international facilities as part of stage (9) prior to disposal. LAW can be disposed at five licensed facilities but HAW is stored at sites until disposal facilities are available. Treatment, disposal or recycling of radioactive metals in stage (9) of the nuclear fuel cycle is the topic of this research.

The majority of radioactive waste in the UK is from the facilities in the nuclear fuel cycle above. However, small quantities of radioactive waste are also generated by medical, academic and non-nuclear industries.

1.3 Radioactive Waste

This section includes discussion of radioactive contamination, waste categories, waste treatment and conditioning plus waste storage and disposal.

1.3.1 Radioactive Contamination

Metals at nuclear facilities that do not come into contact with, or get contaminated by, radioactivity can be segregated and returned to the UK metal cycle as they are exempt from regulatory control. Metals that are radioactively contaminated or irradiated can be stored, to benefit from radioactive decay, or decontaminated to levels that allow them to be returned to the metal cycle or made into products for beneficial re-use in the nuclear industry. Metals that cannot be decontaminated to such levels must be stored and eventually disposed in licensed waste repositories.

1.3.2 Radioactive Waste Categories

The general classification of LAW and HAW in Section 1.1 is a convenient way to group the UK radioactive wastes. However, wastes are split into specific categories differentiated by their radioactive content and heat generation capacity. The categories are; exempt waste, low level waste (LLW) and its sub-category very low level waste (VLLW), intermediate level waste (ILW) and high level waste (HLW)

are defined in Box 1-1. ILW has a higher activity than LLW but does not generate heat. HLW has the highest activity and generates heat that needs to be taken into account for storage and disposal.

Low activity waste includes LLW, low volume VLLW, high volume VLLW and waste contaminated by naturally occurring radioactive material (NORM), predominately from the UK oil and gas industry. Higher activity waste includes LLW that cannot be disposed at a LLW repository (e.g. radioactive graphite from reactor cores), ILW and HLW. The radioactive waste categories are discussed in more detail in Chapter 2.

1.3.3 Waste Treatment and Conditioning

The characterisation of radioactive wastes by their main radionuclides, activity level and physical or chemical properties allows them to be segregated for treatment, conditioning and storage. Treatment of solid wastes involves dismantling, size reducing and decontaminating of plant and equipment. Chemical decontamination can be done as post operational clean out of a facility prior to dismantling. Physical or chemical decontamination can also be done manually or remotely after removal. Melting metals is a useful supplementary decontamination method for large and complex metal components.

LLW metals can be decontaminated to VLLW or exempt levels. ILW metals can be decontaminated to LLW for further treatment in the UK or abroad. Radioactive metals treatment is discussed in more detail in Chapters 2 and 5.

Conditioning involves any other treatment that is needed to produce a passively safe waste-form for storage and/or disposal. This is commonly achieved by encapsulating the waste to produce a solid waste-form.

1.3.4 Waste Storage and Disposal

Waste storage can either be in-situ or containerised. In-situ storage results in the treatment, conditioning and packaging of waste late in the decommissioning process.

Waste produced during operation, or early in decommissioning, is normally retained in steel containers (as listed in Box 1-2) but concrete containers can also be used. This waste can either be conditioned immediately or left unconditioned for storage pending disposal. More detailed waste container physical and cost data are given in Box A-1 and Sections A.1 and A.2.

Box 1-2 UK radioactive waste containers discussed in the thesis

LAW Containers

High Volume VLLW: 200 – 210 litre drum, carbon steel, internal volume ~0.2m3, external volume ~0.25m^3, empty mass 300kg

LLW: Full height ISO (FHISO)[1] freight container: carbon steel, internal volume 31.4m^3, external volume 38.3m^3, empty mass 4800kg

Half Height ISO (HHISO) container: carbon steel, internal volume 17.9 m^3 (15.5 m^3 useable), external volume 19.5m^3, empty 3080kg

ILW – Unshielded Containers

50litre keg: stainless steel, internal volume 0.05m^3, external volume 0.07m^3, empty mass 8kg

200 litre drum: stainless steel drum, internal volume ~0.2m^3, external volume ~0.25m^3, empty mass 300kg

500 litre drum: stainless steel drum, internal volume ~0.5m^3, external volume ~0.6m^3, empty mass 120kg

3m^3 drum: stainless steel, internal volume 2.61m^3, external volume 2.85m^3, empty mass 634kg

ILW – Shielded containers

4m box[2]: stainless steel, internal volume 16.49m^3, external

2m box[3]: stainless steel, internal volume 8.17m^3, external volume 10.56m^3, empty metal mass 3000kg

Ductile cast iron containers (DCICs)[4]

Notes:

1 - There are two main variants of the FHISO containers with different masses. The data for these are presented in Box A-1. The FHISO for depleted uranium discussed in Chapter 9 is a special variant of the designed to transport the 50 litre kegs from Sellafield to a disposal facility or for re-conversion.

2 - There are 4 variants of the 4m stainless steel box; one with no internal shielding and with 100mm, 200mm and 300mm thick internal shielding. The data for these are presented in Box A-1. The internal volume changes but the external volume and the empty metal mass remain the same.

3 - There are 2 variants of the 2m stainless steel box; one with no internal shielding and one with 100mm thick internal shielding. The data for these are presented in Box A-1. The internal volume changes but the external volume and the empty metal mass remain the same.

4 – The DCICs are discussed briefly in Section 2.5.4 and shown in Figure 2-20. They have cast iron masses of between 6000kg and 18000kg, are only used for specified ILW and of no interest in metals disposal (unless high irradiated). Hence, they are not included in the research.

Containerised waste is then transported off-site for disposal or storage at another site prior to final disposal (e.g. Magnox legacy depleted uranium to Capenhurst). Waste

is normally cement-grouted into the containers to give a stable waste-form and produce a load bearing waste package (i.e. the waste-form plus container) for disposal.

The main UK LLW disposal facility is the repository near Drigg in Cumbria. The LLW Repository is classed as a scarce natural resource. The current LLW policy (DEFRA et al 2007), strategy (NDA 2010a), programmes and plans were developed to conserve this capacity by minimising the volume of LLW for disposal. A new LAW disposal facility is being constructed at Dounreay in Scotland (see map in Figure 2-7). However, it is only licensed and authorised to dispose of wastes from Dounreay and the adjoining naval propulsion test facility and is therefore not considered in this research.

Low volume VLLW, predominately from the non-nuclear industries, can be disposed to conventional landfill sites. High volume VLLW can be disposed to three licensed commercial facilities in England. VLLW may also be disposed to designated areas at current nuclear sites if licensed and authorised by the regulators.

There are currently no HAW disposal facilities in the UK. The Committee on Radioactive Waste Management (CoRWM) made several recommendations for managing radioactive waste safely. CoRWM Recommendation 1 was deep geological disposal of HAW (CoRWM 2006). The recommendation was accepted for England and Wales and the location of a future geological disposal facility (GDF) is being explored. The Scottish Government rejected geological disposal in favour of the long-term management of HAW in a near site near-surface facility or facilities. Both disposition options are addressed in the current HAW policy (initially DEFRA 2008 but updated in DECC 2014) and developing strategy and plans (e.g. NDA 2009a and 2012a). CoRWM realised that developing these disposal facilities would take a considerable time hence their Recommendation 2 advocates long-term interim storage at sites until disposal facilities are available (CoRWM 2006). This requirement is reflected in the current HAW policy, strategy and plans, with long-

term interim storage up to 100 years to allow for contingencies and uncertainties in the availability of disposal facilities (e.g. DECC 2014).

LAW and HAW disposal is discussed in more detail in Chapter 2 hence attention is now placed on outlining the analysis of disposal and recycling within this research.

1.4 Environmental and Financial Cost Analysis

This section discussed the environmental issue of the UK nuclear industry, life cycle assessment and costs analysis.

1.4.1 Environmental issues in the UK Nuclear Industry

A raft of environmental, health and safety legal requirements and guidance underpin the LAW and HAW policies, strategy, plans and programmes. These include environmental management system requirements from the early 1990s, environmental impact assessment of individual projects and strategic environmental assessment (SEA) and sustainability reports (e.g. NDA and Entec 2009). Also important are best practicable environmental option (BPEO) and best available techniques (BAT) requirements of UK regulations from the mid 1990s (e.g. the LLW metals strategic BPEO reported in Studsvik 2006a and 2006b).

Environmental impacts are important attributes/criteria of decision making for UK radioactive waste management (e.g. Studsvik 2006a and 2006b). UK radioactive waste management projects must consider BAT in England and Wales but BPEO and best practicable means in Scotland and Northern Ireland. Decisions also require consideration of as low as reasonably achievable studies for radiation dose to nuclear industry workers and the public. These requirements help structure the assessment of radioactive waste treatment and disposal options and associated impacts.

These legal requirements, project business cases and stakeholder engagement are the foundations of the current UK policies and strategies for solid LLW and HAW. The solid LLW strategy is underpinned by the SEA study (NDA and Entec 2009) and brings together the triple bottom line of environment, social and economic

considerations of the strategy. Similar considerations are also included in the developing framework for the disposal of HAW.

1.4.2 Life Cycle Assessment

Life cycle assessment (LCA) is one of the many tools in environmental management decision-making. LCA is defined as "an objective process to evaluate the environmental burdens associated with a product, process or activity by identifying and quantifying energy and materials used and wastes released to the environment and to evaluate and implement opportunities to affect environment improvements", SETAC 1993. It is a tried and tested methodology prescribed in the International Standards Organisation (ISO) standard 14040 series. LCA has an overall global, continental or regional perspective leaving local assessment to environmental management audits and environmental impact assessments. LCA therefore appears ideally suited to assess the potential environmental impacts of the nuclear fuel cycle presented in Figure 1-1 and the general metals cycle. LCA can either address the whole of these cycles or be applied to discrete stages of the life cycle but still using the holistic thinking to address key aspects.

There are some shortcomings of BPEO in assessing process selection, design and optimisation and Azapagic 1999 advocates the use of LCA to give holistic cover of environmental issues in such situations. The details of the LCA software used and the life cycle impact analysis (LCIA) method adopted for the research are discussed in detail in Chapter 3. The use of LCA in assessing the potential environmental impacts of the international nuclear industry and radioactive waste and decommissioning in particular are discussed in detail in Chapter 4.

1.4.3 Cost Analysis

The UK solid LLW strategy, supporting SEA, BPEO and BAT studies recognise the need to include economic consideration in decisions for radioactive waste management. Therefore cost analysis goes hand in hand with environmental analysis.

It is not within the scope of the research to develop a full cost benefit analysis, or life cycle cost analysis, of disposal and recycling of radioactive metals. Rather, the

research uses costs derived by others taken from the literature to supplement the LCA results to compare the scenarios investigated.

Treatment, packaging, transport and disposal of waste accrue significant financial costs as well as environmental costs. These costs can be offset for metals by treatment for recycling or re-use. This would reduce disposal costs and generate revenue from the sale of the metals. Alternatively, the use of recycled metals could offset the costs of procuring virgin metals for future application in the nuclear industry. The background to the cost analysis method is discussed in more detail in Chapter 6 and additional data are given in Sections A.2 and A.3.

1.5 Research Aim and Research Objectives

Having summarised the UK nuclear fuel cycle, radioactive waste and life cycle assessment and costs analysis for disposal and recycling of radioactive metals attention can now be given to the design of the research project.

Aim

The research aim was how best to maximise the environmental benefits, or minimise the environmental detriments, of radioactive waste management decision-making.

The research objectives were:

a) to create a small number of representative models to determine the potential environmental impacts of interesting scenarios for treating, packaging, disposing and/or recycling of legacy radioactive metals and the long term interim storage and disposal of legacy UK civil depleted uranium by addressing the spectrum of materials and energy conservation and waste minimisation in a holistic way using Life Cycle Assessment (LCA), and

b) to develop indicative financial costs for each scenario.

The aims and objectives will be achieved by answering the following general questions:

1. What level of potential environmental benefits does recycling radioactive metals bring about within the current UK radioactive waste management policies, strategies and plans?
2. What are the indicative financial costs of disposal and recycling and how do we balance these with environmental benefits?
3. What other considerations could be significant and hence need to be included in the assessment of management options for radioactive waste including metals?

The research is intended to support, or refute, practices underpinning existing radioactive waste management policies and inform future decisions in areas of perceived gaps and emerging concerns.

1.6 Thesis Structure

The thesis is developed over 10 chapters including this introduction, which outlines the research. Chapter 2 discusses the problem situation, i.e. that the estimates of radioactive waste volume from UK nuclear facilities are increasing, legacy low activity waste disposal facilities are limited and their capacity is scarce and there are currently no disposal facilities for legacy higher activity waste. The chapter discusses what radioactive waste is, how and where it's produced, how much waste there is and what solutions have been developed to address the problem by the UK government, the nuclear operators and nuclear industry supply chain. The research is primarily interested in assessing the potential environmental effects of the disposal and treatment of radioactive waste hence Chapter 3 presents a brief background to the LCA methodology. The chapter discusses the history of the methodology the process stages to generate a life cycle assessment and highlights some limitations of the methodology.

Chapter 4 presents a review of a selection of international LCAs for the nuclear fuel cycle as a whole, and UK LCAs for waste and decommissioning in particular. This is to identify gaps in current knowledge that are explored in the research. The significance of metals and their environmental impacts in general are discussed in Chapter 5. The chapter discusses why metals are important, what significant

environment impacts are associated with their production and how metal demands are changing. This allows identification of why recycling metals is important and how metals production and recycling are being improved.

Chapter 6 describes the research methodology that includes a combination of life cycle assessment using propriety software and published nuclear industry data and indicative cost estimates for each research scenario using public domain data. Chapters 7, 8 and 9 present the results of the analyses linking them to the research aims and objectives. Chapter 7 is a hypothetical case study to develop and test the research methodology. Chapter 8 is the application of the methodology to the entire UK radioactive metals inventory and Chapter 9 presents the results a potential significant future problem, i.e. the disposal of depleted uranium if classified as waste. Chapter 10 reviews the research summarising the conclusion and identifying some limitations of the approach. It brings the thesis to a close by identifying areas for further research and makes recommendations to the industry.

This chapter outlined the current position of UK nuclear industry, presented the focus and scope of the research and the structure of the thesis. Attention now turns to discussing the radioactive waste in the UK in more detail.

2 Overview of UK Radioactive Waste

Chapter 1 outlined the research context, aims and focus. It is now important to understand the key issues of UK radioactive wastes and their management. As stated previously, the UK has a fully developed nuclear fuel cycle and related activities. This can be summarised by six business areas (NDA and DECC 2014a):

- Fuel fabrication and uranium enrichment (nuclear fuel cycle stages 3, 4 and 5)
- Nuclear power reactors (for civil power generation (stage 6)),
- Defence (for propulsion and weapon systems (stage 6)),
- Spent fuel reprocessing for platinum and uranium recovery (stages 7 and 8)
- Nuclear energy R&D (covering all stages), and
- Medical and industry (outside the main nuclear fuel cycle)

Radioactive waste treatment, storage and disposal (stage 9) is not separately identified as it applies across all the business areas.

Radioactive waste arises during the operation and maintenance of all the nuclear facilities and particularly in their decommissioning and dismantling. Although the radioactive waste volumes are very small compared to conventional waste as shown in Chapter 1 it is highly regulated to protect the public, industry employees and the environment.

Radioactive waste management can be conveniently discussed using the following headings from NDA and DECC 2014a which are used to structure this chapter:

What is radioactive waste? (2.1)

How is radioactive waste produced? (2.2)

Where is radioactive waste produced? (2.3)

How much radioactive waste is there? (2.4)

How is radioactive waste dealt with? (2.5)

What is the long-term management solution? (2.6)

2.1 What is radioactive waste?

This section discussed lower activity wastes and higher activity waste and looks at the subcategories under these general headings.

"Radioactive waste is any material that is either radioactive itself or is contaminated by radioactivity, for which no further use is envisaged" (HSE et al. 2007). Most nuclear facilities or operations are split into two distinct areas, a radioactive designated area (the primary side) and a non-radioactive area (the secondary side). The primary side generates both radioactive and non-radioactive waste. The secondary side generates primarily non-radioactive waste but may contain very small quantities of radioactive waste. The different radioactive waste categories and specifications currently used in the UK are presented in Table 2-1. This section will now look at these wastes

2.1.1 Lower Activity Wastes (LAW)

Exempt Waste

Waste from the secondary side of nuclear facilities, or wastes from the primary side that can be decontaminated to below radiological regulatory control limits, are known as exempt wastes. Exempt wastes can be disposed to normal landfill or recycled. Exempt metals from the secondary side of nuclear facilities are excluded from the research. Radioactive metals from the primary side that can be decontaminated to exempt levels are included in this research as they can generate revenue that off-sets the cost of radioactive waste management.

Very Low Level Waste (VLLW)

The current solid low level radioactive waste (LLW) policy (DEFRA et al. 2007) formalised VLLW as a sub-category of LLW. The new policy addressed deficiencies in the previous policy (HMG 1995) recognising the high volume of waste generated during decommissioning of nuclear facilities. There is a very small quantity of low volume VLLW metals hence they are excluded from the research. There is a significant quantity of high volume VLLW metals hence they are included in the research.

Low Level Waste (LLW)

LLW from operational and decommission activities generally does not need shielding and can be manually handled because of its very low radioactive content. The low radioactivity presents limited harm to those handling it but is none-the-less strictly regulated and monitored. LLW dominates the waste volume at 93.6% but constitutes only 0.00005% of the net radioactivity (NDA and DECC 2014a and 2014b). It contains mainly short-lived radionuclides that decay to safe levels in several years or a few decades. LLW metals treatment and recycling are central to the research.

Table 2-1 UK Radioactive Waste Categories and Specifications

Waste Category	Waste Specification
Exempt Waste	"The lower activity limit for LLW, below which waste is not required to be subject to specific regulatory control, is covered by exemption orders under Radioactive Substances Act 1993." "The most notable of these is the Substances of Low Activity (SoLA) Exemption Order. This specified a level of exemption from regulatory control of 0.4 Becquerel (Bq/g) for wastes that are substantially insoluble in water" (HSE et al. 2007)
Low Volume Very Low Level Waste	"…('dustbin loads') -wastes that can be safely disposed of to an unspecified destination with municipal, commercial or industrial waste, each 0.1 cubic metre of material containing less than 400kBq (kilobecquerels) of total activity, or single items containing less than 40KBq of total activity. There are additional limits for C14 and tritium in wastes containing these radionuclides" (HSE et al. 2007)
HighVolumeVery Low Level Waste	"… (bulk disposals) – wastes with a maximum concentration of 4MBq (megabecquerels) per tonne of total activity that can be disposed of to specified landfill sites. There is an additional limit for tritium in wastes containing this radionuclide." "The Environment Agency has issued permits to the operators of certain landfill sites for the disposal of LLW with an activity of 200MBq per tonne." (HSE et al. 2007)
Low Level Waste	"Radioactive waste having a radioactive content not exceeding four gigabecquerels per tonne (GBq/te) of alpha or 12 GBq/te of beta/gamma activity" (DEFRA et al. 2007, HSE et al. 2007, NDA and DECC 2014c)
Intermediate Level Waste	"Waste exceeding the upper boundaries of LLW, but which do not need heat to be taken into account in the design of storage and disposal facilities" (NDA and DECC 2014c and HSE et al. 2007). "IAEA guidance is that ILW thermal power is below about 2 kW/m^3" (IAEA 2003)
High Level Waste	"Waste in which the temperature may rise significantly as a result of their radioactivity, so this factor has to be taken into account in the design of storage and disposal facilities" (NDA and DECC 2014c and HSE et al. 2007). "IAEA guidance is that HLW thermal power exceeds about 2 kW/m^3" (IAEA 2003)

2.1.2 Higher Activity Wastes (LAW)

LLW not acceptable to a LLW Repository

About 17,000m^3 of LLW, mostly radioactive graphite, cannot be accepted at the LLW repository near Drigg or the new LLW facility at Dounreay (NDA and DECC 2011) because of their long-lived radionuclide content. There are currently no metals in this category.

Naturally Occurring Radioactive Material (NORM)

Materials and equipment contaminated by NORM, originate predominately from the UK oil and gas industry. They are currently chemically treated off-shore and radioactive liquid waste is discharged to the North Sea. However, some NORM contaminated metals may be generated on-shore and must be disposed or recycled. The radium-226 and 228, polonium-210 and lead-210 remaining in the waste are long lived and only accepted in limited quantities at the LLW Repository (Drigg). UK Nirex Ltd estimated 65,000t/year of NORM in 2003 but decreasing as the oil and gas industry decline over the next 20 years. The 2003 Nirex estimate suggested that there was sufficient UK NORM contaminated metals to warrant inclusion in this research. Further investigation in SNIFFER 2005 and Wareing 2008 estimated only ~850m^3 of NORM from 2004 to 2008 and falling to ~90m^3 by 2040. NORM contaminated metals are therefore excluded from the research.

However, should regulations change, and more NORM contaminated plant and pipework be decontaminated on-shore, it may have to be categorised as HAW for GDF disposal or to long-term management in Scotland.

Intermediate Level Waste (ILW)

ILW arises during operational and decommission activities. It generally requires shielding, remote handling and strict regulation and monitoring to protect the workforce. ILW constitutes 6.4% of the net waste volume and 5% of the net radioactivity (NDA and DECC 2014b). There is a significant quantity of ILW metals but far less than LLW metals. ILW metals contain a mixture of short-lived and long-

lived radionuclides, the latter taking hundreds or thousands of years to decay. ILW metals that can decay or be decontaminated to LLW by circa 2110 are included here.

High Level Waste (HLW)

Very small quantities of HLW arise from reprocessing spent nuclear fuel. It is mainly high activity liquor that generates heat from fission product and trans-uranic nuclides. It requires significant shielding and remote handling. HLW constitutes only 0.02% of the net waste volume but 95% of the net radioactivity (NDA and DECC 2014b). HLW metals are excluded from the research because of their very small quantity and high radio-toxicity.

2.2 How is radioactive waste produced?

This section discussed radioactive contamination, decommissioning facilities, waste treatment plus waste storage and disposal.

Radioactive wastes are generated by the contamination, or irradiation in a strong radiation flux, of materials in the primary side of a nuclear facility or operation. Leaks or releases from the primary side can cross-contaminate materials on the secondary side of a facility or operation, resulting in small quantities for radioactive waste.

2.2.1 Radioactive Contamination

Radioactive contamination of some plant and equipment in the business areas mentioned above generates radioactive waste. The level of contamination varies with the processes undertaken and the materials involved. There is predominately high volume VLLW and LLW in conversion, enrichment and fuel fabrication but only very small amounts of ILW. There is considerably more LLW, ILW and HLW from electricity generation, nuclear propulsion, reprocessing and plutonium and uranium recovery. These wastes must be treated, stored, disposed or recycled according to national policies, strategies, plans and regulations.

Metal vessels, pipes and/or ducts, support structures and equipment exposed to radioactive liquids, gases or a high radiation flux can be contaminated on internal

and/or external surfaces or activated. Surface contamination can be loose or fixed. Loose surface contamination is relatively easy to remove by basic physical means such as manually wiping or abrading, or chemically with mild reagents. This normally requires minimum protective clothing and can be done in designated areas within the facility. Fixed surface contamination needs more aggressive decontamination techniques such as shot blasting, high pressure water jetting or pickling with suitable acids etc. These generally require enclosed facilities and operators may need to wear specialised clothing if the process is not remotely controlled.

Activation can either result in volumetric contamination of the metals or be contained in a tenacious oxide film attached to the metals. Volumetric contamination of the metal generally arises from the bombardment in a neutron flux generated by an operating nuclear reactor. Oxide films containing activation products and or fission products can be strongly bonded to inner or outer surfaces of plant and equipment. In both cases the radiation levels when handling and treating these metals can be relatively high and warrant careful management to protect the operators.

As an example of possible radioactive contamination consider the two schematics in Figure 2-1 and Figure 2-2. They show the general layout and basic generation cycle for a Generation II Advanced Gas-cooled Reactor (AGR) nuclear power station. Everything within the reactor pressure vessels (e.g. the reactor core, boilers, gas circulator impellers, steel work etc) will be highly contaminated and/or activated.

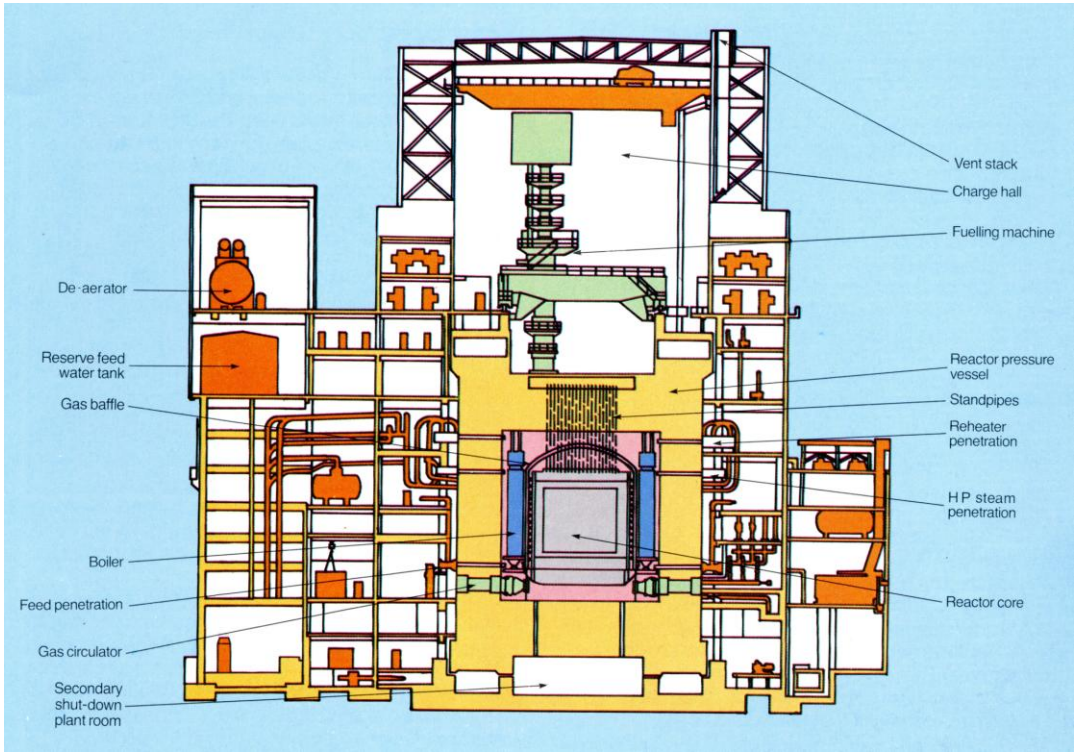

Figure 2-1 Schematic of general layout of Torness power station (from the South of Scotland Electricity Board, circa 1980, Torness Power Station)

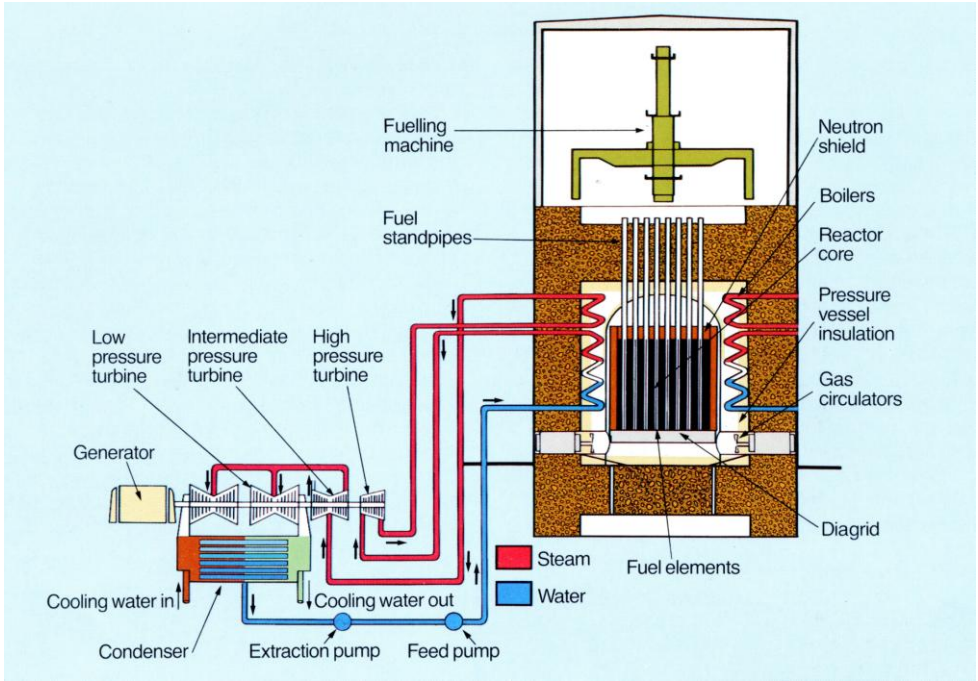

Figure 2-2 Schematic of the basic AGR generation cycle for Torness power station (from the South of Scotland Electricity Board, circa 1980, Torness Power Station)

Any leakage between the reactor circuit and secondary circuits (e.g. gas feed, bypass and blow-down systems, fuelling machine, the steam side of the turbo-generation

system and gas monitoring) could result in radioactive contamination of surfaces. Generation I Magnox nuclear power stations, Generation III civil Pressurised Water Reactor (PWR) nuclear power stations, naval PWRs and research reactor facilities have similar sources of radioactive waste.

Radioactive contamination is also possible between primary and secondary systems in fuel production, enrichment and fabrication, spent fuel reprocessing and recovery, and in the atomic weapons establishments of the Ministry of Defence. Plant and equipment within each stage of the nuclear fuel cycle contain varying types and significant amounts of radioactive metals.

2.2.2 Decommissioning Facilities

Decommissioning nuclear facilities such as the Magnox power station at Berkeley shown in Figure 2-3 and Figure 2-4 requires the management of significant amounts of VLLW and LLW metals. This is also true for AGR and PWR power stations and research reactors at Winfrith, Harwell, Dounreay and Sellafield. It is also true for the 16 nuclear-powered submarines that have left naval service, 11 of which have been defueled. The submarines are being stored afloat at Rosyth and Devonport dockyards (NDA and DECC 2011).

As a result of the 2007 LLW policy change radioactive metals treatment and recycling is now an established alternative to LLW disposal. Figure 2-3 and Figure 2-4 show the progress made at Berkeley power station. Figure 2-3 shows the power station when it was operational. The reactors are housed in two central buildings connected to eight boilers in external buildings by large gas ducts at the top and bottom. The upper ducts are clearly shown in the figure.

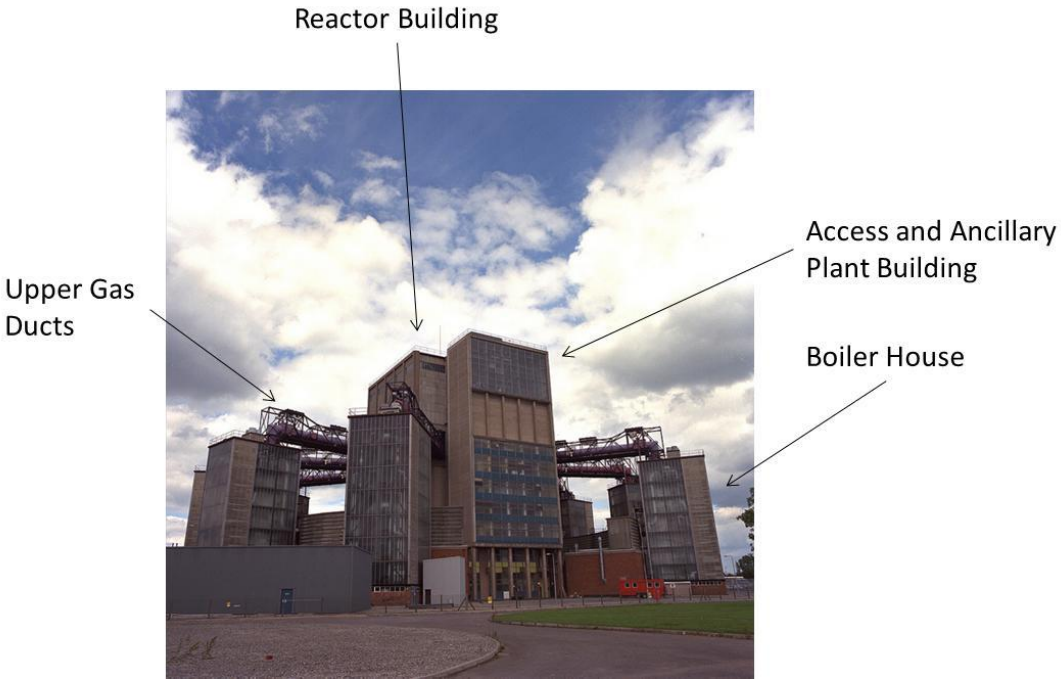

Figure 2-3 Berkeley Magnox power station during operation (modified form an image courtesy of Magnox Sites)

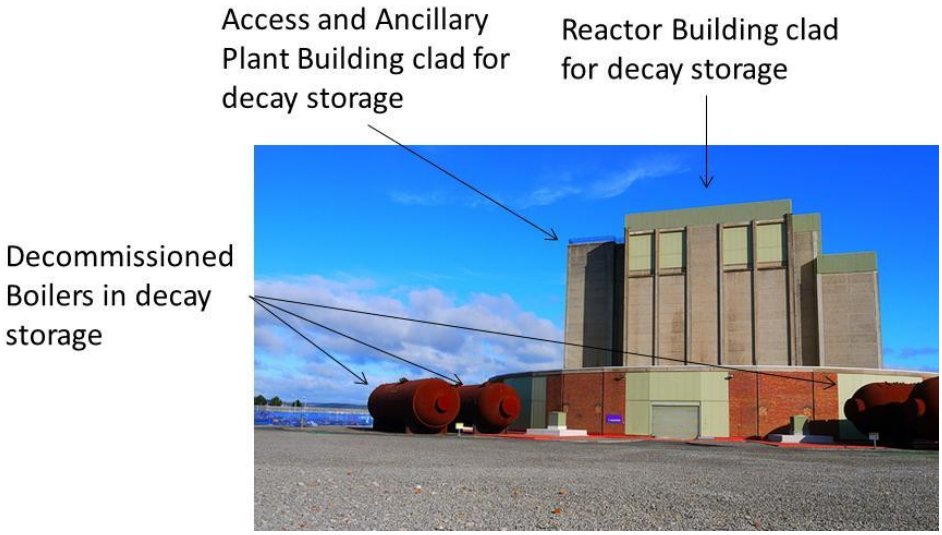

Figure 2-4 Berkeley Magnox power station during decommissioning before the boilers (large cylindrical vessels) were removed for treatment and recycling at Studsvik's Nykoping facility in Sweden (modified from an image courtesy of Magnox Sites)

Figure 2-4 shows a similar view of the power station during decommissioning. The gas ducts have been removed, the boiler building dismantled and the large boilers (310te each) laid on their side for radioactive decay. The first boiler was dismantled,

decontaminated and the steel sold for scrap in the early 1990s (British Nuclear Group 2005). The remaining 15, however, were transported to the Studsvik AB Nykoping facility in Sweden for treatment and recycling in 2011 and 2012.

2.2.3 Waste Treatment, Storage and Disposal

Radioactive waste is treated for storage and disposal at sites and at specified locations in the UK (LLWR and NDA 2009a). The treatment facilities for LLW metals exist in the UK and abroad (LLWR and NDA 2009a). The national LLW disposal repository at Drigg is shown in Figure 2-5 and Figure 2-6.

Figure 2-5 Low Level Waste Repository near Drigg in Cumbria showing the current disposal vaults and grouted ISO freight waste containers in the foreground (reproduced from NDA website)

Figure 2-6 Third Height ISO freight containers being stacked at the LLW Repository near Drigg in Cumbria (reproduced from NDA website)

High volume VLLW facilities already exist at Lillyhall, Clifton Marsh, Kingscliffe, Dounreay, Sellafield and at some nuclear power stations and research reactor sites. As mentioned previously, ILW and HLW disposal facilities do not yet exist. Hence, these wastes are retained in long-term interim stores, or in-situ, at sites pending final disposal.

2.3 Where is radioactive waste produced?

This section discusses the location of the wastes and how the waste is split across the UK nuclear industry business.

2.3.1 Location

The locations of the UK nuclear facilities are shown by industry type in Figure 2-7. The figure shows that nuclear facilities are generally located in rural areas hence well away from major centres of population. They are also predominately located on the coast or adjacent to lakes or major rivers to provide cooling water for the nuclear power stations and process facilities. The distance between operating and decommissioning facilities and treatment or disposal facilities can be substantial. Hence, transport logistics for new fuel, spent fuel and radioactive waste for treatment, disposal or recycling are potentially significant.

There are 35 major nuclear sites plus hospital, university and non-nuclear industry facilities. About 91% of the radioactive waste is produced in England, 6% in Scotland and 3% in Wales (NDA and DECC 2014a).

Figure 2-7 Location of UK nuclear facilities (reproduced from NDA and DECC 2014a)

2.3.2 Waste by Business Area

Figure 2-8 shows the total volume of legacy waste to circa 2110 estimated in the 2013 UK radioactive waste inventory (UKRWI) (NDA and DECC 2014a) by business area. About 72% of the waste volume is from spent fuel processing activities at Sellafield. This includes wastes from commercial fuel reprocessing and legacy defence industry reprocessing. The next important source of waste is from operational nuclear power stations and those undergoing decommissioning. This accounts for ~16% of the waste volume. Industry research activities plus enrichment and fuel production are small at between 4 -5% each. Medical and industrial plus defence wastes are very small at 1% each. Therefore ~90% of the radioactive waste

arises from two main business areas, spent fuel reprocessing and nuclear power reactors, both of which have large quantities of radioactive metals that could be treated to conserve scarce waste disposal volume and minimise waste containers.

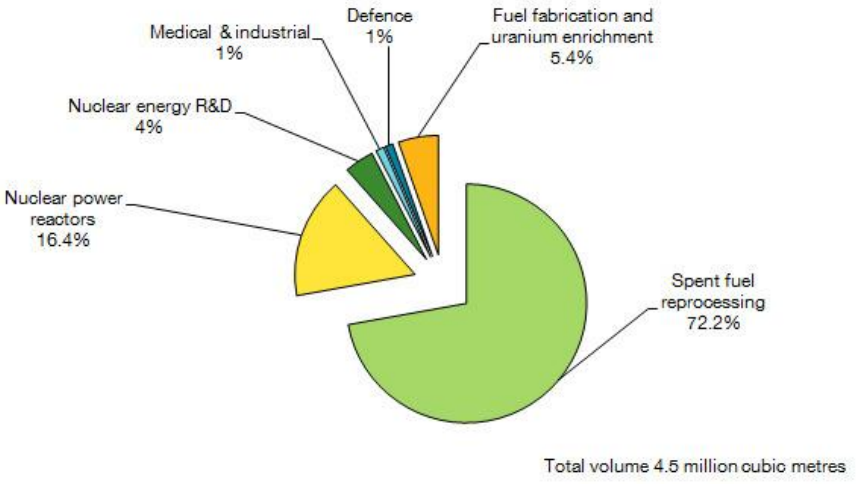

Figure 2-8 Breakdown of legacy radioactive waste by business area (reproduced from NDA and DECC 2014a)

2.4 How much radioactive waste is there?

This section shows the breakdown of waste by waste type, trends in the estimation of wastes in general and waste metals in particular.

The current breakdown of the total legacy waste volume is shown in Figure 2-9.

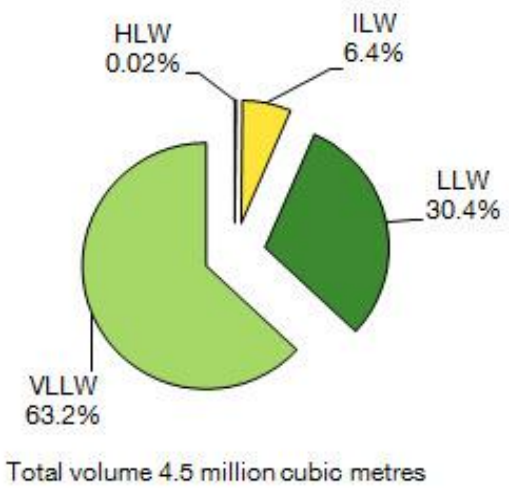

Figure 2-9 Breakdown of legacy radioactive waste by category (reproduced from NDA and DECC 2014a)

The waste volume is dominated by LLW and VLLW, together making about 94% of the total waste volume. VLLW is estimated to be about twice the volume of LLW. ILW makes up ~6% and HLW very much less than 1% (NDA and DECC 2014a) as discussed earlier.

2.4.1 Trends in waste volume over two decades

The trend in UKRWI from data published, nominally every three years, from 1991 to 2013 is shown in Figure 2-10. The figure shows the increase in the estimates of the total radioactive waste volume due to arise from legacy facilities as characterisation and calculations improve. It also demonstrates that the waste volume is dominated by LLW (high volume VLLW as a sub-set is included in the LLW values).

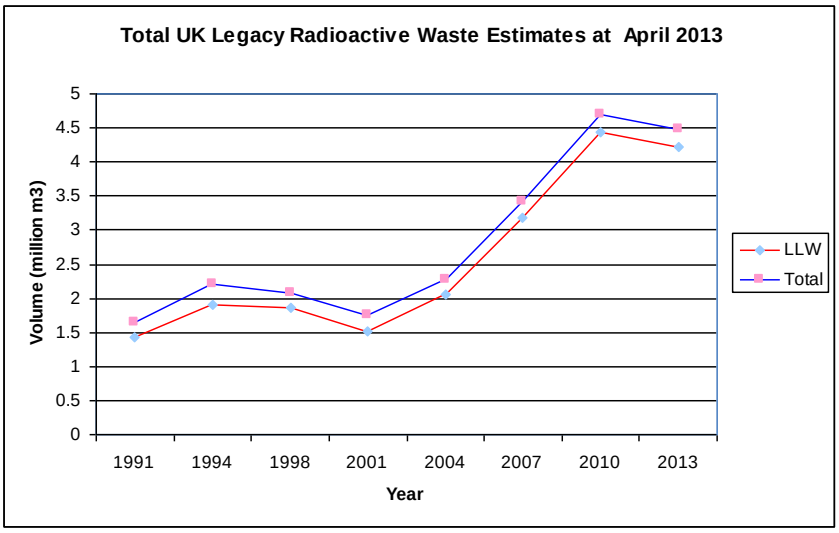

Figure 2-10 Trend in LLW and Total legacy waste volume from successive UKRWIs

2.4.2 Trends in metallic waste masses over two decades

This research is primarily interested in radioactive metals. Figure 2-11 shows the variation in estimates of LLW metal mass (including high volume VLLW) and ILW metals mass from legacy facilities made in UKRWIs from 1994 to the present.

The figure shows that the estimate of total ILW metals is relatively constant over the time period but the total LLW metal estimates, including high volume VLLW metals, fluctuate considerably. This is due to changes in assumptions and calculation methods. It is possible that these estimates may now be stabilising as assumptions

and methods are being standardised. However, the estimates may fluctuate further as better information on the actual wastes becomes available during the decommissioning process.

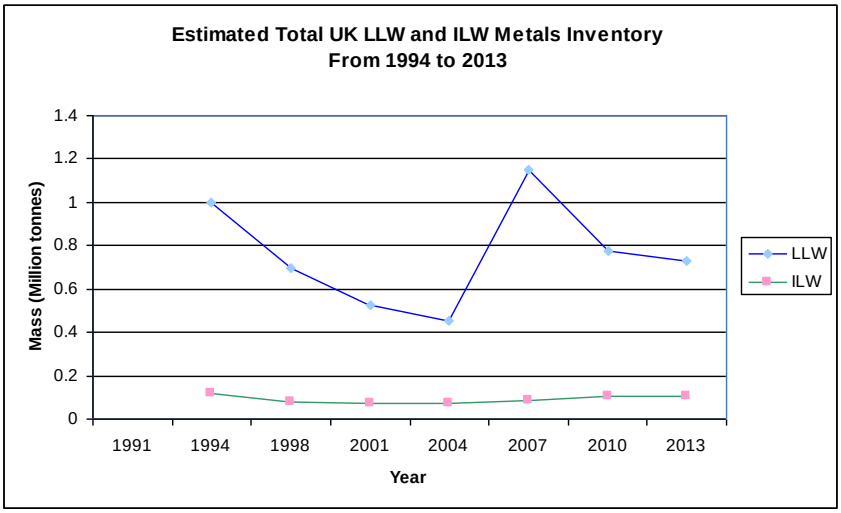

Figure 2-11 Trend in LLW and ILW total legacy metal waste masses from successive UKRWIs

The current estimates are a total of ~141,000tonne (te) of VLLW metals, ~586,000tonne of LLW metals and ~102,000tonne of ILW metals (NDA and DECC 2014d). These constitute 5%, 34% and 38% of the total VLLW, LLW and ILW inventories respectively (NDA and DECC 2014d). These are not insignificant quantities of metals, and as shown later they arise at fluctuating periods during the operation and decommissioning of nuclear facilities. The data for the waste volume and metal waste mass trends is contained in the compact disc provided in Appendix D and notes in 'General Metals' in Table D-2.

2.5 How is radioactive waste dealt with?

This section briefly outlines the control of wastes at nuclear site and radioactive metals treatment processes. The section also discusses conserving scarce disposal volume and issues related to higher activity waste containers and interim storage, pending final disposal to an approved facility.

2.5.1 Control of wastes at nuclear sites

The accumulation of radioactive waste at nuclear sites is discouraged under the nuclear site license issued by the Office of Nuclear Regulation to the operator under the Nuclear Installations Act 1965. The release of liquid, gaseous and solid radioactive waste to the environment is highly regulated by the Environment Agency under Environmental Protection Regulations 2010 in England and Wales and the Scottish Environmental Protection Agency under the Radioactive Substance Act 1995 in Scotland. Waste producers are actively encouraged to minimise the disposal of radioactive wastes by seeking alternatives such as decontamination, super-compaction and incineration of solid wastes, and evaporation and filtration of liquid wastes.

2.5.2 Radioactive metals treatment

Of particular interest in this research are the treatment and/or melting of LLW and VLLW metals. This can be for free release into UK and international metal markets, for beneficial re-use as shielding blocks or containers within the nuclear industry, or for waste volume reduction only in the case of activated metals. Also of interest is the decontamination or decay of ILW metals to LLW levels acceptable for treatment by melting (i.e. the used fuel skips for Magnox nuclear power stations such as Hinkley Point A, Sizewell A and Bradwell).

Decontamination of Wastes

Decontamination of surface contaminated metallic wastes (Figure 2-12 and Figure 2-13) is undertaken at individual sites or at specified facilities in the UK.

Figure 2-12 Size reduction of metals for decontamination and melting (reproduced from Studsvik AB website)

For example the Studsvik metal recycling facility at Lillyhall, Inutec at Winfrith and National Nuclear Laboratory at Spingfields (LLWR and NDA 2009a, LLWR 2012a, NDA and Entec 2009). LLW metals can also be further decontaminated by melting in France, Germany, Sweden (Figure 2-14) and the USA. UK and International treatment facility capabilities are summaried in Section A.3.1. The UK and international facilities are available to UK nuclear operators via the LLW Repository Ltd national waste framework agreement. The LAW Framework Agreement was developed by the LLW Repository Ltd to meet the requirements of the solid LLW strategy and developing plans and programmes. It offers an integrated LAW management service for LAW waste producers for the characterisation, packaging and transport of LLW and VLLW disposal in the UK, metal waste treatment, incineration and supercompaction for major projects using agreed supply chain facilities. It offers a flexible, efficient process and competitive prices for the waste management services between producers and suppliers. It is an integral part of the current structured process for LAW in Figure 2-23.

Figure 2-13 Demonstration of cleaning a 200litre drum in a decontamination cabinet (reproduced from NDA website)

Figure 2-14 Metal melting at an international facility for recycling into metal markets or beneficial reuse in nuclear industry (reproduced from Studsvik website)

Compaction of Wastes

Low force compaction facilities are available at some sites. High force compaction facilities are available at Sellafield, Dounreay and Winfrith (e.g. Lee 2010 p25), via the framework agreement by LLW Repository Ltd. The super-compaction is done in accordance with LLWR 2012b. The compaction process is suitable for soft or thin VLLW and LLW metals or drummed LLW.

Combustion, Evaporation and Filtration of Wastes

Incineration of combustible waste is also available under the LLW Repository Ltd framework agreement and in accordance with LLWR 2012c. Incineration is not of interest in the research as there is no energy recovery from the incineration of VLLW and LLW metals. Likewise, evaporation and filtration are only applicable to liquid wastes and hence are of no interest here.

2.5.3 Conserving disposal capacity

LLW Disposal

The treatment processes help preserve the scarce disposal volume at the LLW Repository near Drigg. Dounreay has its own metal treatment facility and is installing a super-compaction facility to minimise its disposal volume to the new VLLW/LLW disposal facility currently under construction. Solid wastes that cannot be processed through the above treatment processes are disposed to Drigg and Dounreay. LLW, including metals that can be decontaminated to VLLW can be disposed to licensed facilities at Clifton Marsh, Lillyhall and Kingscliffe.

The typical LLW disposal scenario is the transport of waste in commercial 'single use' carbon steel ISO freight containers to Drigg, where it is grouted and capped with cement, and emplaced in the repository vaults. This process is similar for Dounreay. Non-metallic containers and re-useable ISO transport containers are being developed for VLLW disposal.

The progress with the metallic, super-compaction and incineration of LLW, plus the diversion to high volume VLLW disposal after re-characterisation or decontamination, is displayed in the monthly Waste Metric Dashboard. These documents are available on the LLW Repository Ltd website. The Dashboards show the predicted and actual values for these waste treatment options as part of the National Waste Programme. A five-year forward prediction of the amount of waste for these treatment options is given in the Joint Waste Management Plans for NDA sites and other nuclear sites (e.g. Shipton and Falconer 2013).

Higher Activity Waste (HAW) Storage and Disposal

Safe and secure decay storage of HAW in long-term interim stores at individual sites in needed until a GDF becomes available for England and Wales. Similar stores will be needed at Scottish nuclear sites until near site near-surface facilities for the long-term management of HAW are available in Scotland. As stated in section 1.3.4 of Chapter 1 the policy for the long-term management of Scottish HAW has been issued and an implementation strategy is being developed (Scottish Government 2007 and 2011). The current HAW policy (DECC 2014) considers the Scottish requirement in conjunction with a GDF, however, the Scottish position is excluded from the research.

HAW storage and disposal can be in shielded packages that can be contact-handled or unshielded packages that require remote-handling. Similarly, interim stores can be unshielded for shielded packages or shielded for unshielded packages.

2.5.4 HAW containers and interim stores

Stainless Steel Packages and Store Options

Radioactive Waste Management (RWM) Ltd endorsed packages for GDF disposal are deemed suitable for long-term interim storage of HAW at sites and for the Scottish long-term HAW management requirement, pending final disposal.

Until recently the preferred option for ILW was grouting and capping with cement to produce a stable waste-form package in thin walled unshielded 'single use' stainless steel 200litre and 500litre drums (Figure 2-15), $3m^3$ boxes and drums (Figure 2-16) or shielded 4m and 2m boxes or concrete containers (see Box 1-2 and Box A-1 for container information). These constituted endorsed waste packages covered by Final Stage Letters of Compliance (LoC) issued by RWM Ltd (NDA 2012b for packages and NDA 2013a for LoC). Depending on the radiation levels from unshielded waste packages an external concrete over-pack may be required to allow handling, storage and transport from the store to the disposal facility.

The Letter of Compliance (LoC) system is part of the NDA disposability assessment process (NDA 2014c) to ensure that HAW waste packages for disposal to a future GDF meet the safety, environment, quality and regulatory requirements for the transport to, and handling and emplacement at, a GDF. It also assures the continuation of these requirements in the long-term post closure period of a GDF. There are four stages to the LOC process; pre-conceptual assessment of initial package design options, through the conceptual and interim assessments, to the final LoC for disposal of HAW to a GDF. This confirms to the Radioactive Waste Management Ltd (RWM Ltd), as the operator of a future GDF, that the HAW packages will be fully compliant with the GDF safety case.

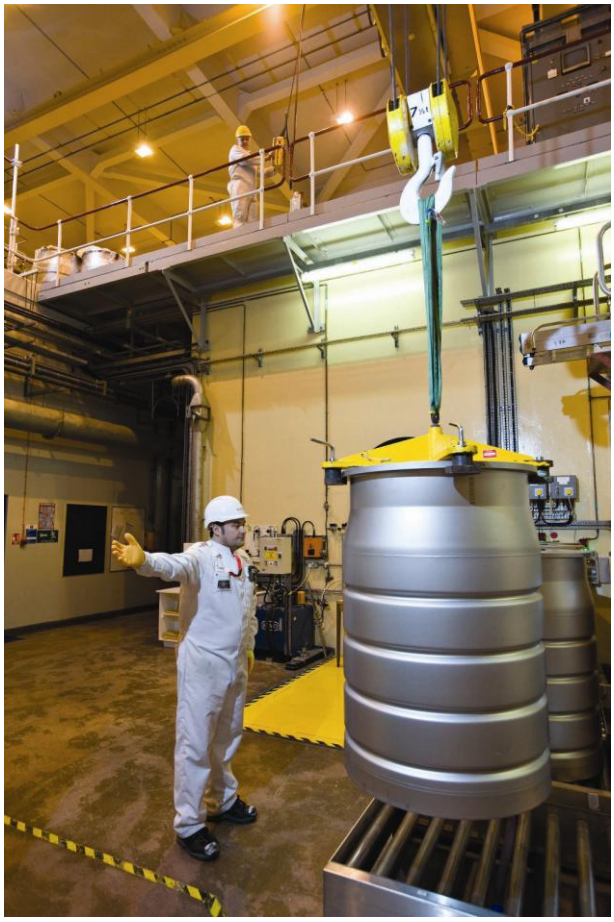

Figure 2-15 Demonstration handling of 500litre stainless steel drum for ILW (Courtesy of NDA)

Figure 2-16 An unshielded $3m^3$ ILW stainless steel drum of similar dimensions to the unshielded $3m^3$ stainless steel boxes at Hunterston A shown below (Courtesy of NDA)

The Trawsfynydd power station decommissioning programme adopted an unshielded long-term interim store (Figure 2-17) and 'single use' shielded packages (Figure 2-18). The Hunterston A power station decommissioning programme chose a shielded store with 'single use' unshielded packages (Figure 2-19). Both of these options can be extremely expensive.

Figure 2-17 Trawsfynydd unshielded long-term interim store (Courtesy of Magnox Sites)

Figure 2-18 Trawsfynydd shielded concrete over-packaged ILW in in 'single use' 3m^3 stainless steel drums or boxes prior to transfer to the store above (Courtesy of Magnox Sites)

Figure 2-19 Hunterston A stacking of demonstration 'single use' unshielded 3m^3 stainless steel boxes for ILW in their shielded long term-interim stores (Courtesy of Magnox Sites)

Ductile Cast Iron Containers and Unshielded Stores

Recently there has been a growing interest in thick walled ductile cast iron containers (Figure 2-20) from Germany. Summary DCIC data can be found in Box A-1.

Figure 2-20 Thick walled DCIC self-shielding Mini-Stores at Dungeness A power station (Courtesy of NDA)

The containers are self-shielding packages that can be stored in lightweight steel and concrete long-term interim stores. There is also precedence for some of these containers being made from 15-25% recycled LLW steels (Quade and Kluth 2009, CoRWM 2008a). The ductile cast iron containers give a potential for beneficial re-use of treated and melted UK LLW steels. There are, however, concerns about the use of these containers by the Office of Nuclear Regulation on safety grounds (ONR 2011a and 2011b) and by the RWM Ltd on the acceptability of these packages for a GDF as they are still going through the Letter of Compliance process (NDA 2013a).

The Radioactive Materials Transport Team of the ONR raised concerns about certifying DCICs made with some recycled material. Magnox have confirmed that they will not use recycled material in their initial DCICs but recognise that this could be an option for manufacturing their DCICs at some time and they may wish to explore the possibility in the future (Butler 2015 pers. comm.).

Waste Package Issues

Minimising the waste package numbers and sizing interim stores to allow adequate contingency are important engineering, planning, regulating and costs issues. They also have a knock-on effect on final disposal requirements and costs.

The 2010 UKRWI (NDA and DECC 2011) estimated that about 64,000 ‘single use’ carbon steel ISO freight containers (see Box 1-2 and Box A-1) will be needed for LLW disposal. These containers are readily available and cost around £8000 to £10,000 each (LLWR 2014 and Table A-1).

The 2010 UKRWI also estimated that a mixture of about 230,000 ‘single use’ stainless steel containers and ductile cast iron containers (see Box 1-2) will be needed for HAW storage and disposal. The cost of these containers is commercially sensitive and hence not readily available. However, costs do exist in the public domain, e.g. $3m^3$ drum prices have varied between ~£5000 in Ove Arup 1997 to ~£25000 in CoRWM 2008b. Bryan 2005 states that a 500litre drum costs about £1500. NDA 2012a uses production costs of ~£20,000 for a $3m^3$ box, £60,000 for a 4m box and £23,000 to £49,000 for bespoke packages. These costs are significant given the number of packages needed for disposal. See Table A-1for a summary of container costs.

The ‘single use’ carbon steel ISO freight containers for LLW and stainless steel containers for HAW use cement based grouts to bind and cap the wastes to produce a final waste package. Concrete is also used for some LLW and HAW containers, shielding over-packs, long-term interim stores for ILW and backfill for LLW repositories at Drigg and Dounreay plus vaults in a future GDF. Hence the environmental impacts of container materials, waste storage, transport, grouting, backfill and disposal, and associated costs, are investigated in Chapters 7, 8 and 9.

2.6 What are the current proposed waste solutions?

This section discussed the overall current position for legacy wastes and the policies, strategies and plans for their long-term management.

2.6.1 General position

The legacy wastes from the UK nuclear fuel cycle date from the mid1940s and will be managed beyond circa 2110. The waste arising from operation and

decommissioning is sporadic as shown in Figure 2-21 and Figure 2-22 for LLW and ILW. These projections are taken from the 2013 UKRWI (NDA and DECC 2014e).

Figure 2-21 shows a significant reduction in LLW generation between now and about 2033. This corresponds to the care and maintenance preparations phase for Magnox decommissioning and entry to in-situ and containerised decay storage. There is then a peak between circa 2038 and 2050 for Springfields decommissioning. This is followed by a quiescent period until circa 2070 for the Magnox decay storage. Final decommissioning, demolition and disposal of wastes from the remaining facilities are from ~2070 to beyond 2110. The waste level fluctuates according to individual facility decommissioning plans.

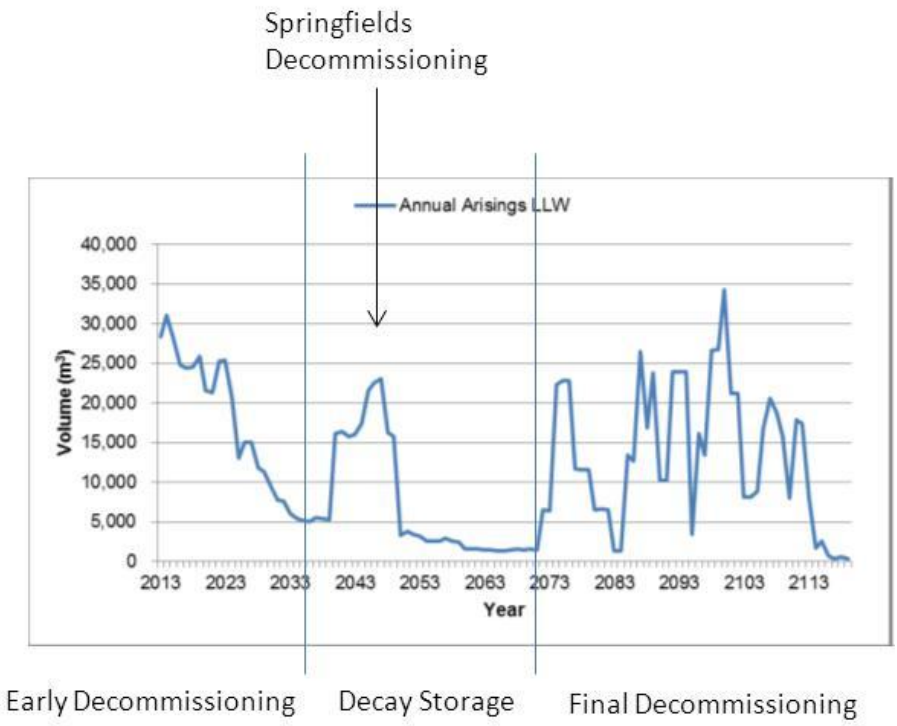

Figure 2-21 Estimated annual waste arising for LLW (based NDA and DECC 2014e Figure 3)

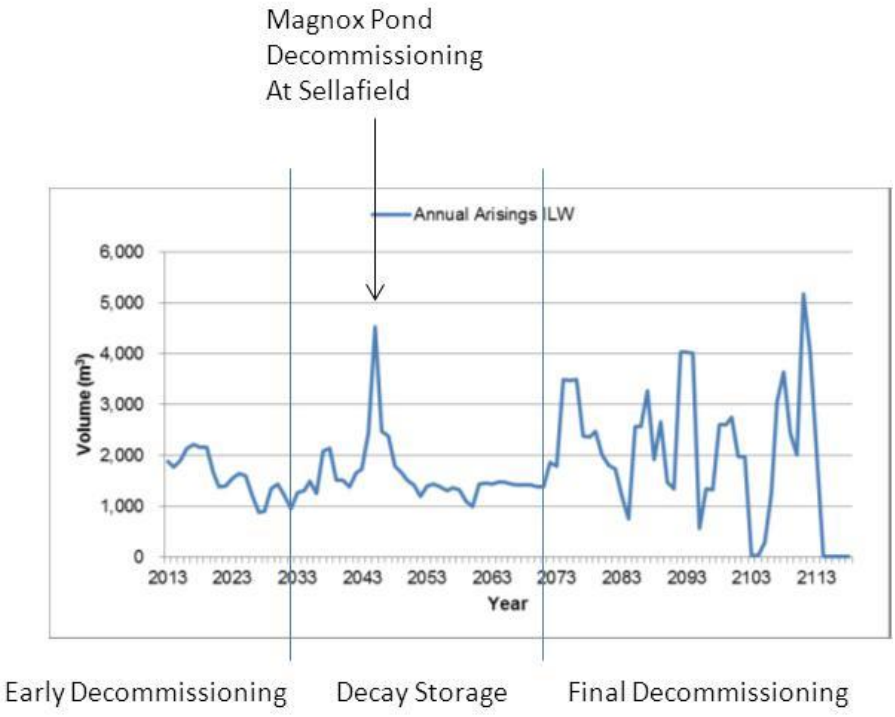

Figure 2-22 Estimated annual arising for ILW (based on NDA and DECC 2014e, Figure2)

Figure 2-22 shows fluctuating generation of ILW wastes between now and circa 2040. This represents individual short to medium term ILW recovery and encapsulation projects. The peak in circa 2043 is the Magnox ponds decommissioning at Sellafield and the 30 years plateau after is the quiescent period for Magnox decay storage. The period beyond 2073 is final decommissioning and varies according to individual programmes.

Both the figures include radioactive wastes from all legacy nuclear facilities, which will be decommissioned over the same time period. The figures do not include radioactive wastes for the proposed new nuclear power stations as previously discussed.

All three periods will result in treatment, disposal and recycling of radioactive metals. However, the exact timing and quantities of metals are hard to predict as will be discussed in Chapter 8.

2.6.2 Policy, strategy and plans

The Structured Approach to LAW

Figure 2-23 shows the flow of radioactive waste management documentation from the UK solid LLW policy (DEFRA et al. 2007) and the LLW strategy (NDA 2010a) through to the Joint Waste Management Plans (e.g. Shipton and Falconer 2013) and the monthly progress reporting under the National Waste Management Programme (NDA and LLWR 2009).

Figure 2-23 Key document flow for LLW management from LLW Repository Ltd website (relating to the National Waste Programme reproduced from LLWR Ltd website)

The figure shows the structured approach adopted to meet regulatory requirements and to encourage a culture change within the industry to:

- improve characterisation of LAW to allow some waste to be classified as high volume VLLW rather than LLW as seen in Figure 2-9,
- segregate LLW to allow super compaction, incineration and metallic waste treatment as discussed in section 2.5
- apply the Waste Hierarchy (LLWR and NDA 2009b), as shown in Figure 2-24 , to encourage the re-use and recycling of materials in preference to their disposal,
- conserve the scarce LLW disposal capacity and delay the requirement to construct a new LLW repository.

Waste minimisation and the application of the waste hierarchy are important issues to ensure the optimal use of interim storage and final disposal facilities. They underpin the costs effectiveness and affordability of the proposed programmes and plans. They are equally, if not more, important for HAWs (NDA 2012b and DECC

2014). HAW volumes are significantly smaller than LAW but packaging, storage, transport and disposal volume environmental and financial costs are significantly higher.

Figure 2-24 The Waste Hierarchy promoted by current UK LAW policy, strategy and plans (reproduced from NDA and DECC 2014a)

Developing HAW Management Structure

There is a growing concern about materials that could become waste in the future, such as depleted uranium products. Depleted, natural and low-enriched uranium (DNLEU) is currently classified as a zero value asset rather than waste but if eventually categorised as waste could constitute ~17% of the proposed future GDF volume (NDA 2013b). DNLEU and potential future wastes such as NORM will need to be retained in suitable containers and stores until a GDF or long-term management facilities are available. Legacy DNLEU stores like the one at Capenhurst and at Sellafield already exists. Urenco UK Ltd is currently spending in excess of €500million in developing a uranium tails management facility, including a uranium oxide store, for the storage of deconverted U_3O_8 pending future commercial and policy decisions on DNLEU. This is discussed in detail in Chapter 9.

Mandatory Mechanisms

A raft of environmental, health and safety legal requirements and guidance (e.g. HSE et al. 2007, NIEA and EA 2009 and NIEA et al 2009), underpin the policy, strategy and option analyses for storage and disposal of radioactive wastes. These

requirements and guidance help nuclear operators structure the assessment of treatment, conditioning, storage, transport and disposal of radioactive wastes and the recycling or re-use of treated radioactive metals.

The Environment Agencies are responsible for providing liquid, gaseous and solid waste authorisation and issuing transfrontier shipment authorisation to send wastes abroad for treatment and recycling plus the return of residual wastes where practicable. The transport and treatment of radioactive metals, at the international facilities, are discussed in Chapters 5, 7 and 8.

The Office of Nuclear Regulation is responsible for regulating the accumulation of solid wastes at sites, the disposability of secondary waste returned from international treatment and melting, and the transport of radioactive waste. The role of the Office of Nuclear Regulation now includes the roles and responsibilities of the former Office of Civil Nuclear Security for ensuring compliance with the Nuclear Industry Security Regulations 2003 and Ionising Radiation Regulation 1999. They are also responsible for ensuring compliance with the Class 7 (Radioactive Materials) requirements under the Carriage of Dangerous Goods Regulation 2009 and IAEA transport of radioactive material regulations (IAEA 2013).

2.7 Summary

This chapter summarised the background to radioactive waste and its management and showed that there were significant quantities of radioactive metals that could be recycled rather than disposed in Figure 2-11. It also raised potential issues with environmental impacts associated with the ‘single use’ of steel containers and the cement needed for waste packaging in Section 2.5.4. These are two aspects that are addressed in this research. The next chapter addresses the background to life cycle assessment and Chapter 6 outlines the analysis of financial costs, both of which are required to investigate the disposal and recycling scenarios considered in this research.

3 Background of Life Cycle Assessment

Chapter 2 summarised the background to radioactive waste management in the UK. It recognised that metals were not only fundamental to the infrastructure and processes of UK nuclear facilities but that they were a significant proportion of the radioactive waste produced by the facilities and are utilised for waste disposal containers.

This chapter discusses the use of life cycle assessment (LCA) to analyse the potential environmental impacts of radioactive waste management options and waste packaging. LCA is one of many tools (e.g. environmental risk assessment, environmental auditing, environmental impact analysis and strategic environmental assessment) used in environmental management decision making. It was developed to address the increased awareness and concern by the public, industry and government of the importance of protecting people and the environment from aspects resulting from industrial processes.

LCA focuses on the environmental impacts on human health, the eco-system and resources of the life cycle of products (goods or services) from extraction of materials to their disposals. It attempts to prevent shifting environmental burdens between the processes of the product system being considered or into other systems outside the study (Finnveden et al. 2009). LCA addresses the relative potential environmental impacts of the basic unit of the system being studied, i.e. the functional unit. This enables alternative products and scenarios to be analysed and compared to help identify possible areas for improvement in current situations or to address future potential problems. It is an iterative process, it generally simplifies relationships between processes and impacts and is linear and static (Baumann and Tillman 2004, Rebitzer et al. 2004, Pennington et al. 2004). LCAs can be extensive and complex hence it is important to make the modelling and assumptions as clear as possible to aid the interpretation of assessment results.

This chapter summarises the background to LCA and outlines the LCA process stages. It then discusses some of the limitations of LCA and the links with overall decision-making processes for radioactive waste management. A literature review of LCAs for the nuclear industry is discussed Chapter 4.

3.1 The History of Life Cycle Assessment Methodology

This section outlines the history of LCA by highlighting key developments that can be attributed to specific decades and organisations.

A history of LCA development is common in books (e.g. Baumann and Tillman 2004 and Sonnemann et al. 2004), articles such as Azapagic 1999 and Guinee et al. 2011 and dissertations or theses on the subject (e.g. Solberg-Johansen 1998, Lopes 2010, and Hetherington 2013). Guinee et al. 2011 usefully present a review of LCA in terms of the Past, Present and Future; this approach has been adopted here.

3.1.1 The Past

Early Developments

The period between 1970 and 1990 was called the "Decades of Conception" by Guinee et al. 2011 and represents the early development work. The energy analysis by Smith 1969 is usually credited as one of the early foundations of LCA (Sonnemann et al. 2004 and Hetherington 2013). The early work on energy analysis is noted by Azapagic 1999 and Sonnemann et al. 2004 who point out that Smith's work was in the public domain from cira1963.

These early analyses were expanded from primarily energy analysis to include raw material and solid waste flows in a study of beverage containers for Coca Cola by the Midwest Research Institute in 1969. The study was commercially sensitive and intended for internal use only but is generally credited with having introduced the concept of a systematic analysis of containers from "cradle to grave". The work was expanded to a resource and environmental profile analysis which quantified the environmental loading of the entire production system by Hunt et al. 1974 (Lopes 2010). The work by Hunt, Franklin and their colleagues (Franklin Associates) is also cited as an important initial step in the development of LCA by Azapagic 1999,

Sonnemann et al. 2004, Baumann and Tillman 2004, Lopes 2010, and Guinee et al. 2011.

Concurrent work by Boustead and Hankock on plastic and glass milk bottles in the UK, Sundstrom on energy requirements for beer containers in Sweden and by Basler and Hofman in the early 70s are discussed in Azapagic 1999 and Guinee et al. 2011. Their work, plus the work of the Franklin Associates, is also considered as laying the foundation of LCA (Guinee et al. 2011).

There was a hiatus of LCA studies after this initial development but interest grew again following the publication of a Swiss Federal Laboratories of Materials Testing and Research report on LCA in 1984 (Guinee et al. 2011). The "report presented a comprehensive list of data needed for LCA" and "introduced a first impact assessment method" (Guinee et al. 2011). This led to an increase in LCAs comparing consumer products using a diverse range of methods and resulted in heated debates on results and methodologies. This problem, especially in the USA, is noted by Hetherington 2013 and is seen as the driver for standardisation of LCA methodology.

Standardisation

The period between 1990 and 2000 was called the 'Decade of Standardization" by Guinee et al. 2011. The Society of Environmental Toxicology and Chemistry (SETAC) formalised LCA methodology between 1990 and 1994. This was followed by the development of the standards used today by the International Standardization Organisation (ISO) between 1997 and 1998. The harmonisation of LCA methodology and ISO standards currently used are summarised in Table 3-1.

It should be borne in mind that there were some differences between the SETAC and ISO frameworks. The most obvious difference is the specification a scope definition in ISO Stage 1 and that Valuation in the SETAC Impact Assessment is moved to in Interpretation in ISO. Also, ISO Interpretation includes other applications of the LCA results as well as the process improvement opportunities identified by the

SETAC framework. The remainder of this chapter, and the thesis, assumes the ISO terminology and stages.

Table 3-1 SETAC and ISO LCA Methodology Framework and Standards

SETAC Stages	**ISO Stages**	**Current ISO Standard**
1 - Goal Definition	1 Goal and Scope Definition	ISO 14040:2006 and ISO 14044:2006
2 - Inventory Analysis	2 - Inventory Analysis	ISO 14040:2006 and ISO 14044:2006
3- Impact Assessment; Classification Characterisation Valuation	3- Impact Assessment; Classification Characterisation	ISO 14040:2006 and ISO 14044:2006
4 – Improvement Assessment	4 – Interpretation; Valuation	ISO 14040:2006 and ISO 14044:2006

Based on Solberg-Johansen 1998, Azapagic 1999 and Hetherington 2013.

In addition to the standards presented in Table 3-1 ISO also produces technical reports (TR) and technical specifications (TS) for guidance. Guidance on overall principles and framework are given in ISO 14044 2006, examples of applying life cycle impact analysis and goal and scope definitions are given in ISO/TR 14047 2012 and ISO/TR 14049 2012. Guidance on LCA data documentation format and requirements is given in ISO/TS 14048 2002.

The development of LCA methodology and standards continues to improve and initiatives are in place for the future. It should be noted that ISO recognises that there is no single method for LCA and other valid approaches can exist. Hence, it is not necessary to adhere strictly to the ISO standards when conducting an LCA as it will depend on the goals and scope of the LCA as discussed later.

3.1.2 The Present

The period between 2000 and 2010, following standardisation, was called the "Decade of Elaboration" by Guinee et al. 2011. The period takes into account the effort made in Europe and America to incorporate LCA thinking in integrated product policy, sustainable resource use, waste prevention and recycling. It also includes the drive to promulgate "…quality-assured life cycle data, methods and studies for reliable decision support in (EU) public policy and in business" (Guinee et al. 2011). The late 1990s and the decade to 2010 saw a sharp rise in the production

of LCA papers for academic journals indicating the increased adoption of the standardised LCA methodology ((Finnveden et al. 2009 and Guinee et al. 2011).

The current standard (ISO 14040 2006) makes it clear that it does not specify in detail the methods used to conduct a LCA. The European Commission Joint Research Centre (EC/JRC 2010a) notes that "… the ISO process did not bring about detailed standardisation …" and that the United Nations Environment Programme (UNEP) and SETAC are working to develop best practice. Improvements are being made to LCA methodology through the UNEP/SETAC Life Cycle Initiative (Lopes 2010, Guinee et al. 2011 and Hetherington 2013). Phase I of the UNEP/SETAC Life Cycle Initiative programme 'Creating a community' ran between 2002 and 2007. It included programmes on life cycle management, life cycle inventory, life cycle impact assessment and life cycle cross-cutting (www.lifecycleinitiative.org, 2014). These programmes sought to improve decision making by providing information, sharing good practice, enhancing training, giving wider access to quality assured data via internet tools, access to expert groups and improved life cycle indicators across the whole life cycle approach. Phase II, 'Becoming a stakeholder', ran from 2007 to 2012 and sought to make LCA more participative. It aimed to involve global stakeholders in developing a better appreciation and agreement on life cycle strategies and tools. Phase III, 'Response to new challenges", started in 2012 and is planned to run until 2016. This phase aims to support the development of the type of Life Cycle Sustainability Analysis discussed by Guinee et al. 2011 that applies across society in general plus business and government for sustainable production and consumption.

Concurrently, the EC/JRC has developed a suite of documents, under the title of International Reference Life Cycle Data (ILCD) handbook (Wolf et al. 2012). These documents include for example EC/JRC 2010a addressing the analysis of existing LCIA methods, EC/JRC 2010b on detailed guidance for LCIA and EC/JRC 2011 on recommendations for LCIA in Europe. The benefits of the UNEP/SETAC and EC/JRC programmes to expand the environmental impacts considered and improve

assessment methods, hence quality and standing of LCA, are recognised in Finnveden et al. 2009.

3.1.3 The Future

The period from 2010 to 2020 was called the "Decade of Life Cycle Sustainability Analysis" by Guinee et al. 2011. As the name suggests it looks towards broadening LCAs from products and services to encompass the 'triple bottom line' of sustainability development, i.e. environment, economics and social aspects. It is beyond the scope of this research to investigate the potential of the holistic representation of sustainability in LCAs but it will be interesting to see how this progresses.

3.2 Industrial Processes and LCA Stages

This section discussed the basis of the unit process and products system that produces products, consumes material and energy and produces waste emitted to the environment. It also discusses the key stages of performing an LCA of a product system; goal and scope definition, inventory analysis, impact analysis an improvement assessment and interpretation.

Unit Processes and Product Systems

The SETAC definition of LCA presented in Section 1.4 included all the essential objectives of the life cycle assessment process. However, the current ISO standard defines LCA as the "compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system through its life cycle" (ISO 14040 2006). Life cycles are built up of individual unit processes, i.e. the "smallest element considered in the life cycle inventory analysis for which input data are quantified" (ISO 14040 2006). A generic unit process is shown in Figure 3-1.

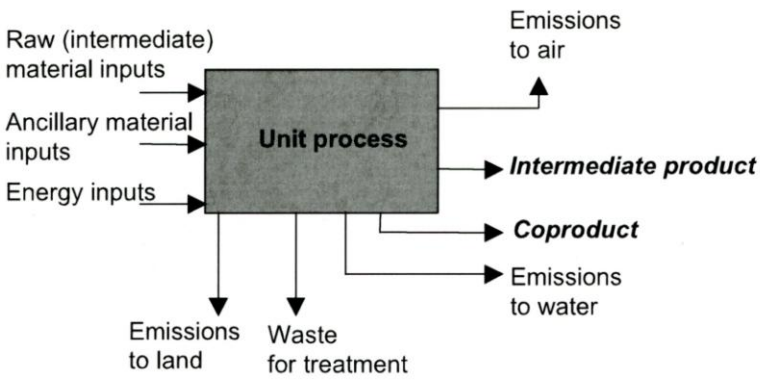

Figure 3-1 Generic unit process (reproduced from ISO/TR 14049 2000, Figure 4)

An industrial product system is a collection of unit processes that provides a product. Inputs to a product system include raw materials such as minerals, metals or chemicals and secondary materials such as recycled materials or packaging. Another input is the energy required to produce the products and manage wastes. The outputs are a main product, co-product(s) and/or by-product(s) plus solid, liquid or gaseous wastes. Co-products may be chemicals or materials that are produced in addition to the main product and of commercial value. For example, the hydrogen fluoride generated by de-converting uranium hexafluoride (UF_6) to uranium concentrate (U_3O_8) discussed in Chapter 9 is a very valuable co-product. A by-product is normally of low value and used in another product system. For example, blast furnace slag produced in steel making can be used as road fill, or as a supplement in cement-grout for solid radioactive waste packaging. The wastes are either emitted directly to the environment or processed through waste treatment facilities. A simplified product system, made up of a number of individual unit processes, elementary and product flows is presented in Figure 3-2.

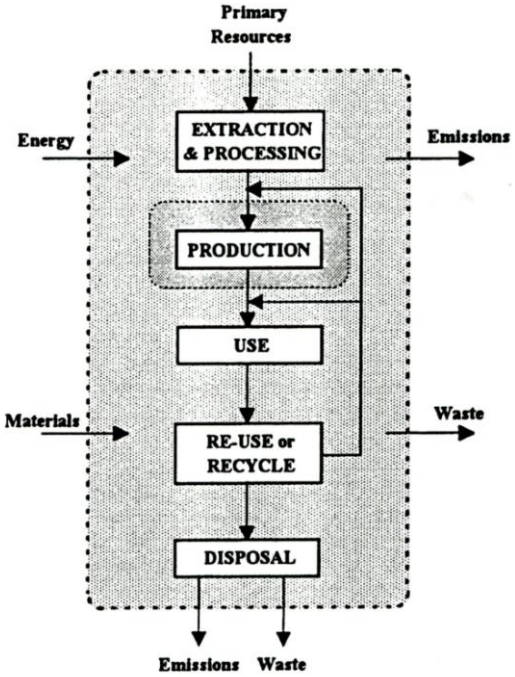

Figure 3-2 Simplified product system (reproduction of Azapagic 1999, Fig 1)

Overview of LCA Stages

The LCA methodology standardised by ISO is commonly represented by the four iterative stages as shown in Figure 3-3 (Sonnemann et al. 2004, Lopes and Hetherington 2013). The LCA stages model the material, energy and waste flows of individual unit process inputs plus, their outputs, to create a coherent life cycle output of potential environmental impacts. As noted in ISO/TS 14048 2002, LCAs can be done for the whole life cycle (Cradle-to-grave), from extraction to production (Cradle-to-gate), for production on a single site (Gate-to-gate) or from use to disposal (Gate-to-grave). However, all assessment should follow the stages outlined below.

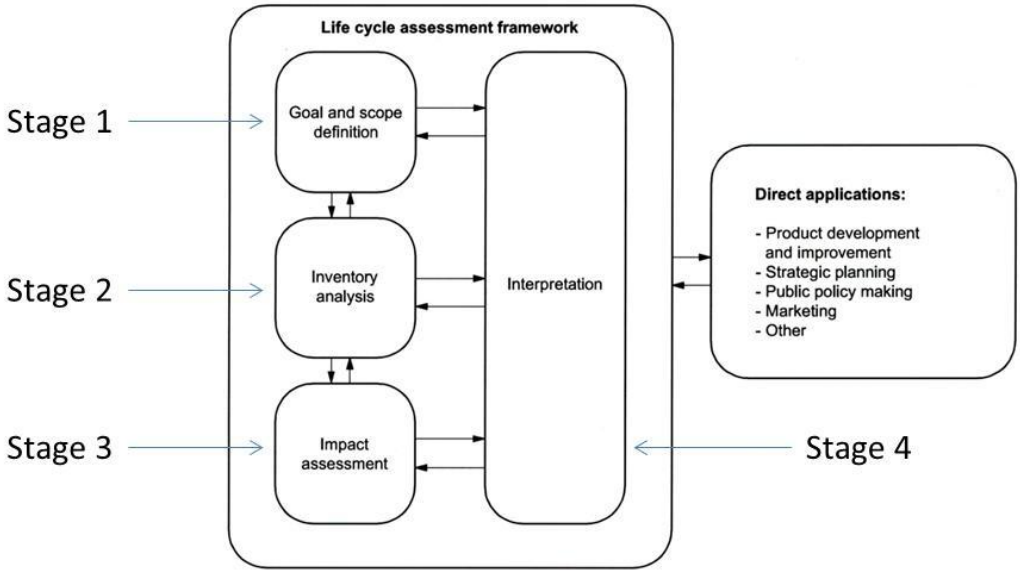

Figure 3-3 Stages of an LCA (based on ISO 14040:2006)

The ultimate outcome of an LCA is recommendations to decision-makers. Some of the applications of an LCA output are shown in Figure 3-3. Each of the four stages is now discussed in more detail.

3.2.1 Goal and Scope Definition

Defining the goal and scope is the initial stage of any type of LCA. They are the foundation of the methodology. The goal is described as follows in ISO 14040 1997, "The goal of an LCA study shall unambiguously state the intended application, the reasons for carrying out the study and the intended audience, i.e. to whom the results of the study are intended to be communicated". The scope can be then be structured to deliver the goal. The scope is described as follows in ISO 14040 1997, "The scope should be sufficiently well defined to ensure that the breadth, depth and details of the study are compatible and sufficient to address the stated goal". These statements are reflected in the current standard showing that the fundamental basis of an LCA has not changed (ISO 14040 2006).

The scope considers the product system and the function(s) to be compared, the functional unit, the system boundary, the allocation procedures and the impact assessment methods used. Hence, it forms the basis of the LCA results to be evaluated in the Interpretation stage. It also includes the data requirements, a critical review if necessary, and the format for the final report (ISO 14040 2006).

Functional unit: "quantified performance of a product system for use as a reference unit"

The functional unit is essential in all LCA. It has to be clearly defined, measureable and consistent with the goal and scope. It is the reference point for the development of the inputs and outputs for the life cycle inventory and to ensure comparability of the results generated by the LCA models. For example, the general format of a functional unit may be similar to: the production of 1kg of product 'A' by the product system 'B' for a specified use 'C'.

System boundary: *"set of criteria specifying which unit processes are part of a product system"*

There will be boundaries between the product system and the environment, and the processes within and across product systems. There will also be time and geography boundaries to consider as noted by Finnveden et al. 2009 and Guinee et al. 2011. A simple system boundary is visualised by the dotted enclosure in Figure 3-2. Process flows within and across the boundaries are important in the LCA, particularly when generating the life cycle inventory. A comprehensive list of what the system boundary, unit process and flows should consider is given in ISO 14040 2006 and ISO 14044 2006. The list includes raw materials, input and outputs of manufacturing, transport, fuels and power generation, waste disposal and recycling of products. Facility maintenance, infrastructure and management requirements can be included if required by the goal and scope.

Material and energy allocation for systems with multiple products and recycling are discussed in Section 3.2.2. The data evaluation is discussed in 3.2.2 and subsequent chapters. Impact assessment is discussed in Section 3.2.3 and interpretation in Section 3.2.4. The critical review and report requirements are taken to be an integral part of the thesis process in this case.

Life cycles are iterative as shown by the directional arrows in Figure 3-3. Hence, the goal and scope needs to be re-visited as the iterations proceed to ensure that their fundamental requirements are still being met or modified.

3.2.2 Inventory Analysis

The second stage of the methodology is life cycle inventory analysis. This involves building a system flow model identifying and collating the data for all energy and mass flows, data calculation (i.e. the normalisation of all data to the functional unit) and the allocation (i.e. quantified distribution) of resources, energy and environmental emissions from transport, land use, solid wastes disposal and liquid and gaseous discharges from the product system (Baumann and Tillman 2004, Rebitzer et al. 2004, Sonnemann et al. 2004). It is an iterative process recording and manipulating these data consistent with the requirements of the goal and scope, and evaluating the quality of the data used. However, Baumann and Tillman 2004 note that it is an "incomplete" energy and mass balance for the system as only the most environmentally relevant flow are included (based on time, money and experience constraints). The output is an inventory table identifying and quantifying resources depletion, (minerals use, fossil fuel for electricity generation or oil as feedstock for plastics), land use and damage, plus emissions to air, water and soil (e.g. carbon dioxide (CO_2), greenhouse gases (GHGs), sulphur dioxide (SO_2) and other gases and heavy metals to soil and water courses). The inventory analysis is the basis of impacts assessment that attributes inventory data to damaging environmental effects of the product system.

Data collection

Good data is the key to a successful LCA. Hence, where possible recent published data were used within the research and where historical data was used it was highlighted. The process of choosing and recording the data can be aided by producing a flow diagram, similar to Figure 3-2, to identify the requirements. These requirements may change as the LCA process iterates and more data may be needed. The benefit of devising a foreground and background system to assist in developing the system boundaries is noted in Azapagic 1999, Azapagic in Clarke and Macquarrie 2002 and Baumann and Tillman 2004. The foreground system contains

all the key processes for the LCA. It receives inputs from the environment and background systems and rejects outputs to them. This helps define the data requirements, i.e. where practicable the data should be specific for the foreground system but can be from published data or electronic databases for the background system. The development of industry standard databases (e.g. the Ecoinvent database) and how they can be used is discussed in Finnveden et al. 2009.

Data calculation

This includes data validation, relating the data to the functional unit and hence to the unit processes and flows for the product system. Sonnemann et al. 2004 cite 1kg of material or 1km travelled as examples, but it will be defined by the functional unit. The current ISO standard states the LCA also needs to consider different fuels and electricity sources, efficiency of conversion and distribution of the energy systems adopted (ISO 14040 2006). The mix of coal, gas, nuclear and renewable energy used in a LCA may be particularly important when the processes being modelled take place in different countries.

Allocation of flows and releases

Allocation procedures, i.e. "partitioning the input and output flows of a process or product system between the product system under study and one or more other product systems" need to be considered in conjunction with system boundaries (ISO 14040 2006 and ISO 14044 2006). These can be particularly important when processes are shared with other product systems, where there are multiple products or co-products, wastes and recycling. A three step allocation process is outlined in ISO 14044 2006:

1) avoid allocation by dividing or expanding the system to include additional functions,
2) when allocation cannot be avoided separate inputs and outputs between different products or functions such that they reflect the relationship between them,
3) where physical allocation cannot be established or used use an independent parameter such as economic value.

Allocation is the most commonly discussed issue in LCA according to Finnveden et al 2009. The authors note that whereas system expansion (1) is advocated by Weidema 2003[1] as always possible and will reduce the allocation issues, Heijungs and Guinee 2007[1] argue that it "…is impractical because of the large uncertainties involved and the lack of data on what is avoided" by adopting system expansion. They also note that economic, mass and energy for (3) are the most commonly used allocation according to Lundie et al. 2007[1].

Data Evaluation

The LCA data requirements are specified in detail in ISO 14040 1997 and ISO 14044 2006. Data should be related to the goal and scope and be of suitable quality to make the results of the LCA credible. Data quality considerations include; the age and timeliness of the data, their geographical and technological coverage, precision, completeness and representativeness, consistency and reproducibility and their uncertainty. It is also important to note data gaps and how the missing data are treated. Where practicable data for the research was taken from the UK radioactive waste inventories and published documents as part of the NDA, LLWR and nuclear sites research activities and are cited throughout the thesis.

As noted by Lopes 2010 and Hetherington 2013 data collation and manipulation can require the use of computer software packages. Several software packages exist and Sonnemann et al. 2004 list the most common packages.

3.2.3 Impact Assessment

The inventory table created by the inventory analysis can be large and difficult to understand. Hence a third stage, life cycle impact assessment (LCIA), is needed. The impact assessment stage estimates the level of potential environmental impacts using the data in the inventory table. This is to better understand the significance of the inventory results and to provide information for the interpretation stage.

[1] These references are cited in Finnveden et al 2009 but are not included in the thesis references

The current ISO standard (ISO 14040 2006) notes the introduction of a degree of subjectivity in the assessment stage and splits it in to three mandatory elements and three optional elements as shown in Figure 3-4. Using the optional elements is contentious as the subjectivity is higher.

Mandatory Elements

The first element of LCIA is choosing the impact categories to link to the inventory table results in accordance with the LCA goal and scope. Category indicators quantify the impact categories such as global warming, photochemical oxidation (smog) and acidification. There are two types of category indicators - midpoint and endpoint (Pennington et al 2004, Sonnemann et al. 2004, Finnveden et al. 2009 and Hetherington 2013).

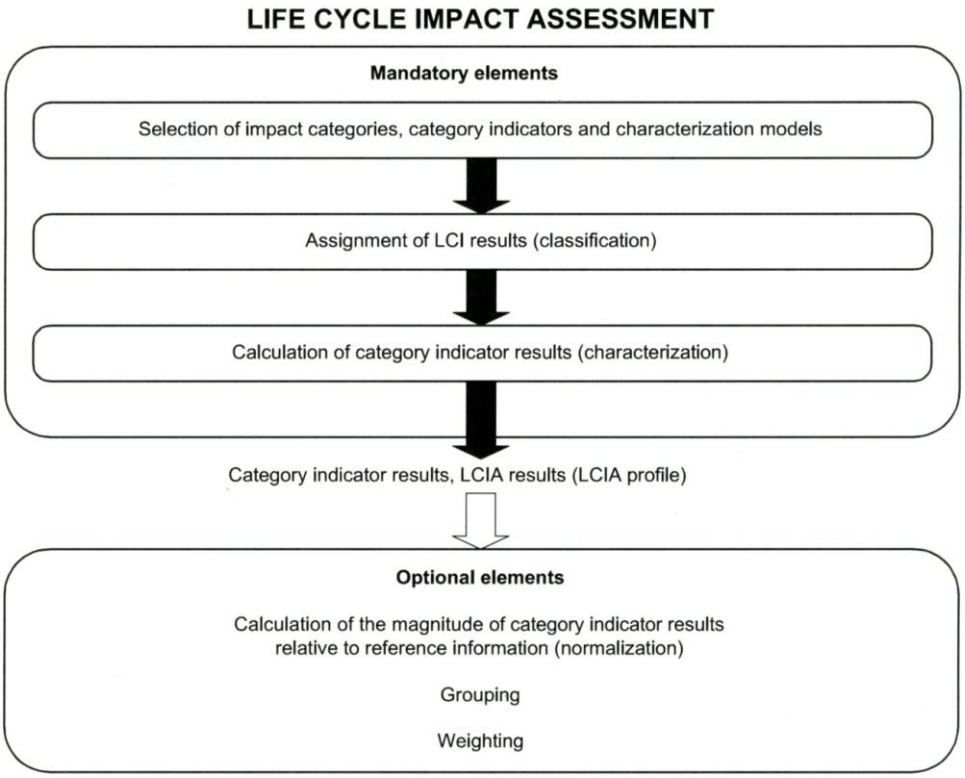

Figure 3-4 Elements of the LCIA Phase (reproduction of ISO 14040 2006, Figure 4)

Midpoint indicators, such as radiative forcing by CO_2 and CH_4, are estimated using long established techniques. They are part way between the emissions and the end

points (Finnveden et al. 2009) and have relatively low subjectivity and uncertainty (Hetherington 2013). However, decision makers may not find them easy to interpret.

Endpoint indicators apply to areas of protection (Pennington et al 2004) or damage categories (Goedkoop and Spriensma 2001a), i.e. human health, ecosystem quality and resources. These may be easier to relate to for decision makers. ISO/TR 14047 2003 shows the relationship between the three endpoints and specific impact categories used in the life cycle impact analysis method Eco-indicator 99 (Goedkoop and Spriensma 2001a). This is reproduced in Figure 3-5. Eco-indicator 99 is the life cycle impact assessment method used in the research and is discussed further in the research methodology in Chapter 6.

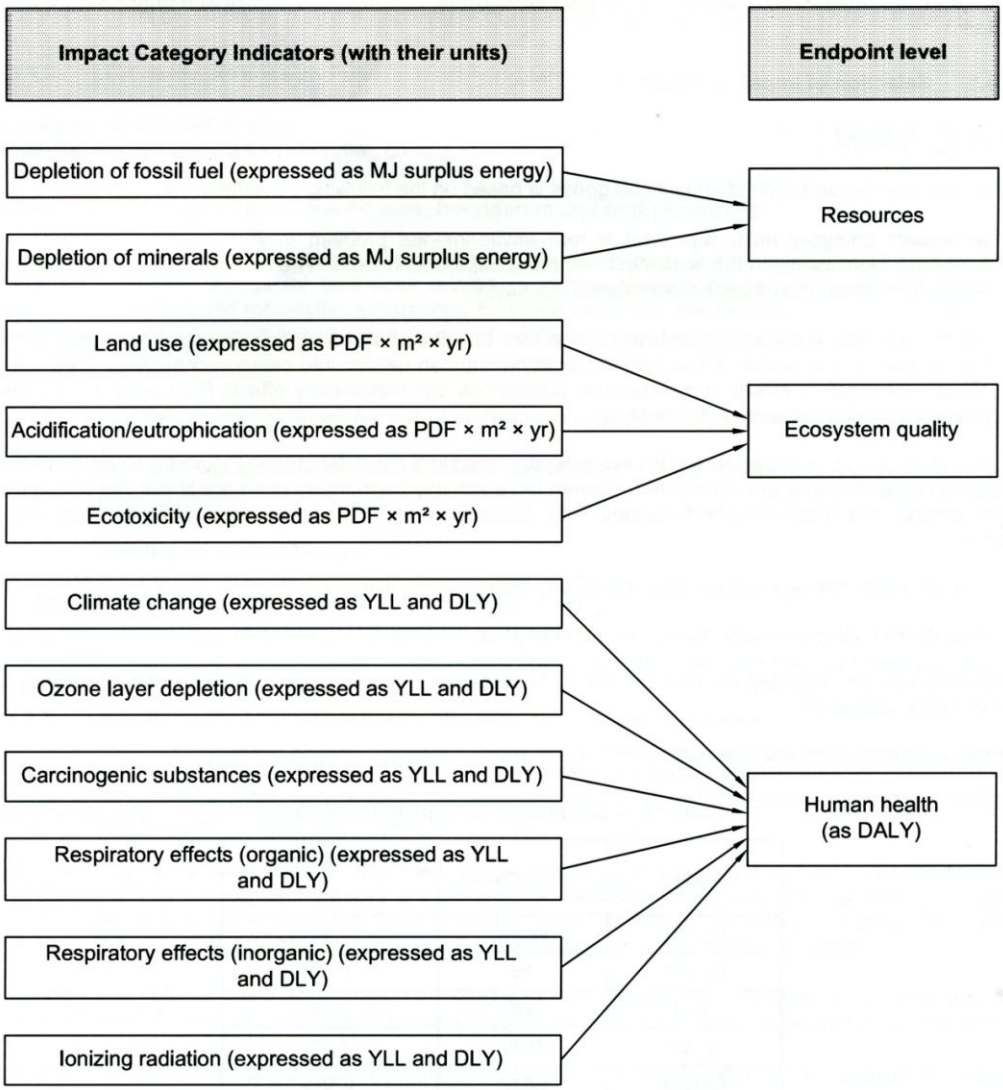

Figure 3-5 Environmental Impact Categories linked to endpoints, or damage categories (reproduced from ISO/TR 14047 2003, note this is an older version of the current 2012 technical report but shows clearly how individual impact category indicators can be associated with endpoints, MJ – Mega Joules, PDF – Potentially Disappeared Fraction, YLL – Years of Life Lost, DLY – Disabled Lived Years, DALY – Disability-Adjusted Life Years**)**

Classification

The second mandatory element is classification. It assigns inventory data to one or more impact categories. For example CH_4 contributions can be attributed to both climate change and photochemical oxidation (smog) which results in respiratory effects. Chloroflourocarbons (CFCs) contributions can be attributed to both climate change and ozone layer depletion. It then aggregates all the contributing stressors for an impact category. The climate change, respiratory effects and ozone layer depletion category indicators would then be included in the human health endpoint.

Characterisation

The final mandatory element is characterisation. This quantifies the classified and aggregated inventory data per category indicator to estimate the potential damage they cause. This is achieved using characterisation factors. These depend on the LCIA method chosen and whether the method uses midpoint or endpoint analysis. A generic characterisation factor for emissions from Pennington et al. 2004 is shown in Equation 3-1, where the subscript (s) represents the chemical per unit mass discharged to the air, water or soil:

$$\text{Category Indicator} = \sum_S \text{Characterisation Factor (s).Emission inventory(s)}$$

Equation 3-1 Generic characterisation factor for emissions (Pennigton et al. 2004)

A similar linear generic characterisation equation is cited by Azapagic in Clarke and Macquarrie 2002 and in Hetherington 2013. The equation for a non-generic characterisation factor for human health or ecosystem quality endpoints is shown in Equation 3-2. The equation relates to the Eco-Indicator 99 Fate – Emission - Exposure – Effect – Damage Analyses shown in Figure 6-1and discussed in Section 6.1.1.

$$\text{Characterisation Factor } (s,i,t) = \sum_j \frac{\text{Effects } (s,j,t)}{\text{Emissions } (s,i)}$$

$$= \sum_j \frac{\text{Fate } (s,j,t)}{\text{Emission } (s,i))} \cdot \frac{\text{Exposure } (s,j,t))}{\text{Fate } (s,j,t))} \cdot \frac{\text{Effect } (s,j,t))}{\text{Exposure } (s,j,t))}$$

Equation 3-2 Non-generic characterisation factor for emissions (Pennigton et al. 2004)

Where the subscript (s) again denoted the chemical, (i) is where the emission occurs, (j) is where the exposure occurs and (t) is the time for which the potential environmental contribution is relevant (Pennington et al. 2004). The equation takes a pollutant from the inventory, assesses where emissions are experienced and how long

they last (Fate), how many people or species are affected (Exposure), what the changes are within the environment (Effect) and then aggregates (and perhaps weights) the impact to show the damage (Kirchain circa 2010) and ISO/TR 14047 2003). Both types of characterisation factors are usually available in databases used in LCA software packages and the literature (e.g. Goedkoop and Spriensma 2001a and 2001b, Baumann and Tillman 2004) and are generally specific of the LCIA method.

Optional elements

Normalisation

The first optional element is normalisation. The category indicators all have different units, hence to compare them to find any anomalies or to understand the relative magnitude of potential environmental impacts it can be beneficial to divide the sum of the individual results by a reference value (Pennington et al. 2004 and ISO 14044, 2006). The reference value will depend on the LCIA method used but could be the total inputs or outputs of a specific area or per capita (e.g. the UK, Europe etc), or against a documented baseline case. Hence the normalisation can be based on natural science, economic or social science. It is typically derived as shown in Equation 3-3, where the subscript (k) is the environmental impact category:

$$\text{Normalised Indicator}_k = \text{Category Indicator from Characterisation }_k / \text{Reference Value }_k$$

Equation 3-3 Normalisation factor (Pennington et al. 2004)

Normalisation factors are usually available in LCA software packages and the literature (e.g. Goedkoop and Spriensma 2001a and 2001b) and are generally specific of the LCIA method. This type of normalisation is used within the Eco-Indicator 99 life cycle impacts assessment method as discussed in Section 6.1.1.

Grouping

Grouping allocates the impact categories to a predefined set(s) specified in the goal and scope (ISO 14040 2006). The sets can be purely qualitative or semi-quantitative to give an indication of priority or ranking of potential environmental impacts e.g.

low, medium or high. Such groupings could be useful for interpretation and communication.

Weighting

Weighting, or valuation, is the most contentious of the three optional elements (Pennington et al. 2004, Finnveden et al, 2009). Normalisation gives an indication of relative magnitude but says nothing about relative importance of the impact category. Grouping gives a broad indication of relative importance but is rather vague.

As noted by Finnveden et al. 2009 "*Despite the controversies, weighting is widely used in practice*", but the method used should be clearly recorded and stated. Weighting can give a more structured approach and be related to midpoint or endpoint stages of the impact assessment. It can be achieved using monetary considerations expert panel (Goedkoop and Spriensma 2001a) or distance to target methods as cited by Pennington et al. 2004. It can also use expressed preferences (e.g. panels and some monetary methods), or revealed preferences (taxes, insurance of the payment of legal fines etc) (Pennigton et al. 2004). Weighted results are not allowed under ISO 14040 2006 for comparisons presented to the public. Where used in other cases the assumptions and method used for weighting must be clearly specified. This thesis is intended for use by decision makers within the UK nuclear industry rather than the public, hence weighted impact category results are discussed in Chapter 7 to 9 as well as normalised results. Weighting factors are usually available in LCA software packages or literature (e.g. Goedkoop and Spriensma 2001a and 2001b, Baumann and Tillman 2004) and are generally specific of the LCIA method.

3.2.4 Improvement Assessment and Interpretation

The final SETAC stage is Improvement Assessment. It aims to minimise the burden of potential environmental impacts by identifying possible improvements or innovations in the environmental performance of the current situation or of proposed future projects.

The final ISO stage is Interpretation. It considers the significance of the LCA results as a whole and the quality of the data, i.e. their completeness, sensitivity and uncertainty, as well as the improvement opportunities. It should also make clear that the LCA addresses relative potential impacts and does not derive actual impacts, environmental safety margins (e.g. the approach to legislated limits or thresholds) or environmental risk.

The impact assessment results are used by both the SETAC and ISO systems to draw conclusions and make recommendations on the applications of the LCA such as those shown in Figure 3-3. Overall, the interpretation stage should confirm that the LCA is readily understandable, effectively complete and consistent with the final goal and scope, thus giving confidence in the methodology and the LCA results.

3.3 Limitations of LCA

This section discussed some of the perceived limitation highlighted in the current standard, the potential obstacles to using LCA and the limitations of using LCA is decision making.

Limitations Highlighted in the ISO Standard

The current LCA standard (ISO 14040 2006) identifies that:

- Economic and social aspects are not commonly considered or easily represented in LCA methodology. However, economic and social science methods can be used to make decisions in LCA if natural science methods cannot be used
- Data collection can be resource intensive, hence time consuming and expensive
- Value choices in modelling and assessing impact categories introduces subjectivity into the assessment phase
- Only environmental issues specified in the goal and scope are assessed in LCA, hence an "LCIA is not a complete assessment of all the environmental issues of the product system under study" (ISO 14040 2006)

- Limitations in the characterisation models, assessment methods plus inadequate and incomplete inventory data mean that "LCIA cannot always demonstrate significant differences between impact categories and related indicator results of different product systems" (ISO 14040 2006)
- The lack of spatial and temporal data, and their variation across impact categories, introduces uncertainty in the assessment results, and
- "There are no generally accepted methodologies for consistently and accurately associating inventory data with specific environmental impacts. Models for impact categories are in different stages of development". (ISO 14040 2006)

Most of these issues are also recorded in the original 1997 issue of the ISO standard, hence are well recognised.

Perceived obstacles to the wider use of LCA

As discussed in Section 3.1.1 there have been issues with the poor credibility in LCA as a result of extravagant claims about findings in the past (Hetherington 2013). This leads to concerns that the LCA commissioner can specify the system boundary and hence the limits of the study, making the LCA subject to bias and subjectivity. Some concerns have also been raised about the need for experts, or groups of experts, to ensure the quality of LCAs (e.g. Wolf et al. 2012). There also appears to be a plethora of category indicators and models used, and the lack of standard benchmarks makes it difficult to assess the comparability of results.

Limitation of LCA in Decision Making

The "Positioning and Application of LCA" by Cowell et al. 1997 - cited by Solberg-Johansen 1998 - identifies why LCA alone is not generally accepted as the basis for decision-making. The reasons noted by the authors were:

- "The decision maker may consider the environmental impacts assessed in the LCA to be irrelevant to the decision
- The cost of a comprehensive LCA may outweigh its usefulness

- LCA does not consider the strategic context of the decision, e.g. decisions with long term implications must consider future innovation alternatives
- LCA does not generally consider the wider implications of decisions such as changes in the structure of an industrial sector".

Hence, for LCA to be used to its fullest potential it must either be combined with other environmental management tools as well as societal and economic methods within the decision making process as discussed in the literature review in Chapter 4, or value judgements included in the LCA process in a structured and transparent way. The research methodology (Chapter 6) discusses the use of a life cycle impact assessment method (Eco-Indicator 99) to address the uncertainties that arise from value judgements on subjective issues involved in LCA.

A number of limitations and obstacles above were encountered during the reseach as discussed for the LCA modelling presented in Chapters 7, 8 and 9. For example, the published 2010 UK radioactive metals inventory data is used in Chapter 8 and the current UK depleted uranium inventory data is used in Chapter 9 to minimise the time and cost of building the LCA models. Value judgements are made on what scenarios and associated processes and materials can be investigated in Chapters 7, 8 and 9, again to minimise time and costs. Direct financial costs had to be estimated from published data to combine with the LCA modelling in Chapters 7, 8 and 9 to form the basis for future decision making, again these are value choices linked to the goal and scope of the research in each chapter.

3.4 Summary

LCA has a sound foundation, its methods and procedures are to international standards. The overall LCA methodology has improved over the last two decades and should continue to improve in the future. It has limitations and obstacles to its wider acceptance, but these can be overcome by the transparent application of the methodology. This is achieved in the research by stating software and life cycle impacts method used to perform the LCAs, presenting or citing the data sources used to develop the inventory and stating clearly what assumptions were made for each

scenario invenstigated. Hence, LCA appears to be a suitable tool to assess the potential environmental impacts of material, energy, equipment and processes for radioactive metals disposal and recycling within the nuclear fuel cycle shown in Figure 1-1.

4 Review of the application of LCA to support decision making in the nuclear industry

There has been a general move in the last few decades from the control and regulation of environmental issues focusing on site specific, or even project specific, end-of-pipe solutions to avoiding and monitoring adverse environmental impacts of products and systems (e.g. Solberg-Johansen 1998, ISO 14040 2006 and European Community/Joint Research Council (EC/JRC) 2010a). This is seen in changing environmental policies globally with a move towards striving for continuous environmental improvement and an incorporation of environmental considerations into strategic decision making by businesses. This focus on a more holistic approach to environmental impact management supports the philosophy of sustainable development and sustainable production and consumption (EC/JRC 2010a). The Bruntland report states "Humanity has the ability to make development sustainable to ensure that it meets the needs of the present without compromising the ability of future generations to meet their own needs" (World Commission on Environmental Development (WCED) 1987(p15)). Sustainable development therefore addresses the economic, environment and social issues of human interventions on the environment. Many tools exist to analyse each aspect of this 'triple bottom line' of sustainable development. This research adopts life cycle assessment (LCA) as the analytical tool for assessing potential environmental impacts within the conceptual framework of Life Cycle Thinking.

This chapter presents a review of past applications of LCA to support environmental impact assessment and decision-making within the nuclear industry. It discusses international LCAs of the whole nuclear fuels cycle, American studies on decontamination and decommissioning in particular and UK nuclear related LCAs focussing on decommissioning to identify gaps to be addressed by this research.

4.1 International LCAs of the Nuclear Fuel Cycle

This section discusses the key issues arising of LCAs from five countries with varying system boundaries and LCA methods, it then compares the results of this

review with more recent studies and focuses on the results of the end-of-life phase of the LCAs to identify gaps to be investigated in this research.

The debate on anthropogenic climate change impacts from burning fossil fuels is a major global issue (Lenzen 2008) that has led to a large number of LCAs on power generation (e.g. Hondo 2005). This is also reflected in the large number of LCAs on energy and greenhouse gas (GHG) emissions for the nine generic nuclear fuel cycle stages shown in Figure 1-1 (e.g. Sovacool 2008, Warner and Heath 2012). The generic stages shown in Figure 1-1 are the basis for the nuclear fuel cycle LCAs. The first review of nuclear fuel cycle LCAs considered here was conducted by Fthenakis and Kim 2007 using data from a range of studies from the USA, Australia, Sweden, Switzerland, Japan and a reference light water reactor study proposed as a worldwide model. Fthenakis and Kim 2007, and other frequently discussed nuclear fuel cycle LCAs from the literature, are discussed here and specific American and UK nuclear decommissioning related LCAs are discussed separately in Sections 4.2 and 4.2.

4.1.1 Key Issues from the review by Fthenakis and Kim 2007

Fthenakis and Kim 2007 compare the GHG emissions from LCAs for solar electricity and nuclear power. Examples of the GHG emissions from nuclear fuel cycles LCAs are presented in Figure 4-1 and highlight several key issues.

The results in Figure 4-1 show that uranium enrichment dominates the GHG emissions of the nuclear fuel cycle, except for the Storm van Leeuwen and Smith 2005 study. The extent of emissions due to enrichment depends on: – the grade of the uranium ore used and the uranium recovery from process tailings - which enrichment process is preferred (gas diffusion or gas centrifuge, where the gas diffusion process in much more energy intensive than the gas centrifuge process) - the desired enrichment level – whether generic or power station specific data is used – whether fuel is used once or reprocessed – what the energy mix is for the country undertaking the enrichment process (Fthenakis and Kim 2007, Lenzen 2008,World Nuclear Association (WNA) 2012 and 2014).

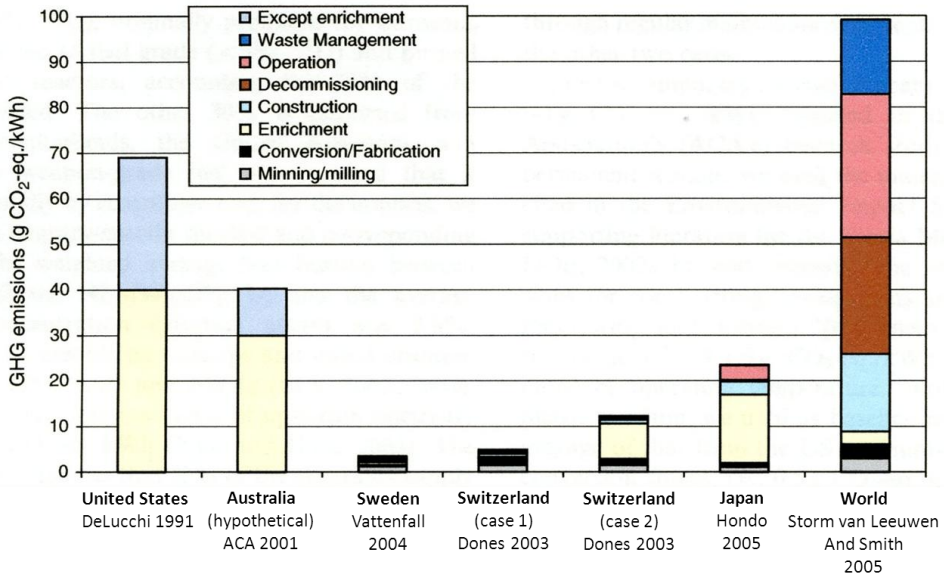

Figure 4-1 Comparison of GHG emission for the nuclear fuel cycle (from Fthenakis and Kim 2007, Fig 4)

So, for example, in the USA, Australia, Switzerland (case 2) and Japan (Figure 4-1), enrichment accounts for over half of the GHG emissions because the USA enrichment is dominated by the gas diffusion process and both the USA and Australia have a high fossil fuel energy mix. The Switzerland (case2) and Japan GHG emission are lower than the American and Australian studies because of the mix of gas diffusion and gas centrifuge enriched fuel, and have a low and moderate fossil fuel energy mix respectively. Where as in Sweden and Switzerland (Case 1) the percentage of emissions is much less because they both have a low fossil fuel mix and they use power station specific data. The enrichment process, energy and LCA model for each country is discussed below in more detail.

The total estimates of GHG emissions in Storm van Leeuwen and Smith 2005 are considerably higher than the other results presented in Figure 4-1, but their enrichment contribution is low. Storm van Leeuwen and Smith use enrichment energy data from 1976/76 with a low gas diffusion (30%) and high centrifuge (70%) fuel mix that results in low enrichment impact (Storm van Leeuwen and Smith 2007). Although the authors do discuss energy mix (see the authors' revised study Storm van Leeuwen and Smith 2007) but it is not clear how this is used in the enrichment estimates.

A nuclear fuel cycle LCA, for the Environmental Product Declaration for Torness nuclear power station was produced in 2005 (Atomic Energy Authority Technology (Environment) (AEAT (E)) 2005). The life cycle stages addressed were; extraction of uranium, uranium conversion, uranium enrichment, fuel fabrication, power station operation, power station construction and decommissioning, spent fuel reprocessing, operational waste facilities and construction of the waste facilities. The key findings of the LCA were:

- Extraction gives the highest contribution for CO_2 (GHG) and SO_2 and NO_x (acidification) emissions. Variations in these pollutants is due to fossil fuel use and uranium content of ores from various countries
- Power station construction, operation and decommissioning are also significant phases in the life cycle for the environmental impact categories considered
- Uranium conversion and fuel fabrication phase environmental impacts are small compared to the phases above
- Uranium enrichment impacts are low as the Torness fuel was made using the gas centrifuge process by Urenco Ltd at Gronau in Germany.

The main pollution levels were 5.06g CO_{2eq}/kWh, 0.01g SO_2in air/kWh and 0.02g NO_xin air/kWh, but acidification, eutrophication and ground level ozone depletion were also considered. The LCA also investigated radioactive waste, biodiversity, land use, safety and security and the recycling of steels and other wastes. The process is a development of the Vattenfall studies in Sweden discussed by other authors and the CO_2 emissions are a useful comparison with those from the international nuclear fuel cycle LCAs discussed.

The choice of which enrichment process, or mix of process, used in the LCAs is an important consideration and hence is discussed in more detail below. Some general observations of the individual studies are also now discussed.

Energy Requirements for the nuclear fuel cycle

Table 4-1 compares generic estimates of the total energy requirement for different components of the nuclear fuel cycle, for two types of enrichment process: Gas diffusion and gas centrifuge, used to produce enriched uranium hexafluoride (UF_6) to make uranium dioxide ($U0_2$) ceramic pellets for fuel fabrication (WNA 2012, 2014).

Table 4-1 Representative energy requirements for nuclear fuel cycle stages for a 1000MW (Electrical) nuclear power plant

Nuclear Fuel Cycle Components	**WNA 2012[9] Gas Diffusion Cycle Energy PJ**	**WNA 2014[9] Gas Centrifuge Cycle Energy PJ**
Mining (Australia[1])	2 (1.2%)	2.51 (14.8%)
Conversion (USA[2])	9.24 (5.3%)	9.24 (17.7%)
Initial Enrichment (diffusion USA[3])	5.18 (3.0%)	Not applicable
Initial Enrichment (centrifuge UK[4])	Not applicable	0.11 (0.2%)
Reload Enrichment (diffusion)	119 (68.8%)	Not applicable
Reload Enrichment (centrifuge UK)	Not applicable	2.48 (4.7%)
Fuel Fabrications (USA[5])	5.76 (3.3%)	5.76 (11%)
Construction and Operation (USA[6])	24.69 (14.3%)	24.69 (47.2%)
Fuel storage (USA)& Waste Storage and Transport (Sweden[7])	1.5 (0.9%)	1.5 (2.8%)
Decommissioning (Canada[8])	6.0 (3.4%)	6.0 (11.5%)
Total	173 (100%)	52.3 (100%)

Assuming the plant is operating at 86% capacity for 40 years generating about 7 TWh/year and a total output of 3024PJ (WNA 2012 and 2014). The estimates in the table are generated from percentages data from several locations and over a large time period (WNA 2012 and 2014). The other contributing sources for each stage are: 1) uranium from mines in Namibia, Niger or Russian Federation, 2) conversion in Canada, 3) diffusion in France, 4) centrifuge enrichment in Russian Federation, 5) fuel fabrication in France and Russian Federation, 6) construction and operation will be country dependent worldwide, 7) waste storage and transport based primarily on Swedish data, 8) decommissioning data is based on Bruce A&B, Darlington and Pickering nuclear power stations in Canada, but primarily Pickering in Ontario, 9) The data estimates are from reports dating back to 1975 for site data between 2000 and 2008.

Table 4-1 shows that for fuel made entirely using the gas diffusion process the enrichment energy is ~72% of the total energy. However, the enrichment energy is only ~5% of the total energy for fuel made entirely from the newer gas centrifuge process. Further, the total energy for the gas diffusion nuclear fuel cycle is about treble the total energy needed for a gas centrifuge nuclear fuel cycle. The data in the table are from mixed sources over a long time horizon and can best be regarded as representing an indicative set of energy requirements for the nuclear fuel cycle. It is

also not clear if the energy consumption for transporting raw material and manufactured items between countries is considered in these estimates.

New nuclear fuel is usually made from a mixture of enriched uranium derived from both gas diffusion and gas centrifuge enrichment processes (WNA 2012 and 2014, Hondo 2005). The estimated breakdown of energy demands for the nuclear fuel cycle using data for the Forsmark nuclear power station in Sweden is shown in Table 4-2.

It is noted that the gas diffusion process contribution has reduce since the shutdown of the Georges Besse 1 plant in 2012, hence a major element of enrichment energy differences and uncertainty have now been removed (Butler 2015 pers. comm.).

Table 4-2 Energy requirements for nuclear fuel cycle stages for the Forsmark 1000MW (Electrical) nuclear power plant

Nuclear Fuel Cycle Components	**Forsmark Nuclear Power Plant Energy PJ**
Mining (Namibia, Australia and Russian Federation)	5.2 (12.7%)
Conversion (Not specified)	4.1 (9.4%)
Enrichment (France (diffusion), UK and Russian Federation (centrifuge)	23.1 (53.2%)
Fuel Fabrications (not specified)	1.2 (2.5%)
Plant Operation (Sweden)	1.1 (2.5%)
Build and decommission the plant (Sweden)	4.1 (9.3%)
Waste management (Sweden)	4.3 (9.9%)
Total	43.4 (100%)

Assuming it generates for 40 years at about 7.47 TWh/year and a total output of 3226PJ (299 TWh) (WNA 2012 and 2014).

The Forsmark fuel is the result of 20% gas diffusion and 80% gas centrifuge processing resulting in the enrichment energy being 53% of the total energy (WNA 2012 and 2014). In comparison, the CO_2 emissions for the once through fuel cycle and the spent fuel reprocessing cycle for fuel, based on a 90%:10% split of gas diffusion and gas centrifuge production in Japan, are presented in Table 4-3 (Hondo 2005).

The results in Table 4-3 show that the Japanese CO_2 emissions are also dominated by uranium enrichment. An estimated 56% of the total nuclear fuel cycle CO_2 emissions arising from spent fuel processing are attributed to enrichment and 72% for once through fuel (Hondo 2005). However, the difference in total CO_2 emissions for both fuel cycles is only about 8% from Hondo's research.

Table 4-3 CO_2 emissions for Nuclear Fuel Cycle Variants (from Hondo 2005, Table 6)

Nuclear Fuel Cycle Components	**Once Through Cycle g-CO_{2eq}/kWh**	**Reprocessing Cycle g-CO_{2eq}/kWh**
Mining and Milling	1.1 (4.5%)	0.9 (4.0%)
Conversion	0.2 (0.9%)	0.2 (0.9%)
Enrichment	15 (61.9%)	12.4 (55.9%)
Fuel fabrications	0.7 (2.8%)	0.6 (2.8%)
Construction	2.8 (11.7%)	3.2 (14.3%)
Generation	3.2 (13.1%)	3.2 (14.4%)
Reprocessing	0.0	0.7 (3.2%)
MOX fabrication	0.0	0.0 (0.0%)
Fuel transport	0.0 (0.2%)	0.0 (0.2%)
Spent fuel storage	0.7 (2.9%)	0.2 (1.0%)
LLW transport and disposal	0.1 (0.3%)	0.1 (0.4%)
HLW storage and disposal	0.0	0.2 (1.0%)
Decommissioning	0.4 (1.8%)	0.5 (2.0%)
Total	24.2 (100%)	22.2 (100%)

The mining and milling, enrichment, waste and decommissioning percentage CO_2 emissions in Table 4-3 are broadly comparable with the energy percentages for the gas diffusion cycle presented in Table 4-1. Also, the inclusion of spent fuel reprocessing and mixed oxide (MOX) fuel fabrication are unique to Table 4-3 but the resulting impacts are very small. Hence there are similarities of data across the three tables, but also some differences offering useful additional data in Table 4-3.

Swedish, Swiss and Japanese LCA results in Figure 4-1

The results in Figure 4-1 show that the LCAs for Sweden and Switzerland give the lowest GHG emissions. This is partially due to the power station specific data used in the Swedish example but is primarily a function of the high non-fossil fuel generation in Sweden (i.e. 51% hydro/43% nuclear) and Switzerland (i.e. 60% hydro/37% nuclear) (Fthenakis and Kim 2007). The Swiss Case 2 and the Japanese

research by Hondo 2005 in Figure 4-1 are similar because their results are linked to the GHG emissions for "… steel and concrete, which account for over 95% of the total impact" (Fthenakis and Kim 2007). The Hondo 2005 results are higher than the Swiss Case 2 in Figure 4-1 because of the additional detail for the nuclear fuel cycle previously discussed for Table 4-3. Hondo combined process LCA for material impacts (e.g. metal and concrete) with economic input-output LCA for manufacture items (e.g. machinery and complex plant components) in the research and this may also have contributed to the higher results.

Australian LCA results *in Figure 4-1*

The Australian LCA results in Figure 4-1 are from a hypothetical case study performed by the Australian Coal Industry Association. The LCA investigates the sustainability of coal in iron and steel production and electricity generation (including nuclear power). The GHG emission results show the dominance of uranium enrichment, estimated to represent about 75% of the GHG emission in the Australian study. This was presumed to be from its wider scope/purpose and system boundary assumptions and the Australian energy mix but the details are unknown.

American LCA results in Figure 4-1

The American study in Figure 4-1 is rather old and has been over taken by more recent US LCAs as discussed for Figure 4-2.

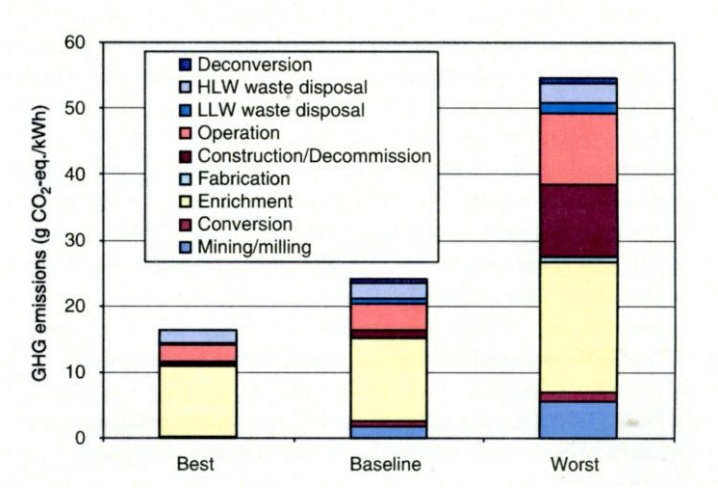

Figure 4-2 Comparison of GHG emissions for American nuclear fuel cycle **(**reproduced from Fthenakis and Kim 2007, Fig 5)

The American baseline results in Figure 4-2 assumed mining and milling of ore with an average grade of 0.2% uranium (based on data from 1987-1990). The best case results plotted in Figure 4-2 assumed a 12.7% uranium ore grade from Canada and a worst case of 0.05% uranium ore grade from Australia. The energy to extract and process ores increases significantly as the ore grade decreases (Chapman and Roberts 1983). Hence, the energy and GHG emissions for mining and milling will be significantly higher for the worst case in Figure 4-2. As a consequence, an increased impact for uranium conversion and enrichment can be expected from the reduced ore grade used (Chapman and Roberts 1983, Fthenakis and Kim 2007, Lenzen 2008, WNA 2012 and 2014). The reasons for the increased impacts from construction, operation and decommissioning for the worst case are not so clear, but may be due to increased material and energy demands.

'World' LCA results in Figure 4-1

The Storm van Leeuwen and Smith 2005 study is an extensive energy analysis, with associated estimated CO_2 burdens and financial analysis and was updated by the authors in 2007. The model assumes a reference 1000MW light water reactor (LWR) power station with an operating life varying from 20 to 40 years and a load factor varying from 82 to 85%, but was a contentious study because of the assumptions and methodology used (Fthenakis and Kim 2007, Lenzen 2008, Sovacool 2008, WNA 2012 and 2014). For example, Storm van Leeuwen and Smith assumed that the energy for construction, all supporting activities for operation (including maintenance and repair), decommissioning and waste management was based on fossil fuel power plants using diesel fuel as a proxy for coal, gas and oil fired plant, No account seems to have been made of representative energy mix for their generic model. In addition the authors assumed that all waste disposal was in concrete packages to deep geological disposal facilities and include uranium mine remediation (Sovacool 2008, Table 7) in the final decommissioning and clean up.

Further, Fthenakis and Kim 2007 note that the Storm van Leeuwen and Smith 2005 construction impacts are ~50% higher than the construction data for the Sizewell B

nuclear power station. However, Storm van Leeuwen and Smith note the Sizewell B data in their study as being within the bounds of their construction estimates (Storm van Leeuwan and Smith 2007). Beerten et al 2009 note that although Storm van Leeuwen and Smith 2005 presents data on conditioning, storage and disposal of waste, their energy estimates are "… assumed to be equal to the construction energy of the power plant per unit mass" and concluded that this was a rather unrealistic assumption leading to high results for storage and disposal. Wallbridge et al 2012a also discuss the methodology of Storm van Leeuwen and Smith 2005. They note that the decommissioning data used by Storm van Leeuwen and Smith are based on decommissioning costs data and is highly dependent on amount of waste assumed and the type of decommissioning processes used. They conclude that the costs data must have a high degree of uncertainty. All of these assumptions and modelling differences lead to the large CO_2 emissions evident for the Storm van Leeuwen and Smith results in Figure 4-1.

4.1.2 A comparison of Fthenakis and Kim 2007 with more recent studies

Table 4-4 presents a comparison of GHG emission for a more narrowly defined nuclear fuel cycle from a range of LCAs to give an understanding of the spread of the data. The Storm van Leeuwen and Smith 2007 study is presented here rather than the 2005 data presented in Figure 4-1 as it represents the authors' revised data.

The table presents the range of GHG emissions for the cross-section of LCAs considered. These results confirm the large variation of GHG estimates that was previously seen in Figure 4-1. The differences arise from the specification and scope of the LCAs - the data and assumptions used - the details of what is included in the designation of the nuclear fuel cycle stages – and political issues (e.g. Tokimatsu et al. 2006) have different political assumption looking forward in the Japanese nuclear options. These issues are also recognised in Sovacool 2008.

Table 4-4 Comparison of GHG emissions (in g-CO_2eq/kWh) of the nuclear fuel cycle for a selection of LCAs

Nuclear Fuel Cycle Stage	**Hondo 2005[1]**	**Tokimatsu et al 2006[2]**	**Fthenakis and Kim 2007[3]**	**Storm Van Leeuwen and Smith 2007[4]**	**Norgate et al 2013[5]**
Frontend	14.1-17	5.9 – 118	12 – 21.7	8.8	21.2
Construction	3.2 – 2.8	1.3 – 26	0.5 - 17.7	11.6 – 34.8	8.4
Operation	3.2	2 – 40	0.1 – 10.8	24.4	3.2
Backend	1.4 - 0.8	0.7 – 14	2.1 – 3.5	16.6	1.0
Decommissioning	0.5 – 0.4	0.1 – 2	1.3	13.2 – 46.4	0.3
Range of Emission	22.2 - 24.2	10-200	16-55	85-130	34

Notes –
1 - Summarised from Hondo 2005 and Table 4-3 for Japan
2 - Summary of Tokimatsu et al 2006 data for Japan reproduced from Sovacool 2008
3 - Summary of Fthenakis and Kim 2007 data range for United States, Europe and Japan reproduced from Sovacool 2008
4 - From Storm van Leeuwen and Smith 2007, for the Baseline case for the reference 1000MW light water reactor
5 - From Norgate et al. 2013

The Storm van Leeuwen and Smith 2007 results in Table 4-4 are again considerably higher than the other LCAs, as in Table 4-1, e.g. about 4 to 5 times higher than the total emission estimated by Hondo 2005 and data taken from Fthenakis and Kim 2007 and the more recent data from Norgate et al 2013. Also, according to Beerten et al 2009 the GHG emission for European nuclear power generation is also about 32 g-CO_{2eq}/kWh and Warner and Heath 2012 estimate the mean to between 18 and 25 g-CO_{2eq}/kWh.

The only GHG emission example close to the Storm van Leeuwen and Smith results in Table 4-4 are for Tokimatsu et al 2006. The Tokimatsu et al 2006 research presents data for four potential scenarios, ranging from the current position to the phase-out of nuclear power of the Japanese nuclear industry. Tokimatsu and his colleagues include interim storage and disposal of LLW and HLW, the containers required and the cement grouting of the waste. They also address impacts from material, energy, transport and waste. Their analysis showed that in general the life cycle impact from CO_2 emissions is dominated by uranium enrichment, power station construction and power station operation. These three stages account for between 65% and 84% of the CO_2 emissions depending on the scenario.

Reprocessing and storage of spent fuel each represented ~6% of the impact depending on the scenario. The authors also show that CO_2 emissions are high for decommissioning and cementation in their phase-out scenario but small overall in the other scenarios. Transport impacts were small in all four scenarios.

Both Hondo 2005 and Tokimatsu et al 2006 were published before the Fukushima 2011 accident and the economic and public acceptance of nuclear power in Japan has changed significantly since the accident. The analyses are valid for the time frame in which they were produced and their results provide a useful comparison with other LCAs.

4.1.3 Focus on End-of-Life phases of previous studies

The GHG impacts of the backend and decommissioning stages together of the nuclear fuel cycle in Table 4-4 range from about 4% of the total impact from Norgate 2013, 5 to 9% from Hondo 2005, 8% from Tokimatsu et al 2006, and 9-21% from the range of studies from Fthenakis and Kim 2007. This is consistent with the WNA 2012 and 2104 results discussed in Table 4-1 and Table 4-2. This is in contrast with the Storm van Leeuwen and Smith 2005 results in Figure 4-1 and the revised 2007 results presented in Table 4-4. The revised data in Storm van Leeuwen and Smith 2007 suggest that the backend and decommissioning stages together could be responsible for 35 to 48% of the total nuclear fuel cycle GHG emissions for the reasons discussed for the 'world' model in Section 4.1.1. Also, including mine remediation increases the decommissioning impacts by about 1/3 according to Sovacool 2008. Further, Lenzen 2008 notes: - most of the energy required for waste storage "… is for processing material such as concrete and steel for storage containers…" - waste management can contribute between 5 to 14% of the total life cycle impact - and "… the majority of greenhouse gases in the nuclear fuel cycle are caused by processes up steam and down steam of the [power station] plants…".

Also of interest in this thesis is the energy required for back-end process such as the long-term storage of depleted uranium, spent nuclear fuel and intermediate level waste ILW (metals). Spent fuel and waste storage was estimated as 1-3% in WNA 2012 and 2014 (Table 4-1) and Hondo 2005 (Table 4-3) but waste management in

general was ~10% for the power station specific data in Table 4-2. There is no indication that energy for waste container material and manufacture, cement encapsulation of waste for storage and disposal or the transport of the waste packages to final repositories is included in either case. Neither do the estimates appear to include the energy requirements for the construction, operation, decommissioning and dismantling of interim ILW stores nor excavation, operation and closure of near-surface or deep geological disposal facilities (GDF). Hence, there are significant gaps in the existing literature that need further investigation to give credence to energy requirements for the waste and decommissioning stages of the nuclear fuel cycle. This research investigates the container and disposal facility environmental impacts and direct costs, but does not address construction, operation and decommissioning of interim ILW stores for the UK metals inventory in Chapter 8 or the depleted uranium stores in Chapter 9.

Some of the WNA 2012 and 2014 references indicate that wider environmental and health issues need to be considered for the nuclear fuels cycle. For example Rashad and Hammad 2000 and Gagnon et al 2002 consider emission of CO_2, SO_2, NO_x, volatile organic compounds and particulates in their LCAs. Also, Schneider et al 2013a and 2013b consider CO_2, land use and water use, in addition to energy for the mining and milling at the front end of the fuel cycle. Considerable excavation will be needed for near-surface waste disposal or construction and closure of a GDF hence the additional impacts considered by Schneider and colleagues may also be significant when estimating impacts for final disposal of radioactive waste.

Fthenakis and Kim 2007 emphasise the role of country specific issues concerning uranium ore grade, energy mix and preference of the enrichment processes, they also discuss uncertainties in commonly used data. In particular they note that when analysing the nuclear fuel cycle "… greater uncertainty applies to decommissioning, a stage for which actual data do not exist".

Lessons to be learned from International Nuclear Fuel Cycle LCAs

Although the goal of individual LCAs (i.e. estimating the impacts of energy consumption and GHG emissions) may be common, the scope (e.g. the scenarios or purpose), system boundaries (i.e. which detailed components are included in each stage) and the LCA type (e.g. process LCA or economic input-output LCA) are not always the same (Fthenakis and Kim 2007, Sovacool 2008, Warner and Heath 2012). This results in large differences in the estimates for energy consumption and for GHG emissions. Overall, the LCAs in this section suggest that there are gaps in the current modelling of waste and decommissioning in the nuclear fuel cycle. The current LCAs appear to neglect the use of steel and concrete in radioactive waste disposal and it is not clear that storage and disposal environmental impacts are rigorously considered. The excavation, operation and closure of near-surface and deep geological disposal facilities could be a large impact that may be comparable to the impacts of mining and milling uranium ore. Finally, some of the more recent front end LCAs, and the metals environmental impacts discussed in Chapter 5, suggest that an assessment of a broader set of environmental impacts could be warranted, rather than just consideration of energy and GHG emissions.

4.2 American Decontamination and Decommissioning Studies using combined LCA and MCDA methodology

The previous section dealt with the energy and GHG emissions for the nine stages of the nuclear fuel cycle shown in Figure 1-1. Only end-of-life disposition options in radioactive waste management and decommissioning are of direct interest in this thesis. Hence, this section summarises American studies exploring the disposition options for radioactive waste metals, soil and concrete from decommissioning using a methodology combining LCA and multi-criteria analysis. It outlines the background to the approach, an outline of the basic methodology, the decision matric developed and some environmental and cost benefit outcomes.

Background to the American Approach

Early American research showed that treating radioactive metal for recycling or new products was feasible (e.g. Worchester et al 1993 and 1995, Murphie et al 1993, Atteridge et al 1994) and is confirmed in the detailed analysis in National Committee

on Radiation Protection & Monitoring (NCRP) 2002. The early work did not consider whether the benefits of treating the metals and fabricating products outweighed the environmental and financial costs (Yuracko et al 1997a and 1997b). Hence, a life cycle decision-making method was developed at the Oak Ridge National Laboratory (ORNL) to assess the wider costs and benefits of decontaminating and recycling radioactive wastes including metals (Yuracko et al 1997a, 1997b, 1998, 1999a, 1999b, Curlee and Yuracko 2000, Yuracko and Morris 2001, hereafter called the ORNL 1997-2001 method).

Outline of ORNL Methodology

The ORNL 1997-2001 methodology is outlined in Figure 4-3.

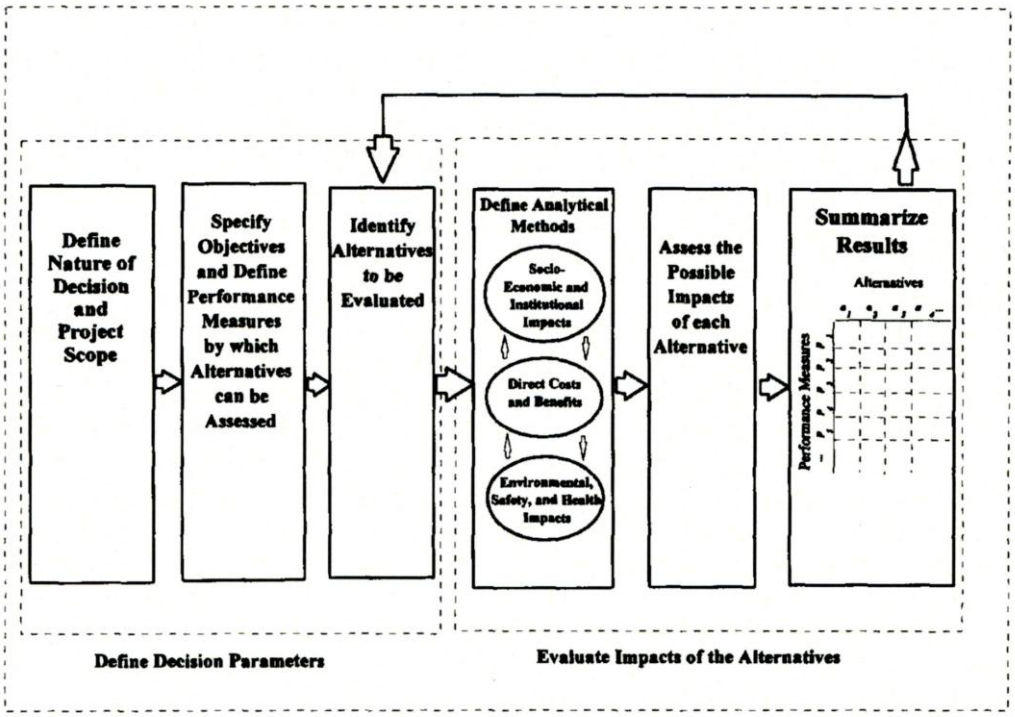

Figure 4-3 Combined LCA and MCDA Framework (reproduced from Yuracko et al 1997b, Fig 1)

The ORNL 1997-2001 methodology consisted of an LCA phase and a decision phase considering quantitative and qualitative factors to assess the net benefit, or detriment, Figure 4-4) of the processes being considered (Yuracko et al 1997a, 1997b, Yuracko et al 1998). The LCA phase assessed the potential environmental impacts of operational and programme requirements. It also defined the values and scope of specified objectives and performance measure for alternative outcomes (Yuracko et

al 1997a, 1997b, 1998). The decision phase addressed the uncertainties and value judgements for the trade-off of competing objectives by comparing and ranking the alternatives. It is very similar to the integrated MCDA methodology discussed by Belton and Stewart 2002 and Petrie et al 2007.

Typical general concerns (orange) and individual attributes (blue) for evaluating alternatives to identify a potential net benefit (green) of a decommissioning decision to be made were as shown in Figure 4-4. This represents a hierarchal approach for general sustainability objectives within decision making and attempts to address the 'triple bottom-line' of environmental, societal and economic considerations.

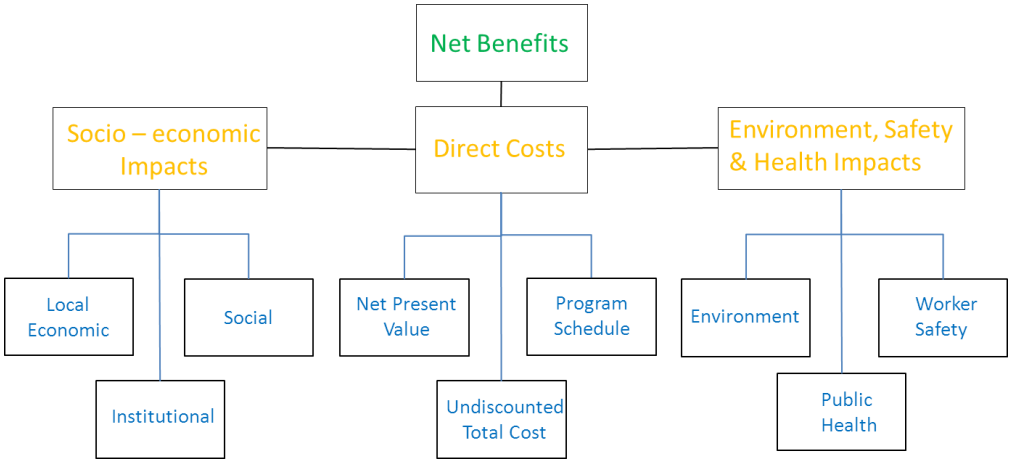

Figure 4-4 General concerns and individual attributes for radioactive scrap metal decision making (reproduced from Yuracko et al 1997b, Fig 2)

Decision Matrix

The output of the ORNL 1997-2001 method was a tabulated decision matrix of alternatives (rows) for the attributes (columns). Typically, the matrix was divided into a Direct Cost column and individual attribute columns for the Socio-economic Impacts and Environmental, Safety & Health Impacts from Figure 4-4. The Program Schedule attribute is commonly included in the decision matrix. The choice of Net Present Value or Undiscounted Total Cost will depend on the purpose of the study and the reference of the decision maker. The rows typically contained several outcomes from full disposal to full recycling. Each cell in the decision matrix contains a 5-point semi-quantified score from best to worst for each alternative and

attribute. The decision matrix was designed to present a simplified visual representation of the analysis to decision makers and for wider communication to stakeholders.

Environmental and Cost Benefit Outcomes

Unfortunately, the articles by Yuracko and her colleagues do not present any data on the environmental benefits accrued from adopting the process. However, they do state that applying their method to decommissioning projects for the gas diffusion plant at Oak Ridge in Tennessee resulted in a decision to recycle over 100,000 tons of metals and saved over 5000 truck deliveries between Tennessee and Nevada (Yuracko et al 1999a, 1999b and Curlee and Yuracko 2002). The study also saved about $80m in direct costs and generated about 2000 jobs in recycling. It is estimated that the recycling saved emissions of 34tons of particulate and 67tons of SO_2. These savings resulted in avoided damage costs of about $2.4/ton of metal recycled for particulate and $8.0/ton of metal recycled for SO_2. At the time, the recycling processes appeared preferable to the regulators and the public. Later the US government, in response to public pressure, decreed that treated radioactive metals could not be released in to the open markets but must be used within the US Department of Energy facilities.

Summary of the Combined LCA/MCDA Approach

The ORNL 1997-2001 studies are the only published international examples of applying LCA to inform decision-making in decommissioning. Whilst unfortunately environmental impact calculations are not published in these studies, they do demonstrate that LCA can be applied effectively as an environmental management tool and, within the framework of a multi-criteria decision analysis tool, can lead to improved environmental and economic decision-making.

4.3 UK Nuclear Related LCAs

This section discussed LCAs for the UK nuclear industry related to spent nuclear fuel and those related specifically to decommissioning, which is of particular interest in this research.

4.3.1 UK Spent Nuclear Fuel LCAs

Two UK nuclear industry LCAs are cited in Azapagic 1999, Griffin 1997 and Solberg-Johansen 1998. Griffin 1997 does not appear to be publically available hence cannot be discussed here. The PhD thesis by Solberg-Johansen 1998 presents the development of a LCA tool that included ionising radiation effects for assessing the potential harm to human health and the environment of emissions from nuclear power generation. However, ionising radiation impact analysis, developed by Frischknecht et al. 2000, has since been included in life cycle impact assessment methods, such as Eco-Indicator 99 and ReCiPe 2008 and is recognised as a specific impact category in EC/JRC 2011. A third LCA, Wooders et al 2007a was part of a uranium and plutonium macro-economic study for the NDA presented in Wooders et al 2007b. The LCA does not provide environmental or LCA methodology information pertinent to this research. All three LCAs relate to spent nuclear fuel, which is excluded from the research, so are not discussed further.

4.3.2 UK Decontamination and Decommissioning LCAs

LCAs for the decommissioning of Magnox power stations and UK legacy nuclear facilities in general are presented in Wallbridge et al 2012a and 2012b. These appear to be the only publications that present LCAs that are specific to decommissioning of UK legacy nuclear facilities. Their findings are discussed below.

Magnox Power Station LCA Goal and Scope

The LCAs presented in Wallbridge et al 2012a are an assessment of the potential environmental impacts of decommissioning Magnox power stations. The underpinning LCA data are bespoke, and were derived for Trawsfynydd power station and extrapolated to estimate the impacts for the entire fleet of 11 Magnox nuclear power stations. The scope of the LCAs included the decommissioning, storage and disposal of radioactive waste, after the nuclear fuel had been removed from the reactors. The spent nuclear fuel was sent to Sellafield for storage and reprocessing but those impacts were not included in the LCA. Similarly, the HLW generated from the fuel processing was excluded from the research.

System and Process Boundaries, Assumptions and Data

The Trawsfynydd LCA addressed key processes in: 1) site management, 2) research and development (R&D) needed for decommissioning, 3) ILW and LLW retrieval and 4) plant deconstruction. The deconstruction includes not only the decontamination and dismantling of the original operating plant but also the dismantling of the support facilities needed to progress decommissioning to completion. The LCA also included processes used in: 5) interim storage and disposal of ILW (including the deconstruction of the interim store), 6) LLW disposal, 7) remediation and disposal of contaminated land, and 8) transport. The authors used primary data from Trawsfynydd, plus secondary data from private communications, from the Ecoinvent database and from the open literature. The materials and energy data were used to build the inventory for Trawsfynydd. For example, nearly 3,000 tonne of steel from civil engineering works, ~28,000 tonne of LLW and ILW steel plus ~2,000 tonne of steel for the interim ILW store (Wallbridge et al 2012a, Table 4) and about 13,500 tonne of steel in final decommissioning. In addition, the authors estimate about 147,000 tonne of concrete for ILW and LLW storage and that about 820 ILW and 6,600 LLW 'single-use' waste containers were needed for disposal of the waste (Wallbridge et al 2012a, Table 4). Steel and concrete were identified as being significant for waste and decommissioning stages. The authors therefore present a comprehensive and detailed data set for the decommissioning of Trawsfynydd. They also note where data is missing and what assumptions had to be made to estimate representative values, e.g. from personal communications and engineering calculations used to estimate the energy needed to cut steels for disposal or recycling (Wallbridge et al 2012a, Table 3).

Summary of Trawsfynydd LCA Results

The Magnox LCAs were analysed using the CML2001 life cycle impact assessment method in the GaBi software packages. The output was potential environmental impact categories for human health, photochemical smog, ozone depletion, human, terrestrial, marine and freshwater ecotoxicity, global warming, acidification, eutrophication and abiotic resource (Wallbridge et al 2012 a, Fig 4 and 5). The dominant impact for Trawsfynydd was global warming at an estimated 3.5 g-

CO_{2eq}/kWh. This is higher than the mean values for decommissioning in Hondo 2005 and Fthenakis and Kim 2007, and is closest to the upper level of the Tokimatsu et al 2006 (Section 4.1). It is, however, considerably less than the results for Storm van Leeuwen and Smith 2007 in Table 4-4. The marine, human and freshwater ecotoxicity impacts estimated by Wallbridge and his colleagues can be listed as the most important impacts in conjunction with global warming. Ozone depletion, terrestrial and abiotic resource depletion impacts were of secondary importance, at about an order of magnitude lower than ecotoxicity and global warming. The other impact categories were considerably smaller. Hence the Wallbridge et al 2012a analysis shows the importance of considering a broad range of environmental impacts rather than simply the GHG emissions considered in the LCAs in Section 4.1. Wallbridge et al 2012a do note however that their broader and more detailed approach does make comparison with other LCAs an issue.

The Trawsfynydd LCA sensitivity analysis for steels showed that if 70% of the steel from decommissioning could be recycled, the impact category results reduced by between 15% and 55% (Wallbridge et al 2012a, Fig 6) with an average benefit of ~34% based on a total global warming potential impact of 3.5 gCO_2eq/kWh for decommissioning at Trawsfynydd. The authors also note that the sensitivity analysis for concrete recycling, from the deconstruction or demolition of existing plant and interim storage facilities is subject to higher uncertainty and gave benefits of only 2% to 4.6% of the total Trawsfynydd decommissioning impact. This is not a topic of interest in this research. However it does highlight the energy needed for, and the waste produced by, deconstructing large reinforced concrete structures like reactor buildings and interim ILW stores that may be missing from other LCAs. The authors also briefly discuss the steel and cement grouting requirements for the 'single-use' containers for waste disposal (Wallbridge et al 2012a). They do recognise that the impacts will reduce as the waste containers and volume reduce, but there is no detailed analysis of these aspects. Optimised waste container utilisation, hence waste volume minimisation, is a significant issue for conserving scarce disposal volume and minimising disposal costs.

The authors also note that there are problems comparing the Magnox power stations with the 1000MW nuclear power plant (normally a pressurised water reactor (PWR)) cited for the international LCAs discussed previously. PWRs generally have a longer operating life, higher operating capacity and higher output, i.e. 299 TWh for Forsmark (Table 4-2) compared to 69TWh for Trawsfynydd (Wallbridge et al 2012a). This can multiply the emission of Magnox plants by a factor of 3 or 4 when normalising impacts to kWh. Wallbridge and his colleagues also note the Trawsfynydd Magnox power station generates about 5-6 times more waste that the later generation PWR at Sizewell B. The difference arises from the physical size of the old Magnox station reactors and steam raising plant compared to a PWR. Magnox reactors are large because of the large amount of un-enriched uranium metal fuel and graphite moderator needed and their boilers are large because of the low steam temperature and quality. PWRs use 3-4% enriched uranium dioxide fuel and cooling water as the moderator hence the plant is much more compact, giving less waste for final disposal. This large waste volume will be a generic issue for all UK Magnox and Advanced Gas-cooled Reactor (AGR) power stations.

Each of the eleven environmental impact categories in the Trawsfynydd LCA of Walbridge et al 2012a has contributions from the eight key processes from site management to transport above. The contribution to each environmental impact category is dominated by plant deconstruction (ranging from 25 to 75% of the total category impact) and ILW storage & disposal (ranging from 25% to 70% of the total category impact). Wallbridge et al 2012a note that "Around 85% of the impacts from deconstruction of the plant are due to steel and concrete used to package the LLW and ILW wastes". The authors also note that construction of the interim ILW store accounted for about 90% of the ILW storage & disposal impact. The other contributions are generally minor, e.g. 3-4% for waste retrieval, LLW disposal and land remediation and 1-10% for site management, R&D and transport.

The authors further note that human health impacts are dominated by deconstruction (~75%) and ILW storage and disposal (~23%) consistent with the general trend. The authors also estimate about 25% of the health effects are from radiation for steel,

concrete, power generation and most importantly from natural radiation in the excavation of the waste repositories. Wallbridge and his colleagues used the Swiss ILW repository data in Ecoinvent 2.2 (see Dones et al 2012) as a proxy for a future UK geological disposal facility to give a first order estimate of disposal impacts. Hence, the human health issues from deconstruction and disposal are important considerations for this and other research addressing nuclear waste and decommissioning. However, it is not clear if this includes human health issues from conventional safety during decommissioning or just radiation dose.

Magnox Fleet Results and General Observations

All 11 Magnox stations have different operational lives and electrical output and different waste volumes. Given the quantities and importance of waste volumes on packaging and disposal requirements the authors chose ILW and LLW volumes to extrapolate the impacts of the entire Magnox fleet from the Trawsfynydd results. These differences have a significant effect on the environmental impacts of individual stations. The results for global warming are shown in Table 4-5. The table shows the range of generation outputs and waste volumes and the effect on the global warming potential, i.e. the lower the output and higher the waste volume, the higher the GHG emissions. This type of variation is similar for the other environmental impacts but global warming was chosen to enable direct comparison with the LCAs published by other authors discussed in Section 4.1. The variation in GHG emissions from 0.89 to 7.14 g-CO_{2eq}/kWh reflects the difference in construction, output and waste volumes from Wylfa and Chapelcross respectively. The large variation of Magnox power station electricity output, plus the variation in low and intermediate radioactive waste volumes, results in large variations in the global warming impacts. This is not similar to the variation in GHG emission across the international LCAs of the whole nuclear fuel cycle discussed previously in Section 4.1 which was the result of different modelling assumptions and data. It does, however, highlight again the need to be clear about what is modelled in LCAs in order to understand the environmental effects of waste and decommissioning for UK legacy nuclear power stations.

Table 4-5 Magnox fleet global warming impacts for decommissioning and waste disposal (taken from Wallbridge et al 2012a, Table 8)

Magnox Station	**Lifetime Output (TWh)**	**ILW Volume (m^3)**	**LLW Volume (m^3)**	**Global Warming (g-CO_2eq/kWh)**
Berkeley	43	6910	30300	2.84
Bradwell	60	5770	51400	2.69
Chalder Hall	60	9410	51000	3.14
Chapelcross	60	6230	167000	7.14
Dungeness A	115	6940	34900	1.16
Hinkley Point A	103	7270	57400	1.81
Hunterston A	74	8350	57600	2.64
Oldbury	125	6120	32900	0.98
Sizewell A	110	6140	38700	1.23
Trawsfynydd	69	13400	60700	3.5
Wylfa	225	8430	59500	0.89
Average				2.55

Note: "A" denotes the first of the power stations at that site. The second power station at these sites is an AGR (or PWRin the Case of Sizewell), however, the data for these are not presented in Wallbridge et al 2012a or 2012b.

In addition to showing the variations across the Magnox stations the authors also draw general conclusions. For example, global warming potential could be reduced by ~50% by delaying decommissioning to benefit from decarbonisation of the energy industry. This is a significant benefit and supplements the benefit of radioactive decay to reduce the volume of HAW for future disposal. Also, if spent nuclear fuel reprocessing at Sellafield is included in the decommissioning, the potential environmental impacts for Magnox stations could increase by a factor of four or five.

The detailed modelling in Wallbridge et al 2012a and the additional impacts for waste packaging and disposal, plus the potential natural radiation impacts from excavating disposal facilities are clearly important additional considerations. The addictional impacts for waste packaging and disposal are included in Chapters 7, 8

and 9. The natural radiation from excavating the disposal facilities is outside the scope of the research except where already included in the Ecoinvent database for the Swiss repositories.

Overall UK Legacy Nuclear Facilities Decommissioning LCA

The Trawsfynydd LCA showed that about 70% of the contribution to individual environmental impacts was from plant deconstruction and ILW storage and disposal (Wallbridge et al 2012a, Fig 8). The authors state that steel containers, cement grouting of waste packages and geological disposal of ILW represent about 90% of the environmental impacts. The Trawsfynydd results, plus the same basic system boundary (i.e. eight systems processes from site management to transport discussed previously), were used as a baseline for the other UK legacy reactor and non-power station nuclear facilities in Wallbridge et al 2012b.

Wallbridge et al 2012b propose that a first order approximation of the waste and decommissioning impacts for UK AGRs and the Sizewell B PWR can be achieved by multiplying their estimated waste volumes by the impacts per m^3 from Trawsfynydd. This is on the basis that decommissioning processes will not be too different across the three types of UK nuclear power stations (Wallbridge et al 2012b). The impacts of overall UK reactor decommissioning were therefore estimated according to the ratio of LLW:ILW volumes. On this basis, the authors showed that at Trawsfynydd LLW management contributed 26 to 58% to the individual environmental impacts and ILW management contributed 42 to 58% (Wallbridge et al 2012b Table 1). They also showed that the ratio of impact per m^3 of LLW:ILW ranged from 3.4 to 13.1, but on average was about 5, i.e. "…1m^3 of ILW appears to have the same impact as 5m^3 of LLW" (Wallbridge et al 2012b). Wallbridge et al 2012a and 2012b predictions show that older Magnox power stations have a global warming impact of 3-7 g-CO_{2eq}/kWh excluding Sellafield processing and 10 -20g CO_{2eq}/kWh including Sellafield reprocessing. Later Magnox stations show 1-3 g-CO_{2eq}/kWh excluding Sellafield processing and 4-7 g-CO_{2eq}/kWh including Sellafield processing. Hence the latter Magnox station GHG contribution is not too dissimilar to the 5g CO2eq/kWh for the Torness AGR

(AEAT(E) 2005) discussed in Chapter 4. Given that PWRs generally have higher outputs, longer lives and are physically smaller one would expect their global warming potential to be somewhat lower. However, Wallbridge and his colleagues do not discuss this aspect.

4.4 Chapter Summary

LCA is one of many environmental management tools and can be combined with other complementary tools. As such, it can form a valuable decision support tool for nuclear decommissioning and waste management.

A large number of LCAs have been previously carried out to assess environmental impacts of the whole nuclear fuel cycle. Individual studies do not always have consistent system boundaries and the data, modelling details and assumptions can vary widely between authors, leading to large variation in the estimates of the environmental impacts of each stage in the nuclear fuel cycle. The majority of the international LCAs focus only on energy, GHG emissions and economic costs and contain only limited data on waste management and decommissioning. However, LCA is a holistic approach and a full range of environmental impact categories can to be considered to maximise the method's potential for informing environmental management decisions.

The recent LCAs for UK nuclear facilities by Wallbridge et al 2012a and 2012b consider the environmental impacts of decommissioning, disposal and waste management in detail. These LCAs show that global warming and ecotoxicity impacts dominate in radioactive waste management and decommissioning, followed by ozone depletion and resource depletion. The majority of their calculated impacts are related to plant deconstruction and to LLW and ILW disposal, with 'single-use' steel containers, the cement grouting of waste in the containers and their disposal all contributing significantly to the total impact, especially for ILW.

5 Metals: An Important Resource

Chapters 1 and 2 outlined metals as an important resource used extensively in the UK nuclear industry and constituting a significant proportion of UK radioactive waste. They are also the main package material for the disposal of all levels of radioactive waste. Chapter 4 showed the importance of life cycle assessments (LCAs) from material extraction, through production, to use and final disposal within the nuclear fuel cycle. Although a number of LCAs discussed in Chapter 4 concentrate on energy and carbon dioxide (CO_2) emissions some also show that there are a number of other adverse environmental impacts affecting human health, ecosystem quality and resource depletion.

This research is concerned with the end-of-life disposition options for UK radioactive metals. Hence, it is necessary to understand the potential environmental benefits of recycling or re-using these metals by comparison to the environmental impacts of virgin metals extraction and manufacture (i.e. production) in general. This chapter therefore addresses five questions for metals - 1) Why are metals important? - 2) What are the significant adverse environmental effects of metals production? - 3) How are metals demands changing? - 4) Why is metals recycling important? - 5) How can we do better? The focus is on iron, steel and aluminium, but copper, zinc, lead and nickel are also briefly discussed. The chapter also presents a summary of key issues for the treatment of the radioactive metals.

5.1 Why are metals Important?

This section discusses ore grade scarcity and depletion and the economic value of metals.

5.1.1 Scarcity, Ore Grade and Depletion

Clark and Washington are credited with initial estimation of the proportion of chemicals in the outer 10 miles of the Earth's crust in 1924 allowing estimates to be made of percentages of minerals available near the surface (Alexander and Street 1976). Lists of "crustal abundance of common elements and metals" are presented in

Alexander and Street 1976 and Chapman and Roberts 1983 and in van Vuuren et al 1999 who discuss the percentage content of metals within ores. A compilation of data from; Chapman and Roberts 1983, Ayres 1997, Stewart and Petrie 2006, Ashby and Jones 2006, Bloomberg Finance, Infomine and Indexmundi websites, is presented in Table 5-1.

Table 5-1 Example Data for Metals of Interest in the Nuclear Fuel Cycle

Metal	**Percentage in the crust (%)[1]**	**Metal content in ores (%)[2]**	**Global production (metric tonne)[3]**	**Relative price per te[4]**	**Metal prices July 2010 (£/tonne)[5]**
Aluminium	8.0	19	2.3E7	300-400	1290
Copper	0.0058	0.4	1.3E7	400	4374
Iron	5.8	52	9.9E8	70 – 90	246[7]
Lead	0.001	6.5	3.0E6	200-250	530
Magnesium	2.8	33	6.0-8.0E5[6]	1000	Unknown
Nickel	0.0072	0.7	1.1E6	20000	12,610
Uranium	0.00016	0.002	3.1E4	Unspecified	58,000
Zinc	0.0082	3.2	7.6E6	350-400	1210
Carbon steel	N/A	N/A	N/A	100	480[7]
Stainless steel	N/A	N/A	N/A	600	2120[7]

Notes: These Data were taken from 1) Chapman and Roberts 1983 2) Ayres 1997 3) Stewart and Petrie 2006 4) Ashby and Jones 2006, 5) Bloomberg Finance and Infomine, 6) Indexmundi (from 2007 to 2011), 7) prices from www.worldsteel, www.steelonthenet and www.meps.co.uk, N/A – Not Applicable. The prices in column 6 are average prices from the graphs in Appendix B converted to £ Sterling and inflated to 2103 prices using a Bank of England standard annual inflation rate.

Although the total predicted quantity of metals is extremely large it is none-the-less a finite resource. Column 2 of Table 5-1 gives the percentage of metals in the crust and shows that iron and aluminium are relatively abundant, but copper, lead and nickel are of medium scarcity and uranium is scarce. This is also reflected in van Vuuren et al 1999.

Further, the quality of ores is decreasing as Ayers 1997 notes, "… ore grades are gradually declining worldwide as high grade deposits are exhausted. In the 19th century copper was being mined from deposits with 10% ore grade. Today the world average is about 0.9%. Much the same situation applies to a number of other non-ferrous metals, including gold, silver, uranium and tin". This is also reflected in van Vuuren et al 1999, Yellishetty et al 2011a and International Council on Mining and Metals (ICMM) 2012. Hence, there are concerns about the potential depletion of even relatively abundant metals and there is a need to conserve these resources

(Azapagic 1999 and 2004, Steen 2006, Ashby 2009, Yellishetty et al 2011a, Allwood et al 2010 and 2011, Allwoood and Cullen 2012). To help conserve metals their intrinsic property of theoretical infinite reusability and has resulted in a thriving metals recycling industry (Phylipsen et al 2002, Dubreuil et al 2010, Yellishetty et al 2011b, Allwood and Cullen 2012, ICMM 2012 plus British Metal Recycling Association and SteelConstruction websites).

5.1.2 Economic Value of Metals

Metals are traded in international markets and their values fluctuate markedly with time as supply and demand changes with economic and political changes. Column 5 of Table 5-1 gives and "approximate relative price per tonne" for a selection of minerals and metals against the US dollar value of mild steel ($100/tonne) (from Ashby and Jones 2006). Table 5-1, column 6 gives a snapshot of market prices for metals used in the nuclear fuel cycle at mid 2010 using data and market values from Bloomberg Finance, Infomine and steel price websites where available.

A comparison of the scarcity data in columns 2 and 3 plus the price data in the final two columns shows that the value of the metals depends both on the estimated crustal abundance and the percentage content of the mineral bearing ores. The value of metals also depends on their demand and their use, hence global production levels are also included for comparison in column 4 of Table 5-1.

The high market value of metals makes their extraction, concentration, refining and manufacturing important elements of national economies, especially in developing and transition countries e.g. Brazil, Russia, India and China - BRIC countries where large ore deposits are located and/or exploited. Investment from major international mining and metal producing companies creates jobs (mostly unskilled and semi-skilled) to local populations and helps to improve countries' Gross Domestic Product (Yellishetty et al 2011a). Production and consumption are discussed in more detail below.

5.2 What are the significant adverse environmental impacts of metals production?

This section discusses a generic metal cylce, energy and CO_2 emission from production, general emissions and wastes as significant environmental impacts for metals production.

5.2.1 The Metal Cycle

The metal cycle varies for each metal due to the production and manufacturing processes and product use. Several representations of the minerals and metal cycle exist (e.g. Ayer 1997, Azapagic 2004, Stewart and Petrie 2006, Norgate et al 2007 and Dubreuil et al 2010). The metal cycle presented in Stewart and Petrie 2006 is reproduced in Figure 5-1 as a generic cycle to aid the discussion of the environmental issues from extraction to disposal.

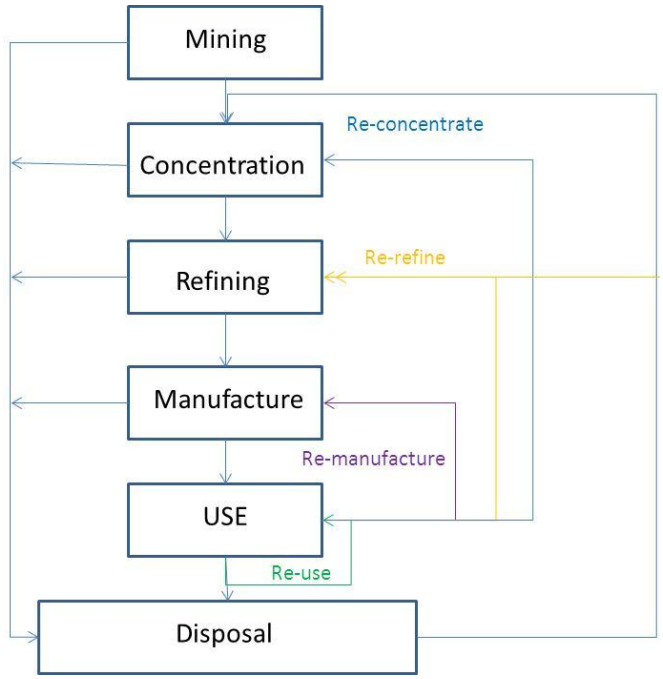

Figure 5-1 A Generic Metal Life Cycle Diagram (from Stewart and Petrie 2006)

The ore extracted in the mining phase is normally crushed in mills for physical and chemical processing near the ore source. Many ores will be low grade with impurities that would contaminate the final products. For example, elements such as "…phosphorous, alumina, silica and other problematic elements" need to be removed from iron ores (Wang et al 2007). Benefication reduces the particle size to

allow separation of mineral and waste. The minerals are then concentrated and refined to produce metals for the manufacture industry. The benefication and concentration phases result in high waste volumes. They also generate significant quantities of liquid and gaseous wastes. All these wastes may contain some elements that are toxic (e.g. Ayres 1997, Phylipsen et al 2002, Allwood and Cullen 2012). However, a proportion of this material will contain significant residual quantities of metals and other minerals. These are kept as tailings for reprocessing rather than regarded as waste. This is particularly important for scarce, high value resources like uranium, as discussed later.

Refining includes the pre-treatment of ores by roasting in air or oxygen (sintering), producing ore pellets or fine particles, dissolution in aqueous chemicals (leaching), as in the Bayer process for extracting alumina from bauxite (Norgate et al 2007), and the roasting of coal to produce coke for ore reduction in furnaces. Primary metal production from virgin ores is by smelting in blast furnaces, basic-oxygen furnace and electric arc furnaces for iron, steel and lead. Primary production can also be a combination of electro-winning by electrolysis for aluminium (via a Hall-Heroult cell ((Phylipsen et al 2002, Allwood and Cullen 2012), copper, zinc and nickel (Norgate et al 2007) and smelting in electrical or fossil fuelled furnaces (Ecobalance inc 2000, Phylipsen et al 2002, European Monitoring and Evaluation Programme/ European Environment Agency (EMEP/EEA) 2013). Limestone and fluxes are added to the furnaces for primary and secondary production to aid the chemical reaction of smelting and to transfer impurities to the melt slag. Alloying elements (such as chromium, molybdenum, nickel and zinc for steelmaking) are added to the furnace charge to produce the desired grade of final metal ingot. Secondary production is primarily from smelting metal scrap in electrical or fossil fuelled furnaces. However about 5% of secondary steel production (Intergovernmental Panel on Climate Change (IPCC) 2007, p460-461) is from direct reduction iron ore using natural gas (methane (CH_4)) (Worrell et al 1997 Fig 1, Phylipsen et al 2002, Birat and Hanrot 2006, Johnson et al 2007, Yellishetty et al 2011b, Allwood and Cullen 2012).

Metals refining uses a considerable amount of fossil fuels and electrical energy as well as producing substantial gaseous, liquid and solid waste with potentially significant environmental effects. However, as Figure 5-1 shows there are opportunities to re-concentrate, re-refine, re-manufacture or re-use metals as well as recycling them back into the manufacturing phase at the end-of-life. These processes retain metals in the cycle rather than losing them to disposal and reduce the environmental impacts when compared to using virgin material.

5.2.2 *Energy and CO_2 Emissions from Production*

The extraction, concentration and refining of metals is energy intensive (Birat and Hanrot 2006, Johnson et al 2007, Ashby 2009, Allwood and Cullen 2012). For example, the stated energy required to produce primary crude steel is 18-25 GJ/ton, mainly due to the blast furnace requirement of 12-15 GJ/ton for pig iron production (Phylipsen et al 2002). Phylipsen and his colleagues also estimate that the energy required for secondary crude steel is 8-13GJ/ton, mainly due to the electric arc furnace. These figures are confirmed in Yellishetty et al 2011b with 9-12.5GJ/tonne of crude steel via an electric arc furnace and 28-31GJ/tonne of crude steel via the blast furnace/basic-oxygen furnace route. On the other hand, Johnson et al 2007 estimate a total of ~53GJ/tonne to make austenitic stainless steel assuming the current operational regime. However, the estimates by Johnson and his colleagues include energy for iron, nickel and chromium ore mining, alloying material production, transport and final steelmaking. Alloy production and final steelmaking dominate the energy consumption in the research of Johnson et al 2007.

No energy intensity figures are given for cast iron or zinc production in Phylipsen et al 2002, but between 47-60GJ/tonne is quoted for primary aluminium. Secondary aluminium production uses about 5% of the primary production energy (Phylipsen et al 2002, IPCC 2007). Phylipsen and his colleagues further estimate about 120GJ/tonne for the electrolysis method for copper production, which is dominated by ~42GJ/tonne from electricity and ~56GJ/tonne from oil, the gas coal and steam contributions are significantly lower. The authors also note that the energy required to produce secondary copper is much lower than primary production, but they do not supply an estimate. No energy intensity data are available for lead from Department

of the Environment (DOE) 1995 or EMEP/EEA 2013. Energy to produce nickel matte was estimated as 25- 65 GJ/tonne nickel (for ores containing 4-15% nickel) and 17-20GJ/tonne nickel for the refining stage (EMEP/EEA 2013).

GHG Emissions

The environmental impacts of the energy used are not restricted to just the amount used in each process but also the energy mix of the supply and the amount of fossil fuels used. These aspects affect the amount of CO_2 and GHG emissions from the power stations supplying electricity and using coal, coke, charcoal and oil in the metal processes. For example, on a global average basis each tonne of austenitic stainless steel produced generates an estimated 3.6 tonne of CO_2 (Johnson et al 2007) and each tonne of primary aluminium produced about 1.55 tonne of CO_2 (IPCC 2007).

Anthropogenic GHG emissions are a significant global concern. The iron and steel industry constitute 10-15% of the annual general industrial energy consumption and contributes ~7% of the global anthropogenic CO_2 emissions (OECD/IEA 2000). Total global GHG emissions increased by ~70% between 1970 and 2004, and 57% of all CO_2 emissions came from fossil fuel use (IPCC 2007). The energy industry, general industry and transport make up an estimated 58.4% of total GHG emissions IPCC 2007). About 72% global energy demand is from the iron and steel, non-ferrous metal, minerals (cement, lime, glass ceramics) and chemical and fertilisers industries (IPCC 2007). Each tonne of primary aluminium production generates about 1.55 tonne of CO_2 (IPCC 2007) and the report notes the industry's drive to mitigate the high energy use and GHG emissions. IPCC 2007, however, says nothing about the climate change effects specifically associated with the production of copper, lead, nickel or zinc.

5.2.3 General Emissions

Metals production has other significant environmental impacts in addition to GHG emission, these are summarised in Table 5-2.

Table 5-2 Summary of significant emission from metals common in the UK nuclear industry *(*DOE 1995, EMEP/EEA 2013, Phylipsen et al 2002, Allwood and Cullen 2012, Wang and Liu 2012)

Metal	**Significant Environmental Impacts**
Iron and Steel	CO, CO_2, hydrogen (H_2), nitrogen (N_2)), sulphur oxides (SO_2 and SO_x) Nitrous Oxide (NO_2 and NO_x) and particulate matter (PM), i.e. dust. Other emission such as CH_4, volatile organic compounds (VOCs) etc are small in comparison. Coking and sintering plants release significant emission to the air (Ayres 1997, Phylipsen et al 2002).
Aluminium	Red mud from the Bayer process (Ayres 1997, Ayres et al 2001, Allwood and Cullen 2012), Polyfluorinated hydrocarbon and fluorides from electrolysis and solid waste from the Hall-Heroult cell. Also dust and dioxins (carcinogenic chlorinated organic compounds) from inefficient secondary furnaces
Copper	Dust, SO_2, copper, lead, arsenic and suspended solid. Dioxins are a significant pollutant from secondary copper production (European Dioxin Inventory 2009).
Zinc	SO_2, NO_x, CO, ammonia (NH_3), zinc and cadmium dust
Lead	SO_2, NO_x, CO and CO_2 are the most important emission to air. The lead, other heavy metals and dust are most important for the process emissions
Nickel	SO_2, NH_3 and hydrogen sulphide, nickel carbonyl from the refining process is highly toxic, PM_{20} dust is also an issue

The processes discussed previously also result in the production of other GHGs such as CH_4, Chlorofluorocarbons (CFCs) and Hydorchlorofluorocarbons (HCFCs) in addition to CO_2, which also contribute to climate change (Phylipsen et al 2002). The CO, SO_2 and SO_x, NO_2 and NO_x, NH_3, CH_4, VOCs and dusts (Phylipsen et al 2002, Yellishetty et al 2011b) resulting in respirator effects, ozone depletion and acidification. Liquid discharges release additional nutrients to water courses causing eutrophication and can deposit heavy metals (lead, cadmium, chromium, nickel etc) to soils potentially causing an ecotoxicity threat to humans, animals and plants.

5.2.4 Wastes

Significant amounts of gaseous and solid wastes are produce in each of the stages of the metal cycle. The mining and processing metal ores produces large quantities of solid waste (Ayer 1997, OECD/IEA 2000 and IPCC 2007). In particular, Ayers 1997 notes, “To produce a ton of pure metal, on the (world) average, it is necessary to process 22 tons of zinc ore, 30 tons of lead ore, 45 tons of nickel ore, 110 tons of

copper ore, 50,000 tons of uranium …". The waste fraction from ore benefication, normally left near the original mines, is known as gangue and is a significant issue in its own right. As a graphical illustration of the waste levels generated in metal production, the relative volumes of waste produced for different processes in the uranium fuel cycle for a pressurised water reactor (PWR) is presented in Figure 5-2.

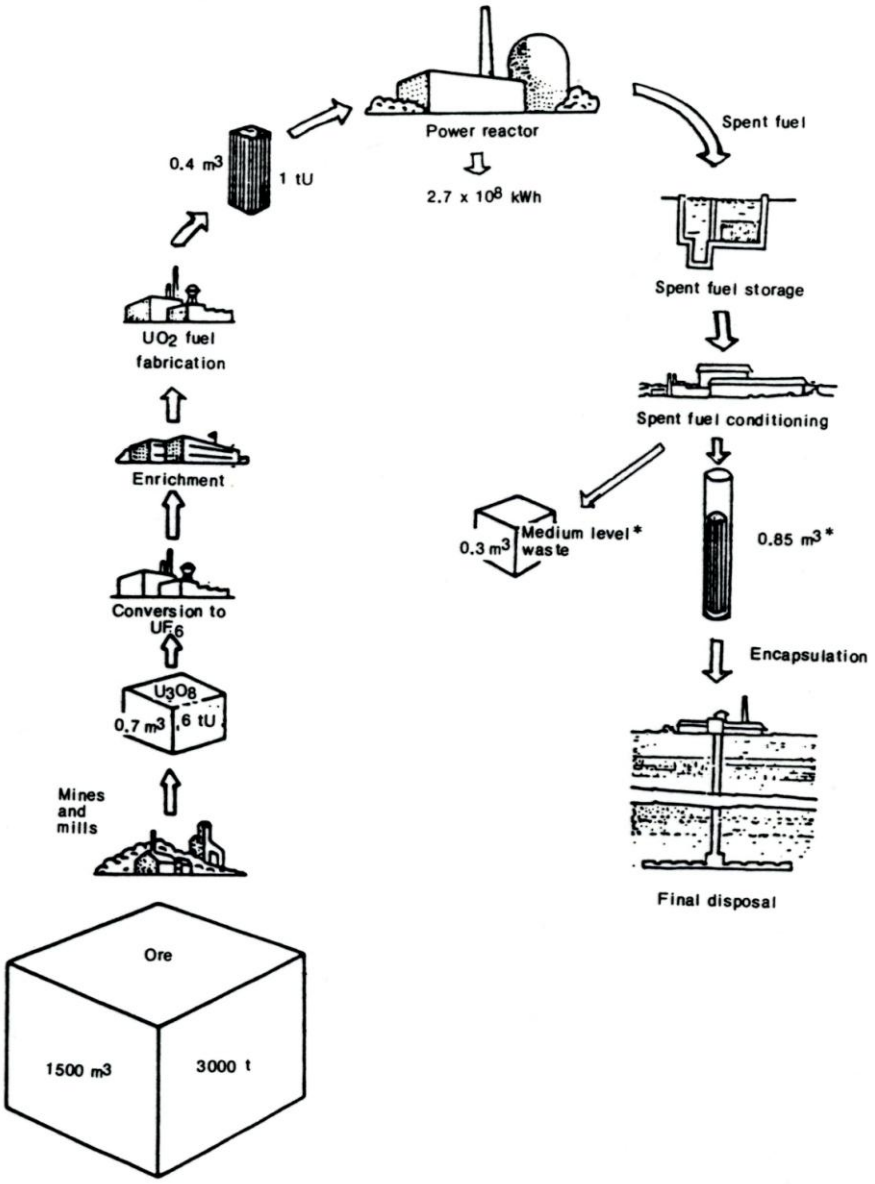

Figure 5-2 Once through Uranium fuel cycle for PWR fuel (from INFCE 1980). The figure is rather old and modern PWRs are closer to $4.2x10^8$ kWh (50GWd/teU) and have higher enrichment and natural uranium feed (i.e. 9teU rather than 6teU) (Butler 2015 pers. comm.)

The figure shows an estimated 3000tons of ore are needed to produce 6tons of uranium concentrate (U_3O_8), commonly called "yellow cake", for conversion to

uranium hexafluoride (UF_6) prior to enrichment. It then shows fabrication of 1ton of uranium dioxide (UO_2) of PWR fuel. The values are different to Ayers 1997 because they are for uranium concentrate rather than a ton of pure uranium metal.

Similarly large amounts of the waste arise from the mining and processing of the other metal ores leading to large land use and landscape degradation (Ayres et al 2001, Phylipsen et al 2002, Ashby 2009, Allwood and Cullen 2012). These wastes can have other significant environmental impacts in their own right. For example, ~30 million tons per year of dry 'Red Mud' waste is produced from the Bayer process for alumina production (Ayres et al 2001). It is made up of unprocessed alumina, iron oxide, titanium oxide, trace elements of other metals and radioactive elements. It is considered to be a significant environmental issue for the mining industry because of the toxicity, the high alkalinity and the storage/drying area requirements (Ayres 1997, Ayres et al 2001, Allwood and Cullen 2012, Wang and Liu 2012).

Gaseous Emissions and Solid Waste Potential Uses

Although there are large gaseous emissions and solid waste arising from the above processes some of the material can be put to beneficial use (Ashby 2009, Allwood and Cullen 2012). For example, SO_2 from the ore roasting for lead and zinc production is commonly converted to sulphuric acid rather than being release to air (Phylipsen et al 2002) and SO_2 from fossil fuel power station can be processed through flue gas desulphurisation plant to produce gypsum for the construction industry (Ayres et al 2001). Similarly, blast furnace slag from the iron and steel industry and fly ash/flue ash from burning coal are used as substitutes for clinker in cement production (Ayres et al 2001, Phylipsen et al 2002, Allwood and Cullen 2012).

5.3 How are metals demands changing?

This section discusses the production rates for key metals and the environmental implications of increased demand.

5.3.1 Production Rates

Global

The annual global production of minerals and metals varies considerably, ranging for example from 2.8E+2tonne/y for platinum, 3.1E+4tonne/y for uranium to 9.9E+8tonne/y for iron ore circa 2006 (Stewart and Petrie 2006). The global production of other ferrous and non-ferrous metals is commonly in the range of millions of tonne/year (Stewart and Petrie 2006). The transport of extracted and concentrated minerals and metals to the refining and production facilities can be trans-global and can add to the significant adverse environmental impacts already discussed.

The global consumption of metals is growing as the World's population increases. This is clear from the world production of iron ore and alloying metals for steel discussed by Yellishetty et al 2011a. The authors present data from 1950 to 2010 (presented here in Figure 5-3 and Figure 5-4) for metals production, whilst varying annually for each metal in response to demand has a general underlying upward trend with time for all metals.

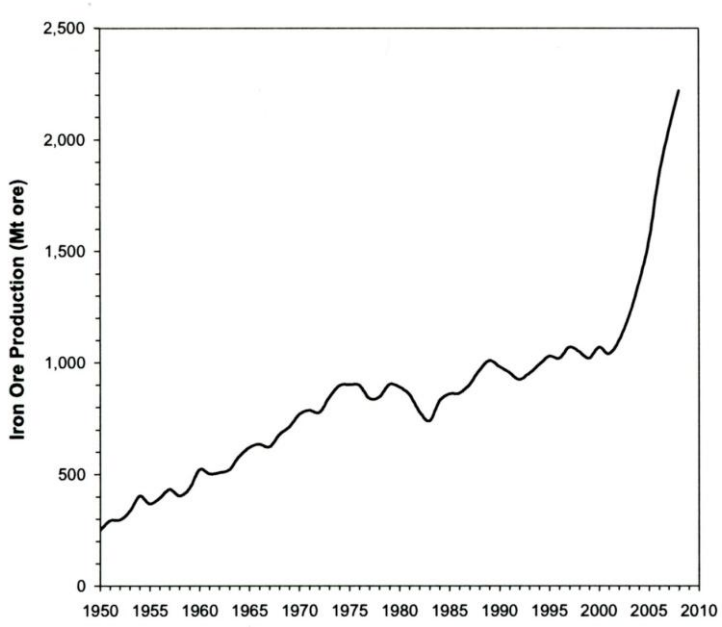

Figure 5-3 Global iron ore production from 1950 to 2010 (reproduced from Yellishetty et al 2011a, Figure 1)

The graphs in both figures show a steep increase from the early to mid1990s giving almost exponential growth of some metals and a more linear growth over the

preceding 60 year period. This is primarily due to the recent industrialisation of the BRIC countries. A consequence of this increased demand is the increasing adverse environmental impacts of metal production and use.

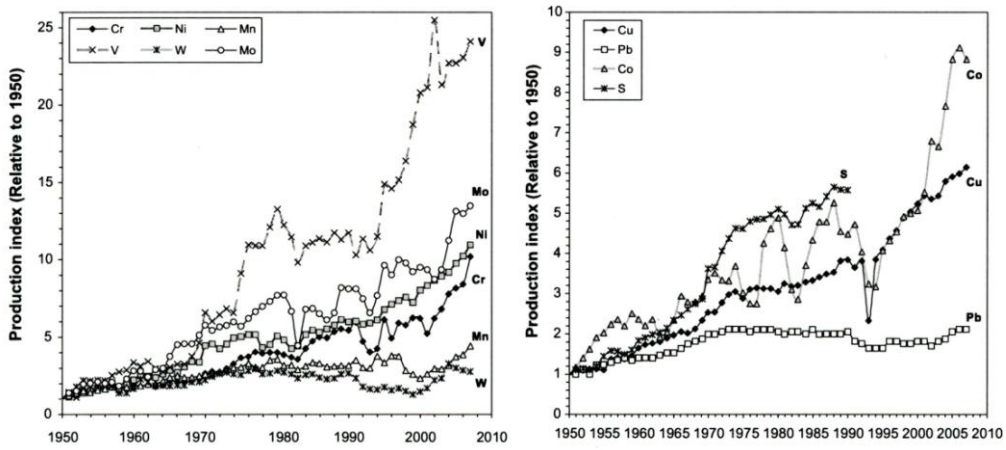

Figure 5-4 Production trends for steel alloying metals from 1950 to 2010 using 1950 as a baseline = 1 (reproduced from Yellishetty et al 2011a, Figure 2), where Co- Cobalt, Cr – Chromium, Cu – Copper, Mn- Manganese, Mo – Molybdenum, Ni – Nickel, Pb – Lead, S – Sulphur, V – Vanadium, W – Tungsten.

European and UK Examples

IPCC 2007 notes that global steel production has increased by 84% and aluminium by 223% since 1970. However, this trend is regional and country dependent. For example, Phylipsen et al 2002 predict stagnation in iron and steel in Western Europe but a growth in aluminium between 2000 and 2030. UK steel production is shown to be fluctuating but generally decreasing by Geyer et al 2007 and the Iron and Steel Statistics Bureau. The UK crude production data from 1998 to 2013 is shown in Figure 5-5. Both basic oxygen and electric arc steel production show a general downward trend since 1998. Basic oxygen steel production halved to about 7 million tonnes per year between 1998 and 2011 but recovered to ~10Mte/y in 2013. Electric arc production has dropped steadily from ~4Mte/ in 1998 to ~2Mte/y in 2013.

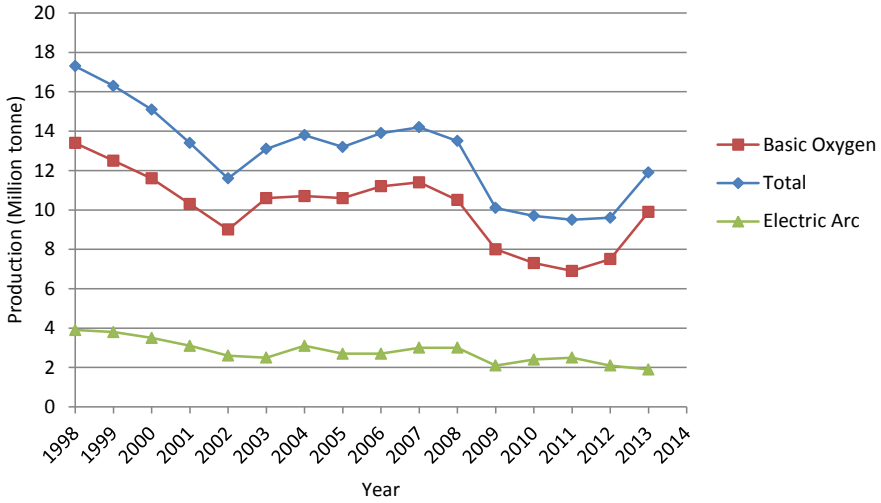

Figure 5-5 UK crude steel production 1998 – 2013 (from data presented on the Iron and Steel Statistics Bureau website (3/10/2014))

5.3.2 Environmental implications of Increased Demand

The predicted global increase in demand presented above is supported in Allwood et al 2010 and 2011, Allwood and Cullen 2012 and Ashby 2009. This implies increasing energy consumption, increasing solid, liquid and gaseous waste production, increasing landfill requirements, all of which contribute to increasing adverse environmental impacts. Allwood and his colleagues and Ashby also discuss the pressure to cut industrial carbon emissions by 50% by 2050, thus driving down the environmental impacts of metals production. To accomplish these requirements means increasing end-of-pipe abatement technologies, 'waste mining', recycling and/or re-use, improving material efficiency and waste minimisation. It also means trying to change waste producer and consumer behaviours in these areas.

5.4 Why is metals recycling important?

This section briefly discusses the end-of-life options for metals and then looks in more detail at iron and steel then aluminium and other metals.

There are five options for end-of-life products; disposal to landfill, combustion for heat recovery, recycling, refurbishment and re-use (Ashby 2009). However, to be viable Ashby suggests two criteria to test option viability:

1. "It can return waste materials into the supply chain"

2. “It can do so at a rate that, potentially, is comparable with that at which waste is generated”.

The author concludes that neither landfill nor combustion meet criterion 1, and refurbishment and re-use do not meet criterion 2 in most circumstances. Hence, only recycling may meet both criteria. However, Johnson et al 2007, Yellishetty et al 201lb, Allwood and Cullen 2012 and others note that metals recycling, although increasing along with production cannot meet production demands alone. Hence, metals still have to be produced from virgin materials. This is primarily due to the long lifetime of metals in products ranging from several years to several decades. Hence this section focuses on metals recycling. It would also be true for increased useage as the associated residence times would also increase (Butler 2015 pers. comm.).

As shown in the generic metal cycle in Figure 5-1 there are several opportunities in the metal cycle to re-use, re-manufacture, re-refine and re-concentrate metals throughout their life cycle. The reported benefits of recycling metals vary with the metal and the processes used for recycling, but in general are substantial.

5.4.1 Iron and Steel

Iron and steel are produced, used and recycled in large quantities each year globally. They also constitute a significant proportion of the materials used in the UK nuclear fuel cycle and in the associated radioactive waste inventory. Global steel production was estimated as 1.5 billion tonne in 2011 using 500 million tonne of scrap (33.3% recycling) according to www.SteelConstruction.info 2015. Similarly, the British Metal Recycling Association (BMRA) estimate that 42% of new steel was made from recycled steel, with an energy saving of 62-67% (www.recyclemetals.org 2015) and Ashby 2009 gives a recycling fraction of 35 - 44% for steels and 60-80% for cast iron.

In general “Each ton of iron recycled saves 12.5 tons of overburden (coal and iron mining), 2.8tons of iron ore, 0.8 tons of coal (exclusive of its use as fuel), and a variety of other inputs. It also estimates at least a ton of CO_2 pollution and significant additional pollution of air and water from coking, pickling and other

associated activities."(Ayres1997). The author also notes that savings from non-ferrous metals will be larger but depend on the quality of the original ore and its impurities. Examples of the benefits of recycling steel from other published data are summarised in Table 5-3. The table shows general estimates of the materials needed for iron and steel making and the recorded benefits of recycling. The data present a compilation of results from several published studies and show a range of savings.

Table 5-3 Summary of published benefits from general steel recycling

Requirements and savings	**Requirements for 1 tonne of steel**
Iron Ore needed	1.5 tonne[1,2]
Coal needed	0.5 tonne[1,2]
Coke needed	375kg/tonne of pig iron[8]
Solid Waste generated	1.3 tonne [1], 1.28 tonne [2]
	Benefits from recycling
Raw materials saved (mostly coal)	90% [5,6]
Mining waste saved	97% [5]
Water saved	40% [2,5,6] - 60% [1]
Water pollution reduction	76% [2,3,5] - 80% [6]
Emissions (Air presumed)	86% [1,2] – 90% [6]
Energy savings	70% [6] – 75% [1,2]
Health risk reduction	50% [6]

Note: 1) Robinson, 2011, 2) Waste Watch, 2011, 3) Corus Group, 2011, 4) Nieves and Chen 1995, 5) Lund, 1993, The 2000 revision of the McGraw-Hill Recycling Handbook updates the data and states that "Every time a ton of steel cans is recycled 2500 pounds of iron ore, 1400 pounds of coal and 120 pounds of limestone are preserved" and "It is about 75% less energy intensive to make new steel from recycled steel rather than start with iron ore", 6) Adams et al. 1999, 7) Alexander and Street 1976 notes that "…600million tons of steel…. requires about 1000million tons of ore" and 8) Alexander and Street,1976

Iron and steel are the most commonly recycled metals with 14 tonnes of steel being recycled every second (Broadbent 2011) and "1.5 tonnes of CO_{2eq}, 1.4 tonnes of iron ore and 13 GJ primary energy" being saved for every tonne of steel recycled (Broadbent 2011). The data in Table 5-3 and the research of Broadbent 2011 both suggest that there should be considerable benefits from treating UK radioactive iron and steel alone either for recycling to open metal markets or beneficial re-use in the nuclear industry.

An early American radioactive metals recycling study showed that nickel was potentially the most important radioactive metal to consider because of its value and

its importance for stainless steel production. Teunckens et al 1993 also showed nickel to be important, calculating that 26 tons of recycled nickel a value of ~£90,000 to metals treatment project for Swedish steam generators, resulting in ~16% cost saving on the whole project. Hence it is important to consider low-volume, high-value metals for their economic benefit, as well as high-volume low-value metals such as iron that are likely to have a greater impact on total environmental emissions.

Steel recycling depends on the source of the scrap, the products recycled and the country (Davis et al 2007, Yellishetty et al 2011b). For example, steel container recycling for Australia, Brazil, China, Europe, Japan and the USA varied from 50% for Brazil to 85% for Japan (Yellishetty 2011b). Product recycling ranges from 38% for containers and vessels, to 85-89% for structural steel, vehicles and mechanical engineering according to Davis et al 2007. The limits of recycling depend on scrap availability, long residence times (one to several decades), cost competitiveness of scrap and recycling, increasing environmental constraints, the re-use of metals by non-melting techniques and the contaminant/residual elements in the scrap (Yellishetty et al 2011b). Yellishetty and colleagues cite five papers investigating this issue between 1954 and 2004 and state "…each time scrap was recycled the concentrations of residuals have gone up making processing more difficult" (Yellishetty et al 2011b). They also note – copper and nickel residuals for example are difficult to remove during recycling – zinc and lead can be partially removed - aluminium can be easily removed. Further, Davis et al 2007 note that 70% of prompt (new) scrap and end-of-life (old) scrap was recycled in the UK in 2001, the remaining 30% was lost to storage for economic reasons or disposed to landfill. Davies and colleagues recognise the improvements made in UK steel recycling but note that further effort is needed to improve this recycling rate.

5.4.2 Aluminium and other metals

Similar environmental benefits are claimed for aluminium, copper, zinc, lead and nickel by the BMRA (www.recyclemetal.org 2015), for example:

- 39% of new aluminium is from recycling, (consistent with Phylipsen et al 2002, Ashby 2009 and Allwood and Cullen 2012 and Aluminium Federation

(www.alfed.org.uk) 2015), with an energy saving of 95% (consistent with Phylipsen et al 2002, IPCC 2007)

- 32% of new copper is from recycling (but it could be as high as 45% according to the International Copper Association (www.copperalliance.org) and 40-60% from Ashby 2009), with an energy saving of 85%,
- 20% of new zinc is from recycling (consistent with Ashby 2009), with an energy saving of 60%, and
- 74% of new lead is from recycling, with an energy saving of 60% (consistent with the International Lead Association (www.ila-lead.org) 2015 and Ashby 2009).

The BMRA do not provide similar values for nickel recycling, neither was it possible to get equivalent representative figures from the Nickel Institute website (www.nickelinstitute.org). However, Ashby 2009 estimated a recycling fraction of 22 to 26% for nickel chrome alloys and nickel based super-alloys, which seems rather low for such a valuable material. Hence further research is needed to obtain more representative values for secondary nickel production from recycling.

5.5 How can we do better?

This section discusses how we can do better in prododuction and recycling metals. It looks at production technology improvement, other general improvement opportunities and what improvements are happening with UK radioactive metals treatment for recycling or re-use.

5.5.1 Production Technology Improvements

Energy costs are a significant proportion of the total costs for steel and aluminium production, 26% and 35% respectively (Allwood and Cullen 2012). These costs have driven the improvements in the production energy efficiency for these metals. It is estimated that the production energy for steel and aluminium is slightly more than double the ideal energy (Gibbs energy) for the world's best production plants (Allwood and Cullen 2012). These improvements have made a substantial reduction in associated CO_2 and other GHG emissions. Hence, further improvements in energy and GHG emissions will have to come from cleaner energy (i.e. decarbonising electricity production) and carbon capture and storage (CCS), improved product

design (to make products lighter), material efficiency, improved recycling or more refurbishment and re-re-use (Phylipsen et al 2002, Ashby 2009,Allwood et al 2010 and 2011, Allwood and Cullen 2012).

Allwood and Cullen 2012 also show that material costs for steel and aluminium production are large, i.e. about 40% and 25% of total production cost respectively. Metal ore costs and scrap metal costs fluctuate depending on national and global political and economic conditions and environmental constraints but the overall trend is ever upwards (Phylipsen et al 2002, Ashby 2009,Yellishetty et al 2011a and 2011b, Allwood and Cullen 2012). Although recycling may often give significant cost savings it is not always the case that scrap metal and recycling are economically favourable (Yellishetty et al 2011b). No data are provided in Allwood and Cullen 2012 for energy costs or efficiency, or material costs, for the productions of the other metals discussed previously but are assumed to be comparable with steel and aluminium. Hence they are only discussed where data is available.

Proposed improved technologies for steel production

Since future GHG reductions and other environmental improvements are important for recycling, current developments in production technologies are discussed here. These include the four main developments being progressed by the Ultra Low CO_2 Steel (ULCOS) making and research into improvements in aluminium production.

The main improvement processes for steel making are being taken forward in the ULCOS research programme (Birat and Hanrot 2006, Allwood and Cullen 2012). ULCOS-I ran from 2004 to 2010 and ULCOS-II will run between 2011to 2015. Key technologies are: 1) smelt reduction (ULCORED), improved coal based steelmaking, 2) combined smelter and cyclone (HISARNA), 3) using a top gas recycling blast furnace (TGRBF) and 4) electro-winning from iron ore (ULCWIN) (Phylipsen et al 2002, Birat and Hanrot 2006, Allwood and Cullen 2012).

The ULCORED process blends and preheats iron ore in a sintering and pelleting plant before the iron ore enters the direct reduction reactor. The off-gas from this plant is scrubbed in filters before being released. Natural gas and off-gas from the

direct reduction reactor are converted to H_2 and CO in a conditioning plant and injected into the direct reduction reactor. The reduction gases reduce the iron ore and it is tapped for transfer to an electric arc furnace for steel making. The CO_2 rejected from the reactor are processed and concentrated in a vacuum pressure swing absorption (VPSA) plant, this reduces CO_2 emissions by ~50% and the waste gas is suitable for CCS (Birat and Hanrot 2006, Allwood and Cullen 2012, ULCO website (www.ulcos.org)).

The HISARNA process combines preheating of coal to produce char in a twin rotary furnace, with iron ore fines melted in a cyclone furnace, which feeds molten iron ore to the convertor section of the reactor for ore reduction and iron production for steel making. The process removes the need for intermediate gas treatment, cooling and dust collection. The off-gas from the top of the reactor is transferred to a waste heat boiler to produce electricity for the process and heat for pre-heating the coal. The process produces highly concentrated CO_2 suitable for CCS hence offers significant (about 60-70%) CO_2 reduction (Birat and Hanrot 2006, Allwood and Cullen 2012, ULCO website (www.ulcos.org)).

The TGRBF plant uses the standard sintering and pelleting plants plus coke plant as a blast furnace but the off-gas is processed in a separation unit, which returns the CO to the furnace and releases the CO_2 to the ait or to a CCS facility. About 50% of the CO is used in ore reduction, hence recycling the CO allows multiple reducing agent use and results in overall CO reductions. Pre-heated O_2 is injected into the furnace rather than air, hence removes the N_2 emissions. The CO_2 reductions are ~15% without CCS and ~60% with CCS (Birat and Hanrot 2006, Allwood and Cullen 2012, ULCO website (www.ulcos.org)).

The ULCOWIN process produces iron metal from ore using electro-winning technology common to other metal production processes previously discussed and electro-platting plants for steel. It uses an alkaline molten oxide and electrolyte solution in a steel cell with liquid iron as the cathode and inert anodes. It is a high temperature electrolysis process (i.e. pyroelectrolysis) operating at about 1600^{o}C. It

is still only experimental but could lead to near zero CO_2 emissions ((Birat and Hanrot 2006, Allwood and Cullen 2012, ULCO website (www.ulcos.org)).

All of these processes are at the experimental or early development phase, as is the CCS process itself and the percentage effectiveness of the CCS plant are still unknown. Also, the savings in energy by decarbonised electricity will also be a factor. Some demonstration plants are operational but commercial production is not expected for 10 - 20 years.

Proposed improved technologies for aluminium production

The desire to produce inert anodes and wettable cathodes for aluminium production are discussed in Phylipsen et al 2002 and Allwood and Cullen 2012. Inert anode research has been active for four decades and seeks to find materials that are not consumed by electrolysis in the Hall- Heroult cells. The anode-cathode clearance is important for production efficiency and the development of inert anodes could also lead to multi-polar electrolysis cells improving production efficiency (Allwood and Cullen 2012). Spherical droplets of aluminium are produced on unwettable cathodes resulting in irregular anode-cathode separation. Wettable cathodes allow the aluminium to form a thin film on the cathode. This gives a more uniform metal film and reduces the anode-cathode separation, thus improving anode-cathode control and hence production (Allwood and Cullen 2012). The authors also discuss potential use of anode tilting systems to match the wave production in the electrolysis solution and preventing shorting between the electrodes by moving the anodes in sync with solution waves. This also raises issues with control system requirements.

As discussed in Section 5.2.2 to 5.2.4 primary aluminium production is energy intensive, has significant environmental impacts and produces significant amounts of waste. Aluminium production has increased significantly in recent decades and is likely to increase further into the future. The overall aims of the new technologies discussed above are to improve anode material consumption and electrolysis efficiency and to reduce energy consumption and waste. If successful, these technologies will result in reduced environmental impacts of primary aluminium production.

5.5.2 Other Improvement Options

The highest stage of the waste hierarchy is waste prevention (Figure 2-24 Chapter 2). Hence, effort needs to be made to minimise the use of metals either by substitution of mechanically equivalent material or material efficiency by making metal components lighter. This can be achieved through improved design for future use, or design for the environment (Ashby 2009, Allwood and Cullen 2012).

Waste re-use (including refurbishment) is higher up the waste hierarchy than recycling because it avoids the melting of metals for most recycling (Phylipsen et al 2002, Ashby 2009, Allwood and Cullen 2012). These authors suggest that re-use can be achieved by designing for re-use, by designing for deconstruction rather than demolition and by extending the life of products using metals. Examples cited are standardising the fitting of steel beams in construction rather than welding, opting for modular design, and improved maintenance and structural refurbishment. These options could improve the initial build and deconstruction by reducing the damage made to steel beams during mechanical demolition. This would allow construction material to be re-used immediately, or with restoration, rather than being cut up to melt as scrap for recycling. A suggested option for aluminium is solid bonding process (Allwood and Cullen 2012). This allows aluminium swarf and chips to be compacted at high pressure and elevated temperature (450-500^{o}C) and extruded to make profiled section. Although some heat is needed it is far less than the melting temperature for aluminium (~660^{o}C). This process uses much less energy than melting the swarf and chips and gives associated reductions in GHG emissions.

The use of these options in the nuclear industry is currently limited due to the high material quality and compliance requirements needed for components of nuclear facilities. Also, more work is needed to prove the processes and technology if the options are to be considered for treated metals re-use within the nuclear industry. Final implementation of any of these options will require demonstration that they are cost effective, functionally and technically acceptable and compliant with nuclear industry requirements.

5.5.3 Improvements from Treating UK Radioactive Metals

Current UK policy (DEFRA et al 2007), strategy (NDA 2010a) and practice (e.g. Shipton and Falconer 2013) for low level waste (LLW) metals promotes their treatment for recycling and/or re-use. Prior to the NDA's 2010 strategy, radioactive metals treatment was not common practice. The exceptions were the decommissioning of the redundant gas diffusion plant at Capenhurst (CDDUEF 1996) and the decontamination and recycling of the first Magnox boiler at Berkeley power station in 1997 (British Nuclear Group 2005). These projects followed experience from the international collaborative programmes for decommissioning nuclear facilities cited in Buckley et al 2004.

Recycling Rates

The annual throughput of UK legacy LLW metals was between 2300 and 4700 tonne/y from 2012 to 2014 (LLW Repository Ltd monthly Dashboard metrics (www.llwrsites.com)). The estimated annual average throughput for the next five years in estimated as 2400 to 3400 tonne/y from the LLW Repository Ltd joint and legacy nuclear site joint waste management plans (e.g. Shipton and Falconer 2013). This rate is within the licensed limits of the individual international treatment facilities (see Table A-3 and Table A-5), most of which appear to have excess capacity and are actively seeking UK contracts. It is expected that the recycling rate of UK LLW metals will continue for the short to medium term of UK legacy facility decommissioning.

UK Treatment Facilities

Until recently UK nuclear operators used fixed or mobile equipment to decontaminate LLW metal to exempt levels at sites (LLWR and NDA 2009a). However, Studsvik UK ltd began operating a radioactive metals recycling facility at Lillyhall in Cumbria in 2009. There is currently no commercial radioactive metal melting facility in the UK. However, a lead melting facility exists and was operated by Sellafield Ltd on behalf of the Nuclear Management Partners, contracted to the Nuclear Decommissioning Agency (NDA) for the Sellafield decommissioning programme. It's operational status and availability to external clients is currently unknown. The capabilities of the UK facilities are discussed in Section A.3.1.

International Treatment Facilities

International LLW metals treatment facilities include; the SOCODEI facility at Marcoule in France, Siempelkamp GmbH CARLA and GERTA facilities at Krefeld in Germany; the Studsvik AB Nykoping facility in Sweden and the Energy Solutions Inc. Bear Creek facility in the USA. There is also a Russian Federation facility at Sosnovy Bor, near St Petersburg, operated by ECOMET-S. The capabilities of these international facilities are summarised in Table A-3,Table A-4 and Table A-5.

The German, Swedish and American facilities are authorised by UK and International regulators and used by UK nuclear operators. The French facility is also authorised and available under the LLW Repository Ltd framework and an Anglo/French governmental agreement, but has not yet been used by UK nuclear operators. The ECOMET-S facility is currently only available for the treatment and recycling of Russian and Ukrainian radioactive metals and not authorised for use by UK nuclear operators.

These facilities cut up, surface decontaminate and melt the LLW metals to achieve regulatory exempt levels of melt ingots for release to scrap metal markets or used to produce waste containers, shielding cylinders or slabs etc. Melting provides additional decontamination as it partitions the remaining radionuclides between the melt ingot, slag, dust and gas as shown in the indicative example presented in Figure 5-6. The low volatile nuclides, Co60, Fe55, Mn54 and Ni63 remains in the melt ingot, the more volatile nuclides such as Cs 134, Cs137, Eu152 and Eu154, H3 etc migrate to the slag and dust, or simply off-gas through the ventilation stack. The actual nuclides present will depend on the component metal, the type of facility the component came from and the radiation history of the facility. The partitioning will depend on the type of melting facility and melt chemistry (including the refractory lining material and melt additives).

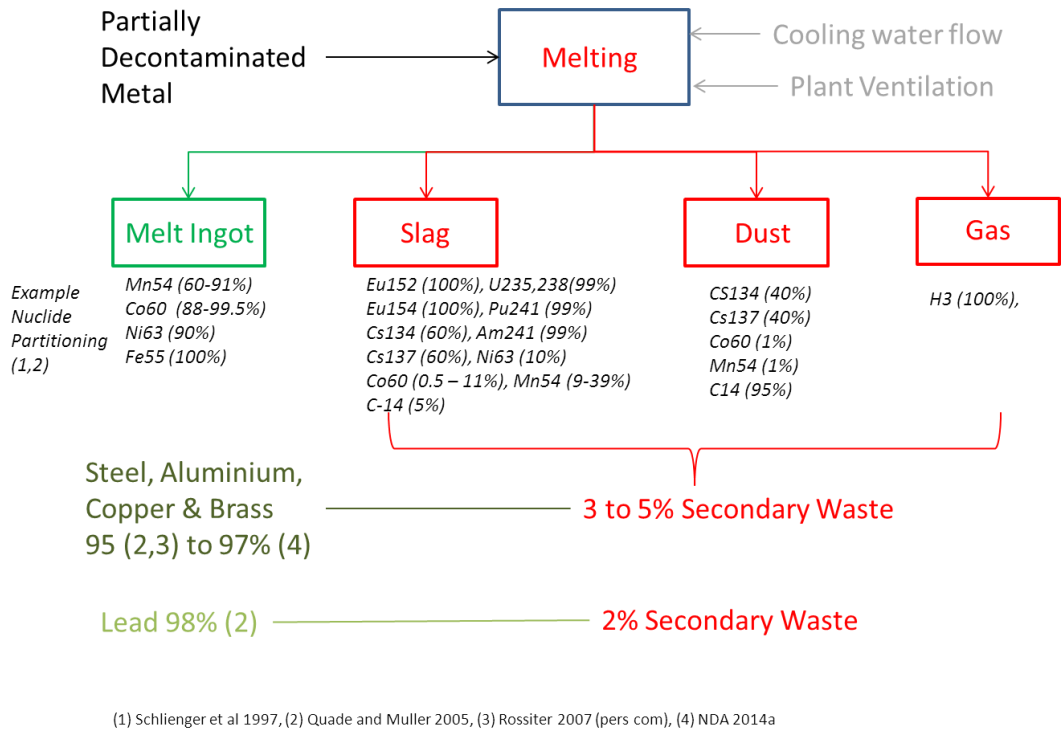

Figure 5-6 Distribution of radionuclides after melting steel (partitioning based on Schlienger et al. 1997 and Quade and Muller 2005). Am – Americium, C – carbon, Co – cobalt, Cs – Caesium, Eu – Europium, Fe – iron, H-hydrogen (tritium), Mn – manganese, Ni – Nickel, Pu – plutonium, U-uranium

The additional decontamination afforded by partitioning nuclides across the ingot, slag, dust and gaseous emission means that about 95 to 98% of the original metals for treatment can typically be recycled or reused, the remaining 2 to 5% is secondary waste. Quade and Kluth 2009 show that with 95% recycling, the 5% secondary waste is typically made up of ~0.5% sweepings (from cutting up and surface decontaminating the metal), ~3% from the slag, ~0.8% from the furnace lining material and ~0.7% from dust collected in filters in the ventilation system. These secondary wastes are normally packaged and returned to the waste consignor for disposal in the UK. Hence the research focuses on identifying and quantifying the potential environmental impacts of these high recycling rates.

Treated melt ingots with activity above exempt levels can be stored at the facilities to benefit from radioactive decay for future release to metal markets or returned to the

UK for disposal. This was not modelled in the research as it was assumed that all the ingots from international treatment facilities could be released or re-used.

Benefits of Radioactive Metals Recycling

In addition to conserving scarce disposal volume benbefits also accrue from reduced packaging. The nominal waste load of a 'single-use' carbon steel half height ISO (HHISO) freight container for LLW disposal is 10tonne. Hence, every 10tonne of LLW metals treated for recycling or re-use saves; the production and transport impacts and costs of one 3tonne HHISO container, several tonnes of cement grout and 20m^3 of disposal volume and associated disposal costs. There would be similar, but potentially higher, savings for treating and recycling intermediate level waste metal that has decayed or been decontaminated to LLW as they require "single-use" stainless steel containers with a much lower waste loading and higher environmental impacts. Intermediate level waste (ILW) metals and activated metals pose handling, transport and treatment operator radiation dose issues. Hence, treatment and recycling are not yet undertaken at the international treatment facilities. Further research would be needed to confirm the technical and economic viability of such an option.

5.6 Chapter Summary

Key observations from this chapter are:

- Metals are a valuable resource, estimated quantities are large but finite, some metals are scarce,
- Metals extraction and processing is energy intensive and has significant environmental impacts, these impacts can be reduced by recycling and re-use,
- Global metal demand is increasing rapidly, steel and aluminium are expected to at least double by 2050,
- Landfill disposal can be reduced by 'waste mining', several important example already exist,
- Metals recycling and reuse are increasing, but cannot meet predicted future demands, hence production from virgin ores is likely to continue for some considerable time,

- Energy efficiency in best-case steel and aluminium production is already close to the ideal limit, upgrading metal production facilities may be problematic,
- Further improvement in technology to mitigate environmental impacts, including de-carbonising/cleaning electricity generation and proving Carbon Capture and Storage (CCS) are under development,
- Improvements in product design for future use, increased product life, designing for deconstruction, material substitution, minimising waste and production losses will all contribute to reducing the environmental impacts of the metals cycle.

Some of these aspects are recognised in new nuclear power station build and in the end-of-life disposition of UK legacy radioactive metals. UK radioactive waste strategy is now prioritising metal recycling and re-use, thus minimising the demand for "single-use" steel disposal packages, reducing cement grout and backfill and conserving scarce disposal volume capacity at national repositories.

The LCAs in this research use historical European data for metals production from the late 1990s with updates to circa 2010. Hence the environmental impacts for metals recycling and avoided impacts are likely to be over estimates if the new technologies are eventually adopted. Therefore the results of this reseach could form part of the baseline of assessing the benefits of future improvement in production. For examples, ULCOS technologies potentially offer 15-20% CO_2 emissions for steels if carbon capture and storage is excluded and 50-70% if it is included. These values are speculative because of the immaturity of emerging steel production and CCS technologies. Future production would be a mix of current and new technologies hence the percentage improvements will be much smaller. Nevertheless, as technology changes, and the waste metals arise from the inventory, production and recycling environmental impacts are likely to reduce.

Whilst new production technologies are likely to reduce environmental impacts, the impacts associated with virgin production of metal ores are likely to rise, since the quality and ease of extraction of ore deposits are reducing: mines are becoming

deeper, the overburden and the tailings are increasing and hence more energy is required. These factors are also likely to result in an increased health and safety risks and a rise in the economic value of the metals as they become more scarce and costly to mine. Mining and processing technologies may mitigate some of these effects, but estimating any improvements is beyond the scope of this research.

6 Research Methodology

The fundamental aim of this research is to assess the potential environmental benefits and costs of treating UK radioactive metals for recycling or re-use as an alternative to disposing of them to authorised facilities in steel containers (Section 1.5). Chapter 4 showed that LCA was used extensively to investigate the potential environmental impacts of the nuclear fuel cycle, but that there were gaps in knowledge particularly for radioactive waste management and decommissioning. Chapter 5 showed that LCAs are commonly used to assess the environmental impacts of metals and that there is growing pressure to conserve virgin metal ores, minimise the energy for metals processing and associated environmental emissions and to reduce losses to disposal.

This chapter discussed LCA modelling and system boundary, the LCA data and the software package used plus the LCA and cost assumptions applied to the assessment of the environmental impacts associated with recycling, re-use and disposal of wastes.

6.1 LCA modelling for this research

This section discusses the construction of LCA models using propriety software, an outline of the life cycle impact methods, the underling conceptual used and an example of the impacts assessment stages.

SimaPro 7.3.3 is one of many LCA software packages. It was released in 1990, has been updated regularly and is widely used internationally by industry and universities (Menke et al 1996, Phylipsen et al 2002, Curran et al 2006). The Ecoinvent 2.2 database is embedded in SimaPro and has been available since the late 1990s. Ecoinvent has in excess of 4000 datasets (Frischknecht et al. 2007), is updated regularly and is commonly used in major LCA software packages, hence can be regarded as the industry standard.

SimaPro LCA models are constructed via assemblies and sub-assemblies, for example in this research a sub-assembly could include the disposal container metal, the container production, the waste metals plus empty and full container transport. The assemblies and sub-assemblies are linked to end-of-life waste treatment and disposal processes to form an integrated life cycle (Lopes 2010). Waste treatment processes in this research include metal component size reduction, decontamination, metal melting, secondary waste processing, avoided future metals (including re-melting) for recycling or re-use and the return of secondary waste for final disposal. Disposal is to current authorised low activity waste (LAW) facilities (i.e. for very low level waste (VLLW) metals or low level waste (LLW) metals) or to a future higher activity waste (HAW) facility (i.e. for intermediate level waste (ILW) metals or depleted uranium). The waste categories are summarised in Box1-1. LLW and ILW metals, residual waste and depleted uranium are normally combined with cement grout to form a stable waste-form to produce an acceptable radioactive waste disposal package. The typical waste packages are summarised in Box1-2.

Ductile cast iron containers have been identified by the nuclear operators and regulators for specific GDF waste streams only. These containers have not been included in the research because of their small waste capacity and very high cast iron inventory and are not suitable for most metal wastes.

6.1.1 Eco Indicator 99

Eco-Indicator 99 is one of a range of life cycle impact assessment (LCIA) methods available in SimaPro 7.3.3. Eco-Indicator 99 has been cited in LCAs for the metals industry (Rebitzer and Buxaman 2005, Lee and Park 2005, Tongpool et al 2010, Awuah Offei and Adekpedjou 2011), abiotic resource depletion (Steen 2006, Yellishetty et al. 2011a), methodology development (Schmidt and Sullivan 2002, Sonnemann et al. 2004, Baumann and Tillman 2004, Pennignton et al. 2004, Koffler et al 2008, Finnveden et al. 2009, Guinee et al. 2011) and materials and clean technology (Phylipsen et al. 2002, Ashby 2009). Hence Eco-Indicator 99 has been chosen and the life cycle impact assessment method for the research.

Eco-Indicator 99 is a damage orientated method. It converts eleven environmental impact category indicators into three damage categories; Human health, ecosystem quality and resources. The processes involved are grouped into three steps as shown in Figure 6-1.

The three steps incorporate three separate but partially linked fields of knowledge and reasoning called the:

- Technosphere - life cycle description and emissions allocation
- Ecosphere - modelling the damages to the environment, and
- Valuesphere - modelling the perceived seriousness of the potential environmental damages and the value choices made in the Technosphere and Ecosphere.

The first step identifies and collates inventory data for energy, materials, products and wastes and models the flows to and from the life cycle processes. The second step allocates the inventory analysis results to resources, land use and emission environmental impact categories to estimate the potential damage from each category. The final step normalises and weights the damages to produce an eco-indicator score. The items in red are the analyses processes for the impact categories. The items in blue are the processes for converting the impact categories to damages and eco-indicator score. These steps are now discussed in more detail using ionising radiation as an example for damage to Human health. The ionising radiation impact category method is based on the research of Frischknecht et al. 2000 and discussed in Goedkoop and Spriensma 2001a and ISO 14047 2003.

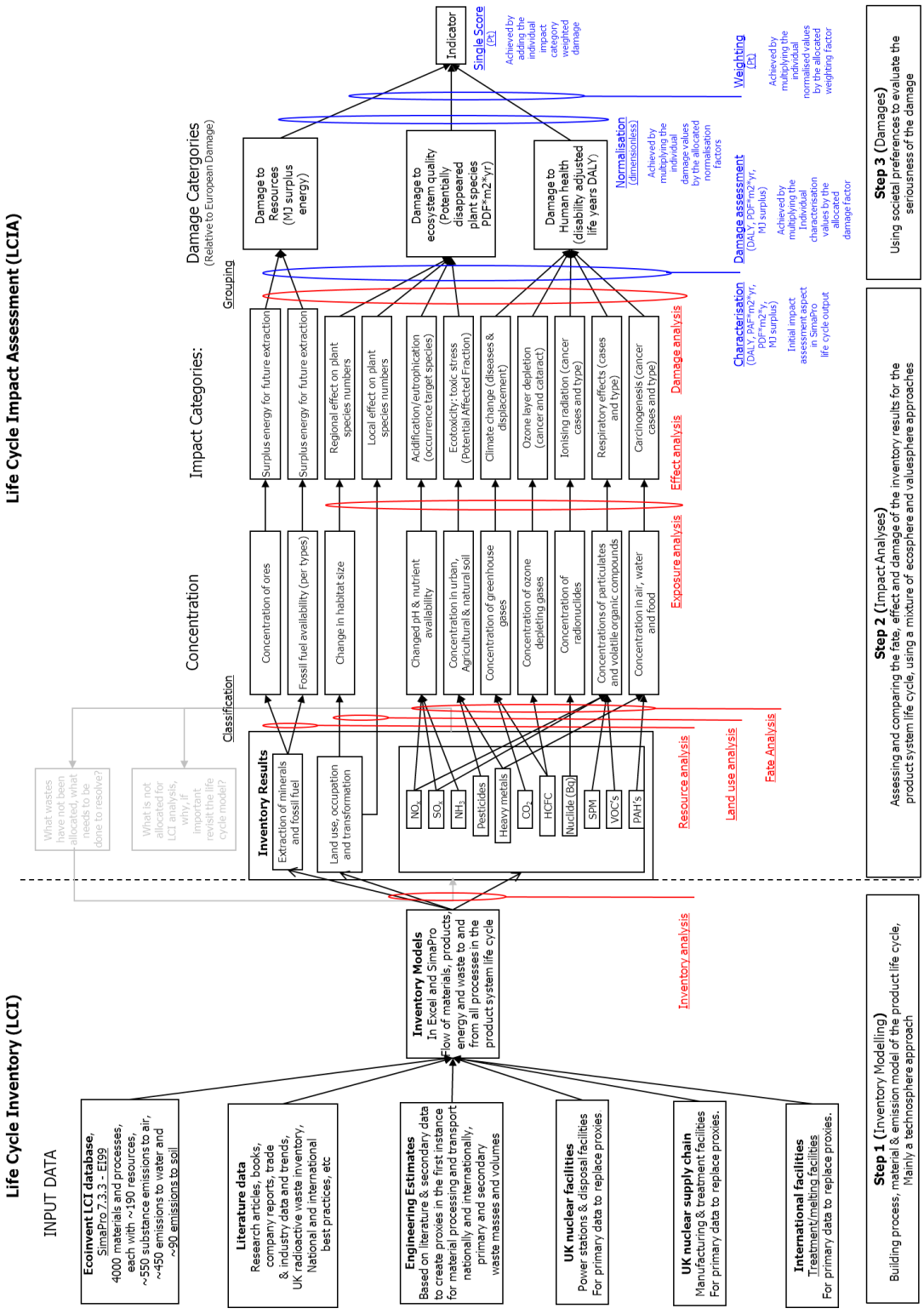

Figure 6-1 Eco-Indicator 99 Methodology (Based on Goedkoop et al. 1998, Goedkoop and Spriensma 2001a, Phylipsen et al. 2002 and Hischier et al. 2010)

Step 1 of Figure 6-1: Inventory Modelling

The inventory includes all the energy and resources used in the product system processes of the life cycle and the solid, liquid and gaseous wastes produced. This is represented in more detail for this research by the end-of-life conceptual model for radioactive metals shown in Figure 6-2.

The conceptual model in Figure 6-2 is the result of the simplification of five earlier iterations. The initial flow diagram was for the whole nuclear fuel cycle, based on Figure 5-2. It linked the key processes requiring energy, material, waste and transport requirements. However, was to be too general and covered too many fuel cycle stages for what was needed. The second flow diagram for radioactive waste within the UK nuclear industry, but it too was too general. The third stage was a radioactive waste metals end-of-life LCA flow chart for waste generation, pre-treatment, treatment, conditioning, storage, retrieval, recycling and disposal commonly used as key process stages for regulation. However, this was too complex to model in the time available and data available. Hence, a further simplification was made to produce a simplified version that could form the basis of a conceptual model. This was finally simplified to the foreground and background model by considering the steel and lead treatment processes diagrams of the Studsvik facility in Nykoping kindly supplied by Rossiter 2007. The full flowcharts and process images used to develop the end-of-life concsptual models are presented in Appendix C.

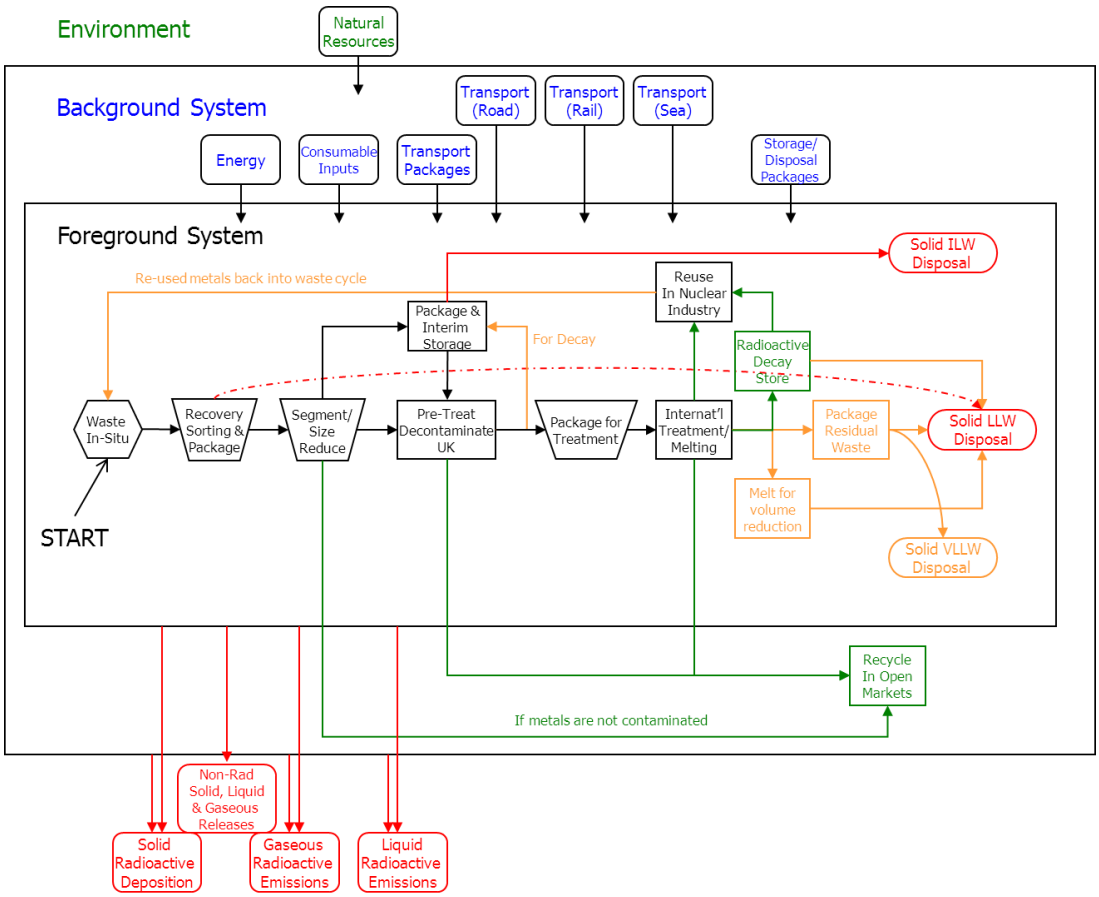

Figure 6-2 End-of-life Conceptual Model for radioactive waste metals (VLLW is very low level waste, LLW is low level waste, ILW is intermediate level waste)

The environment in Figure 6-2 is the source of all materials and the sink for all wastes. The starting point for the model inventory is the radioactive waste metals in-situ at the UK nuclear sites. The main processes of interest for the treatment (from size reduction to melting) for recycling, re-use of the metals in the nuclear industry or for waste volume reduction, and disposal of the residual waste, constitute the foreground system. The ancillary materials (e.g. consumables, storage, transport and/or disposal packages), transport and energy constitute the background system. The processing of exempt level metal ingots, from national and international treatment facilities, in the scrap metal markets is included in the background system (Figure 6-2) as this is outside the nuclear industry.

Items in green, Figure 6-2, denote a potential significant environmental benefit, those in red represent potentially significant environmental detriments. Items in orange represent processes and flows with a potential reduction in environmental detriment.

The items in blue, for the ancillary processes, will incur some environmental detriments while supporting the overall life cycle. Items in black are the main treatment processes and flows. All the aspects depicted in Figure 6-2 are used to estimate the inventory data in Step 1 of Figure 6-1.

Step 1 of Figure 6-1 primarily involves decisions and data from technological processes of the life cycle and some value judgements on what to include or exclude in the inventory model. Hence, it is a mixture of Technosphere and Valuesphere aspects.

Step 2 of Figure 6-1: Impact Analyses

The inventory analysis balances the mass and energy flows for the life cycle models and provides results for resource extraction - land use, occupation and transformation - emissions to air, water and soil. These inventory results are used in the resource analysis, land use analysis and fate analysis discussed next.

In the sections below, as an example, radioactive materials released from the nuclear fuel cycle and from coal-fired power plants (during energy production) are described in terms of their Impact analysis i.e. the process of resource analysis, through land-use analysis to fate analysis (Goedkoop and Spriensma 2001a and 2001b, Frischknecht et al. 2000). These are finally discussed in terms of their impact on human health.

Resource Analysis

Two resources are considered in Eco-indicator 99, potentially sustainable mineral ores and non-sustainable coal and oil deposits. Resource analysis assesses the impacts of extracting ores in terms of the reduced concentration of the remaining deposits (as discussed for metals in Chapter 5). In Eco-Indicator 99 the geo-static model of minerals knows as Laski's Law is assumed i.e. "… the distribution of concentrations of mineral resource is log normal if we plot it against grade" (Goedkoop et al. 1998). Similarly, the supply of easily extractable fossil fuel resources will decline and lower grade sources will have to replace them, e.g. substituting shale oil for liquid oil (Goedkoop and Spriensma 2001a, Phylipsen et al.

2002). Damage analysis links the reduced ore and deposit grades with the increased energy needed to extract them giving an indication of potential damage expressed in terms of MJ of surplus energy per kg needed rather than the financial evaluation used in other LCIA methods (Muller-Wenk 1998). The absolute value of surplus energy is not relevant: what is important is the difference between surplus energies of different resources or processes caused by human activities depleting these large but finite resources over time.

Hence, in terms of radioactive materials, the different assumptions on energy sources will affect the mass, material types and level of radioactivity (in Becquerels (Bq)) of any radioactive materials released during energy production.

Land Use Analysis

Land use analysis considers the use, occupation and transformation (degradation) of the land as a result of human activities on a regional and local basis in Eco-Indicator 99. Many plant species may be exposed to harm, or extinction, as their habitat is lost due to human use. The level of harm depends on the different types of land use, the diversity and population of species affected and the size of the area affected. The final damage is expressed as Potential Disappeared Fraction (PDF) for the area affected each year. The damage is therefore to ecosystem quality. The land-use analysis relates directly to impacts from changes in land use due to industrial development and hence is not relevant in the example of radioactive material emissions.

Fate Analysis

Fate analysis applies to acidification/eutrophication and ecotoxicity affecting ecosystem quality. It also applies to carcinogenic substances, respiratory organic and inorganic emissions, ionising radiation, ozone layer depletion and climate change affecting Human health. The analysis assesses the emission of chemicals to the environmental compartments air, water and soil. The analysis depends on the substance, where it goes, how long it remains in the air, water or soil and the degradation of organic substances in the media. Fate analysis takes the mass of the

emission and represents it as a temporary increase of concentration of the substance in the three environmental compartments. When radionuclides are released they are transported, dispersed and deposited in air (Bq/m^3), water (Bq/l) and soil (Bq/m^2 or Bq/kg). The Eco-Indicator 99 Fate analysis model for ionising radiation is based on the research of routine gaseous and liquid discharges from the French nuclear fuel cycle (Dreicer et al. 1995).

Exposure Analysis

Exposure analysis assesses the impact of the increased concentration of a substance from the Fate analysis and relates it to how much is absorbed by human or plants. The exposure analysis for gaseous release of radionuclides models contamination of air, soil and vegetation (and hence animals). Liquid discharges deposit radionuclides to water, potentially contaminating fish and seafood, and also crops via irrigation. If these depositions are high they can cause harm to humans through inhalation, ingestion of food and exposure to skin. The absorbed dose is measured in Grays (i.e. specific energy (J/kg)). The effective dose in Sieverts (Sv) (i.e. J/kg of body weight) for humans depends on the form of radioactivity (alpha, beta and gamma radiation, or neutrons) and how the radiation affects different organs and tissue. The collective dose (commonly called man.Sv) is also calculated for the effect analysis.

Effect Analysis

Effect analysis takes this absorbed dose of the concentrated substance and estimates the type and number of diseases, cancers and respiratory effects it causes in humans. It also links the increased concentrations to the toxic stress to plants for ecotoxicity, or the changes to acidity or nutrient levels in soil or water, for acidification and eutrophication, also effecting plants. The Effect analysis for radionuclides relates the estimated doses to the rates of non-fatal and fatal cancers and possible intergenerational (hereditary) effect from damage to genetic material. The effect of radiation is measured in number or cases/man.Sv and used in the damage analysis.

Damage Analysis

Damage analysis then estimates how many years lived disabled (YLD) or how many years of life are lost (YLL) due to the cancers and other diseases for damages to Human health. Eco-indicator 99 uses the aggregated and disability weighted World Health Organisation measure; disability adjusted life years (DALYs). The damage analysis for radiation results in DALYs/fatal cancers.

Damage is measured as Potentially Affected Fraction (PAF) for the area affected each year for ecotoxicity due to toxic stress to plant species, and Potentially Disappeared Fraction (PDF) for the area affected each year for identified target plant species only for acidification and eutrophication.

Damages are calculated on a world-wide basis for greenhouse gas emissions, ozone depleting emissions and long lived radionuclides as they can be dispersed globally. The damages for the other eight impact categories are calculated on a European basis (Phylipsen et al. 2002).

Step 2 of Figure 6-1 calculates the environmental damage of emissions from the life cycle inventory analysis. The calculation models use natural science to estimate the changes to the environment but also some value judgements on what to include and how to estimates the impacts and damages. Hence, Step 2 of Figure 6-1is a mixture of Ecosphere modelling and Valuesphere considerations.

Step 3 of Figure 6-1: Damage Categories - Endpoints
The outputs of Eco-indicator 99 in SimaPro are:

- Characterisation values for each of the eleven impact categories - Carcinogens, Respiratory Organics, Respiratory Inorganics, Ionising radiation, Ozone Depletion and Climate Change in DALYs for damage to Human health - Ecotoxicity in PAF.m^2.yr, Acidification/Eurtophication and Land Use in PDF.m^2.yr for damage to ecosystem quality - Minerals and Fossil Fuels in MJ surplus energy for damage to resources.
- Damage assessment - achieved by multiplying the characterisation values of the impact categories by prescribed damage factors. Ecotoxicity results (in

PAF) are multiplied by 0.1 in the Damage assessment to produce PDF values for damage to ecosystem quality. All the other impact categories characterisation values remain unchanged.

- Normalisation - achieved by dividing damage values by prescribed normalisation factors. Normalisation in Eco-indicator 99 is based on European data for population, energy and emissions. It gives the relative magnitude of the impact categories within the damage categories, but gives no indication about the importance or seriousness of damages. The units are dimensionless.
- Weighting - achieved by multiplying normalisation values by prescribed weighting factors to give Eco-Indicator points. Weighting is a particularly controversial stage of LCA and is not allowed by ISO 14040 2006 for LCAs presented to the public. Eco-Indicator 99 uses weighting factors, produced by expert panels, to assign relative importance to the eleven potential environmental impacts within the three damage categories. Weighting is almost totally within the Valuesphere.

Cultural Theory – Subjectivity and Uncertainty

Steps 1 and 2 essentially represent the use of natural science to evaluate the inventory and potential environmental effect results. However, as Goedkoop et al. 1998 and Hofstetter et al. 2000 point out natural science cannot say anything about the importance of the results, as importance is subjective. Eco-Indicator 99 models subjectivity in Step 3, and for choices within the modelling, using social preferences adopted from the socio-cultural viability theory (Cultural Theory) of social science (Thompson et al. 1990, Goedkoop et al. 1998 and Hofstetter et al. 2000).

Cultural theory identifies five archetypes (cultural perspectives), the egalitarian, the fatalist, the hierarchist, the individualist and the autonomist (Thompson et al. 1990, Hofstetter et al. 2000 and Goedkoop and Spriensma 2001a). The social typology of the archetypes used by Thompson and others is constructed in two dimensions - the x-axis is denoted as the Group – the y-axis is known as the Grid. The Group shows "…the extent to which an individual is incorporated into bounded units" and the Grid

shows "…the degree to which an individual's life is circumscribed by externally imposed prescription" (Thompson et al. 1990). A positive x-value denotes strong influencing from within the group and a negative value a weak influence (Goedkoop and Spriensma 2001a). A positive y-value represents strong association with externally imposed prescribed ideas and a negative value a week binding to these prescriptions (Goedkoop and Spriensma 2001a).

It is normally accepted that the fatalist and the autonomist do not generally take part in decision-making groups (Hofstetter et al. 2000 and van Vuuren et al. 1999). The fatalist acts on their own, is easily controlled by external pressure hence guided by other and little or no opinion of their own (Goedkoop and Spriensma 2001a). The autonomist tends not to be influenced by the Group or Grid and tends to think independently of the others in a group. Hence, these two archetypes are excluded from Eco-Indictor 99 and from the long-term perspective of metals used in van Vuuren et al. 1999 discussed in Chapter 5.

The typical world view, management style and characteristics of the egalitarian, hierachist and individualist archetypes are shown in Table 6-1. The attributes in Table 6-1 were selected from tables presented in Goedkoop et al. 1998, van Vuuran et al. 1999, Hofstetter et al. 2000, Phylipsen et al. 2000, VROM 2000 and Goedkoop and Spriensma 2001a. They represent key elements for each archetype for this research. Of course, no-one or no group is entirely one perspective and in reality will be a mixture of all perspectives.

Table 6-1 Summary of main characteristics of the three cultural perspectives of Eco-Indicator 99

Characteristics	**Hierarchist**	**Egalitarian**	**Individualist**
Time perspective	Balance of short and long-term	Very long-term	Short-term
Manageability	Proper policy can avoid many problems	Problems can lead to catastrophe	Technological advances can overcome problems
Management style	Control	Preventive	Adaptive
Procedures applied	Rules	Ethical Standards	Skills
Criteria	Evidence	Argument	Experience
Required level of knowledge	Inclusion based on consensus, largely complete but	Minimum scientific evidence needed, nothing left out,	Only proven effects are included, least complete but lowest

Characteristics	Hierarchist	Egalitarian	Individualist
	moderate level of uncertainty	most complete but highest uncertainty	level of uncertainty
Spatial considerations	Local over global	Global	Local
Intergenerational factors	Present and future are equal	Present less important than the future	Present more important than the future
View of resources	Scarce	Depleting	Abundant
Energy future	Technical advances will address problems	Radical changes needed now	Carry on as usual
Attitude to risk	Risk accepting	Risk averse	Risk seeking

Eco-Indicator 99 has three distinct models; one each of the cultural prespectives.
Many of the steps in the calculation of each impact category within an environmental damage are modified to represent the preferences of each cultural perspectives. For example, in the case of carcinogens:

- The DALY calculation for damage to human health has the potential to discount the financial costs of care, however, discounting is not included in the model for any of the cultural perspectives
- The DALY calculation also includes the potential for age weighting of those affected, hence:
    - The Individualist calculation includes the age weighting as Individualist value the life of people at say 20 more than those at say 40
    - The Hierarchists and Egalitarians calculations do not include age weighting, because Hierarchists adopt the principal that the age weighting is not allowed in law and the Egalitarian see everyone as equal
- Ecoinvent has three data sets for cancers, group 1 are known substances that are proven to cause cancers in humans, group 2 are known substances that could probably cause cancers in humans and group 3 are known substances that could possibly cause cancers in humand
    - Individualists only consider proven damages therefore only the first group of the cancer subtances are used in the calculations

- Hierarchists consider damages that are proven and those considered by experts or governmental as significant, therefore cancers from groups 1 and 2 of cancer substances are used in the calculations
- Egalitarians consider all possible daamges therefore all three groups of cancer substances are used in the calculation

Applying the normalisation and weighting factors prescribed for the three cultural archetypes in Eco-Indicator 99 produces three structurally identical but culturally different outcomes for the eleven impact categories. In addition, the weightings can either be average values (as shown in Table 7-1) or specific to the cultural perspective. Hence, six models can actually be produced (as shown in Figure 7-11). These can then be used to represent the range of possible value judgements of a group of final decision makers.

Normalisation Factors

The normalisation factors for the three archetypes, applied to the impact categories for damages to human health, ecosystem quality and resources, are embedded in Eco-Indicator 99 (e.g. Goedkoop and Spriensma 2001a, Hischier et al. 2010 (Tab 4.2). Changes to the normalisation factors from improvements in the calculation method and/or data are incorporated in updates of SimaPro. SimaPro version 7.3.3 was used for this research.

Weighting Factors

The final weighted impact results calculated are Eco-indicator points (Pt) representing a dimensionless unit, "…1Pt is representative of one thousandth of the yearly environment load of one average European inhabitant" (VROM 2000). The absolute Eco-indicator point values are not significant in terms of their absolute value, but have been developed to compare relative differences between process and material impacts of different environmental options.

The normal consensus is that the Hierarchist/Average normalisation and weighting factors are generally best used for LCAs (e.g. Goedkoop and Spriensma 2001a and

EC/JRC 2011). Hence the Eco-Indicator 99 Hierarchist/Average modelling, normalisation and weighting are used throughout the research. However, sensitivity to the application of the normalisation and weighting factors within all the archetypes is explored for the WAGR boiler case study in Chapter 7.

6.2 LCA System Boundary

As discussed in the literature review in Chapter 4, defining the boundaries of an LCA has a substantial effect on the outcomes. Hence, it is important to consider carefully what the appropriate boundaries are for this research. Since all the wastes considered in this thesis are from current UK nuclear facilities they are designated - legacy wastes. The wastes currently exist, or will exist in future decommissioning of the current UK nuclear facilities as shown in Figure 2-21 and Figure 2-22. As a consequence, no account is taken of the environmental impacts associated with the original production or use phases of metallic components or other waste streams.

The LCAs consider the disposal of VLLW, mixed VLLW/LLW and LLW metals as well as their recycling and re-use. ILW metals that can be decontaminated to LLW or will decay to LLW levels before final decommissioning are also considered. VLLW and LLW disposal facilities already exist but ILW has to be stored until a future UK geological disposal facility (GDF) is available, or near surface near-site long-term management facilities are available for Scotland, as discussed in Chapter 2. The national repository near Drigg in Cumbria and a future GDF adjacent to Sellafield were assumed for LLW and ILW disposal respectively. It is important to note that since the modelling was done a decision has been made not to site a GDF in Cumbria. It is retained here as representative case only. VLLW was assumed to be at one of the three authorised commercial VLLW landfill sites in England. The Studsvik metals recycling facility at Lillyhall in Cumbria was assumed for UK radioactive metals treatment. The use of radioactive metals treatment facilities in France, Germany, Sweden and the USA are also investigated, resulting in consideration of national and international transport, as well as incorporating the different international energy mixes that supply each international recycling facility.

Radioactive waste metals arising from the UK new nuclear build programme are not included in this research. Eight new nuclear power stations are planned for England and Wales by 2025. Modern light water reactor designs (i.e. the European Pressurised water Reactor (EPR) and UK – Advanced Boiler Water Reactor (UK-ABWR)) are progressing through the UK regulatory processes. These facilities will have longer operational lives and higher operating powers and efficiencies compared to legacy nuclear power station, hence will generate less radioactive waste per kilo Watt hour (kWh) of power produced (again, note that light water reactors are generally smaller have a higher power density and use less materials). These wastes have not yet been generated, so other boundaries might well be more appropriate. It is worth noting, however, that the environmental impacts for new nuclear build wastes may be expected to be broadly similar to those of legacy waste; thus this research may help to inform their future management.

In the final research chapter, the LCA inventory changes from radioactive metals to the current inventory for UK civil depleted uranium. The system boundary remains largely unchanged as the investigation focuses on the storage and disposal of depleted uranium, not its production. Depleted uranium, if classified as HAW, would be a significant issue for a future UK GDF. Hence, this research is aimed at informing future decisions regarding the disposal of depleted uranium and the results are discussed in detail in Chapter 9.

6.3 LCA Data

This section discusses the data used for the small scale case study and UK metals inventory disposal, treatment and recycling analysis, the data used for the interim storage and disposal of depleted uranium and the Ecoinvent database data.

6.3.1 Windscale AGR Boiler and General Radioactive Metals Data

The WAGR case study metal data were predominantly derived from the nuclear operator (EC-CND circa 1995, Dixon 1999, Bayliss and Langley 2003) and is utilised in Chapter 7. The VLLW, LLW and ILW metals data for the entire UK legacy radioactive metals investigation, discussed in Chapter 8, were taken from the 2010 UK radioactive waste inventory (NDA and DECC 2011). However, metal

masses for copper, nickel and zinc were subsumed in “Other Metals” after the 2001 UK radioactive waste inventory, estimates of these were taken as the 2001 inventory.

VLLW volumes started to appear in the 2010 UK radioactive waste inventory but a breakdown of VLLW metals was not included. At the time of the research the VLLW from Springfields was reported and this was used to make a first order estimate of VLLW metals. The 2013 UK radioactive waste inventory now presents the VLLW volumes but does not differentiate VLLW metals from the general VLLW (NDA and DECC 2014e).

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 half height ISO (HHISO) freight containers were taken from Yorkshire Marine Containers (www.ymccontainersolutions.com) and from James G. Carrick & Co Ltd (sales@jamesgcarrick.com) for the 210 litre mild steel drums.

6.3.2 Depleted Uranium Data

The depleted, natural and low enriched uranium (DNLEU) inventory data were taken from the NDA Integrated Project Team (Uranium) (NDA 2013b) and is utilised in Chapter 9. The number of DV70 $3m^3$ boxes was estimated from the UF_6 mass converted to U_3O_8 using the 1.18 factor implied in the NDA appraisal of the Urenco UK Ltd application for a Conceptual Letter of Compliance (CLoC) in 2009 (NDA 2009b) assuming 10 to 12tonne per box (Hartmann et al. 2001, Capus and Durante 2007, Kastelein 2009 and Jones 2014 (pers. comm.)). The number of 200l drums for the Magnox depleted uranium and miscellaneous UO_3 was estimated from the DNLEU inventory and drum numbers in (Mason 2009) and discussion with CNS staff. The number of 50litre kegs for Thermal Oxide Reprocessing Plant (THORP) product uranium, i.e. recycled UO_3, was estimated for the DNLEU inventory and Jones 2014 (pers. comm.). The number of special full height ISO freight containers needed for THORP product uranium transport and disposal were taken from

(Southern 2011). Additional waste container data were taken from the NDA website (Poyry 2010, Hickford et al. 2012 and Wilson et al. 2012) plus container designer and manufacturer websites. Depleted uranium and associated container data are summarised in Section A.3.4.

6.3.3 Database Data

Where process or input data were not readily available, data were taken from the industry standard LCA database, Ecoinvent 2.2, which is embedded in SimaPro. The general structure of the Ecoinvent database is described in Frischknecht and Rebitzer 2005. Both Frischknecht and Rebitzer 2005 and Rebitzer et al. 2004 note that the Ecoinvent database structure for data processes, modelling, validation and administration is consistent with the requirements in international standard for data formating for LCAs (ISO/TS 14048 2002).

6.3.4 UK Legacy Inventory Data Quality

The regularly published waste inventory data are the best available estimates of UK legacy radioactive wastes. It uses proven, but developing, calculation methodology and responds to suggestion and comments for improvements from the industry, regulators and stakeholders. The NDA has introduced an improvement to the traditional multi-document, multi-spreadsheet waste inventory by launching a new UK radioactive waste inventory website. This gives easy and transparent access to the inventory for users and other interested parties. It is presumed that the website and data are validated using NDA management system procedures. This is an advance for those interested in using the UK radioactive waste inventory data for LCAs.

The various publications of the UK radioactive waste inventory also state the broad uncertainty in estimated masses, volume, waste arising timescales and radioactivity of wastes in the main texts and in the individual waste streams.

6.4 LCA Sensitivity and Uncertainty Analysis

The current LCA standards (ISO 14040 2006 and ISO 14044 2006) discuss the requirements for sensitivity and uncertainty analysis of data suitability and completeness, life cycle inventory and impact assessments models and assumptions,

especially in comparative studies. Software and embedded databases, discussed below, make clear statements on the uncertainty of the data presented and the limitation of use based on geography, technology and modelling.

As stated previously LCA is an industry tried and tested methodology to published and developing ISO standards. Also, SimaPro 7.3 and the Ecoinvent 2.2 database are industry standard tools and are compatible with the ISO standard approach for LCAs The LCA models were built using data from the published source and scenarios from existing studies, for example the LLW metallic waste treatment in Shipton and Falconer 2013 and the DNLEU studies by NDA 2007, Hickford et al. 2012, Wilson et al. 2012.

6.5 LCA Assumptions

Specific LCA Modelling Assumptions

A number of general assumptions were common to the scenarios. LCA materials and process data for UK radioactive waste disposal facilities environmental impacts are not available. Four types of radioactive waste disposal facilities are of interest in this research, VLLW disposal, LLW disposal, HAW disposal to a future GDF and HAW disposal to a potential near surface long-term management facility as discussed in Chapter 2. The availability of environmental safety case reports for the three VLLW disposal facilities in England is unknown. Extensive environmental safety case reports are available for the LLW Repository near Drigg in Cumbria (LLWR 2011a, 2011b and 2011c) and for the new LAW disposal facility at Dounreay (e.g. Crawford 2010, DSRL 2011a and 2011b). There is currently no UK HAW disposal facility. Generic design requirements and assessments for a future UK GDF for English and Welsh HAW is well developed, but the requirements for a near site near surface long-term management facility for Scottish HAW are only at a preliminary stage of development. It is also worth noting that the majority of Scottish HAW (by activity) ends up at Sellafield and will likely go to a GDF. Although some environmental safety case reports are available it is beyond the scope of this research to convert the data in these reports to LCA databases. Hence it was decided to use the radioactive disposal facility data in the Ecoinvent database.

The Ecoinvent database has extensive data estimated for Swiss LAW, LLW/short-lived ILW (i.e. L/ILW) and long-lived ILW/HLW (i.e. ILW/HLW) disposal facilities (Dones et al. 2012). The low activity disposal impact includes data estimated only for land use and energy to dig the near surface trenches for the repository. In the absence of other data it was assumed that the Ecoinvent Swiss LAW disposal data could be used for UK VLLW disposal.

The L/ILW and ILW/HLW disposal impact data are for the material, energy and transport for creating and backfilling the shafts and tunnels, wastewater from mining and waste disposal plus land use and buildings for surface facilities. The Swiss ILW/HLW disposal impact data are broadly comparable with UK ILW disposal as both are engineered geological vaults (Wallbridge et al. 2012a). In the absence of other data it was assumed that the Swiss ILW/HLW disposal data could be used for UK ILW disposal.

The Ecoinvent database estimated data for a Swiss LLW/ILW repository are for a proposed engineered geological facility (Dones et al 2012), however, the UK LLW repository (Drigg) is an engineered near surface facility. Hence the Swiss L/ILW disposal facility data are not entirely comparable with the UK LLW repository. However, in the absence of other data it was assumed the Swiss L/ILW disposal impacts could be used for UK LLW disposal.

There is no near-surface disposal facility data estimated for a Swiss ILW or other HAW in the Ecoinvent database that could be used for the potential disposal of DNLEU a few tens of metres below the surface as suggested in NDA 2013b. It was therefore decided that Ecoinvent the Swiss L/ILW and the ILW/HLW disposal data could be assumed as the range of disposal impacts for the near surface disposal of DNLEU in Chapter 9.

General Data Assumptions for WAGR Boilers and UK Radioactive Metals Inventory

The transport distance between nuclear sites and the LLW repository and between nuclear sites and the Studsvik UK metal recycling facility in Cumbria was calculated

by using a weighted average of the distance for individual sites. The weights used were the proportion by mass of the radioactive metals at each site. This approach was also used to represent the distance to a future GDF, assumed to be near Sellafield (at time of the initial modelling).

In addition to the Studsvik UK metal recycling facility, and decontamination facilities at some UK nuclear sites, UK nuclear operators have used international treatment and melting facilities in Germany, Sweden and the USA, but not those in France and the Russian Federation. It was assumed that Studsvik's facility at Nykoping in Sweden was representative of a European LLW metals treatment/melting facility.

Waste container impacts were anticipated to be important from Section 4-3, hence a container production impact was added to the container metal impact. It was suggested by a highly experienced LCA practitioner (Edwards 2011 pers. comm.) 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 approach 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.

Scenario specific assumptions for the WAGR boiler and UK radioactive metals inventory LCA models are given in the Chapters 7 and 8 respectively. Scenario specific assumptions for depleted uranium storage and disposal are given in Chapter 9.

6.6 Cost Estimation

A detailed cost analysis of the disposition options for radioactive metals would need to consider three levels of costs (Nieves and Chen 1995, Yuracko et al.1997a and 1997b, Curlee and Yuracko 2000). The first level is direct financial costs and

benefits for waste recovery, treatment, disposal and recycling. The second level is the quantifiable external costs of environmental, health and safety impacts for the public and workers, including the indirect costs such as insurance, liability and accident mitigation and the indirect benefits such as local jobs and economy impacts. The final level is the non-quantifiable costs such as the level of support for Government initiatives, and effect on public perception and acceptance of moving and treating radioactive materials from nuclear facilities. Such an exhaustive cost consideration would represent a significant investment in time and effort and is beyond the scope of this research. Hence, a simpler approach is considered here that allows broad indicative first level direct financial costs to be estimated for each of the three research areas.

Any cost estimates need to consider whether or not to use discounting, i.e. applying a time weighting value to money, and the level of discounting applicable. This is a particular issue for the very long timescales for the treatment, storage and disposal of radioactive wastes from UK legacy nuclear facilities. The ethics and uncertainty in monetary evaluation in general and choice of discount rates in particular were noted as contentious issues (Solberg-Johansen 1998). There are four possibilities: a) infinite discounting representing no moral obligation to future generation, b) positive but finite discounting representing limited obligation to the future, c) zero discounting representing the same obligations to current and future generation, and d) negative discounting representing more obligation to future generations.

The UK Treasury guidance on discount rate for public sector projects is currently 3.5% for the short to medium term (30 years) and a declining discount rate thereafter (Wooders et al 2007b, HM Treasury 2011). The UK Health and Safety Executive support the 3.5% discounting rate up to 30 years, but recommend 1.5% for any longer projects (HSE Cost Benefit Analysis (CBA) checklist on www.hse.gov.uk/risk/theory/alarpcheck.htm, NERA 2007 and ONR 2013). The NDA adopt a 2.2% discounting rate for the disposal costs of spent nuclear fuel in a future GDF (NDA 2012c). Hence, there is a broadly consistent approach to discounting in UK radioactive waste management.

Research on discounting of significant environment projects proposes a declining rate - 4% for the first 5 years, 3% for years 6 to 25, 2% for years 26 to 75, 1% for years 76 to 300 and 0% after 300 years (Weitzman 2001). This approach is supported by other research (e.g. Pearce et al. 2003, Cunningham 2009, Groom et al. 2005 and Hepburn 2007). However, the debate on discounting rates continues and there is still no generally accepted method for discount over long periods. The choice on discounting is therefore the prerogative of the researcher. The costs estimates used in this research are undiscounted, thus allowing decision makers to choose their own discounting rate subject to the prevailing financial, political and social conditions.

Public Domain LAW Management Costs

Costs for UK LAW management are readily available. They are clear and transparent for containers, processing, transport, waste activity supplement and disposal charges and metallic waste treatment. The data are available from documents on the LLW Repository Ltd website (www.llwrsites.com) and are regularly updated. The documents include various waste services contract prices and future estimates in joint waste management plans for the NDA sites (e.g. Shipton and Falconer 2013). These data are used to estimate the cost of disposal and treatment for recycling or re-use of LLW and VLLW radioactive metals in Chapters 7 and 8.

Public Domain HAW Management Costs

Cost for HAW containers, conditioning, storage, transport and disposal are less clear as a definitive position for storage and disposal is still developing. There are several historical assessments but it proved difficult to obtain details on current costs as they are commercially sensitive for waste producers and container manufacturers. However, the NDA kindly supplied documents on the costs of disposal of spent nuclear fuel and reactor core graphite to a GDF to support this research (NDA 2010b, 2012a and 2012b). Additional documents were also kindly supplied by the Committee on Radioactive Waste Management (CoRWM) (e.g. DECC 2009). Both these data sources were used to develop indicative costs for the storage and disposal of HAW in shielded packages (which can be contacted handled) or unshielded packages (requiring remote handling) for storage and disposal (NDA 2007, DECC

2009, NDA 2012a). Where applicable, they were also used to estimate potential cost savings for treatment of ILW toward recycling or re-use rather than disposal.

Summaries of the cost data used in the research is given in Sections A.2 and A.3.1for waste containers and treatment, Section A.3.2 for a proposed American decicated steel mill for radioactive metals and Section A.3.3 historical enrichment plant decommissioning.

6.7 Chapter Summary

SimaPro 7.3.3, with the embedded Ecoinvent 2.2 database, was the chosen software package for the research. The Eco-Indicator 99 life cycle impact assessment method was adopted for investigating the environmental damages to human health, ecosystem quality and resource depletion for UK radioactive wastes to help inform future decision making within the UK nuclear industry. The three cultural perspectives of Eco-Indicator 99 offer the opportunity to model a range of opinions on radioactive waste management for a mixed group of stakeholders. However, the default Hierarchist/Average option was chosen as the baseline approach for each scenario investigated in Chapters 7, 8 and 9. The fundamental boundary condition for each end-of-life scenario is that the radioactive waste already exists and nothing can be done to change that. The inventory data is based on the published UK radioactive waste inventory data for legacy radioactive metals, the current published estimates of legacy civil depleted, natural and low enriched uranium, process and materials data in the Ecoinvent 2.2 databases plus transport distance and national power mixes for UK and international radioactive metals treatment facilities. Public domain cost data can be used to provide undiscounted costs estimates for each end-of-life scenario to be modelled. The results of the LCA modelling and the cost estimates radioactive metals disposal and treatment for recycling and reuse are presented in detail in Chapters 7 and 8. Chapter 9 present the LCA modelling and costs results for the long-term interim storage and disposal of depleted uranium.

7 Windscale Advanced Gas-Cooled Reactor Heat Exchanger Case Study

Chapter 6 outlined the general research method to address the research objectives. This chapter discusses the background to and the disposal and treatment for recycling of the four Windscale heat exchangers (boilers) of Advanced Gas-cooled Reactor (WAGR) at Sellafield in Cumbria. The boilers were chosen as they represented a well defined case study of four large nuclear plant items with published data. The case study was used to gain experience in building LCA models, as a prelude to investigating the potential environmental and costs impacts of the disposal and treatment for recycling or re-use of all UK low and intermediate level waste (LLW and ILW) metals in Chapter 8.

The chapter includes a brief description of the WAGR boilers and their direct disposal whole to the LLW Repository near Drigg in Cumbria in 1995 (Dixon 1999, Bayliss and Langley 2003). It also discusses the hypothetical end-of-life packaged disposal and boiler steel treatment for recycling at the Studsvik radioactive metals melting facility at Nykoping in Sweden. The resultant potential environmental impacts for the two disposal and two treatment and recycling scenarios are presented plus examples of the uncertainty and sensitivity analyses done. Disposal costs estimates for a range of waste package loadings (i.e. tonne/waste package) are also given.

7.1 WAGR Background

This section presents a general background and overview of the WAGR boilers and outlines some of their radiological issues.

7.1.1 General Information

The WAGR was an experimental 100 Mega Watt (thermal) reactor plant at Sellafield in Cumbria. It operated between 1963 and 1981 and has been decommissioning since its nuclear fuel was removed in 1983 (Crossley and Wakefield 1991, Dixon 1999,

McKibbin 2012). It was a prototype development between the Generation I Magnox reactors and the Generation II AGRs. Early decontamination and dismantling (1989 – 1993) was done as part of the Commission of the European Communities research and development programme on decommissioning nuclear installations (Crossley and Wakefield 1991, EC-CND circa 1995). The WAGR decommissioning programme has taken about 30 years and cost approximately £110 million (McKibbin 2012). The project has contributed significantly to the research and development of nuclear decommissioning nationally and internationally through the European Commission and Organisation for Economic Cooperation and Development programmes.

7.1.2 WAGR Boilers Overview

The WAGR nuclear reactor core, boilers and associated plant were originally housed in a 41m diameter containment building. Four identical boilers were used to raise steam to generate electricity. The boilers stood vertically in pairs, either side, of the reactor core and were connected to the core by short horizontal gas ducts towards the base of the boilers. The boilers were housed in concrete biological shields above and external to the core concrete biological shield as shown in Figure 7-1.

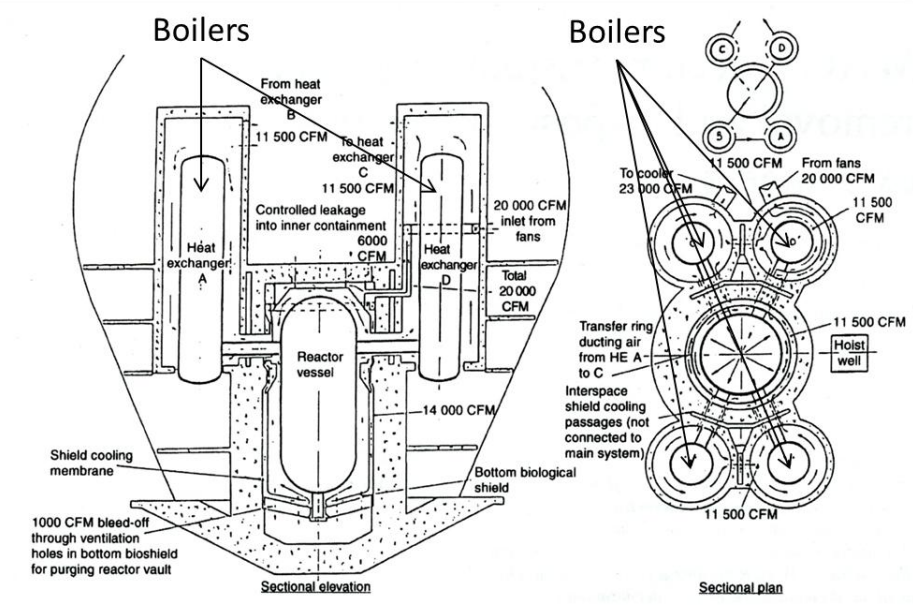

Figure 7-1 WAGR Boilers Orientation (reproduced from Dixon 1999)

Each boiler pressure shell was constructed of carbon/manganese steel cylinder with domed ends made in three flanged sections and bolted together as shown in Figure 7-2 (Dixon 1999). The boilers were 20.6m high, with an internal diameter of 3.35m, a wall thickness between 36.5mm and 63.5mm and a total free volume of 160m^3 (Crossley and Wakefield 1991, Dixon 1999). The weight of each shell was ~100 tonne. Each boiler shell was lagged with asbestos insulation during operation which was removed prior to disposal.

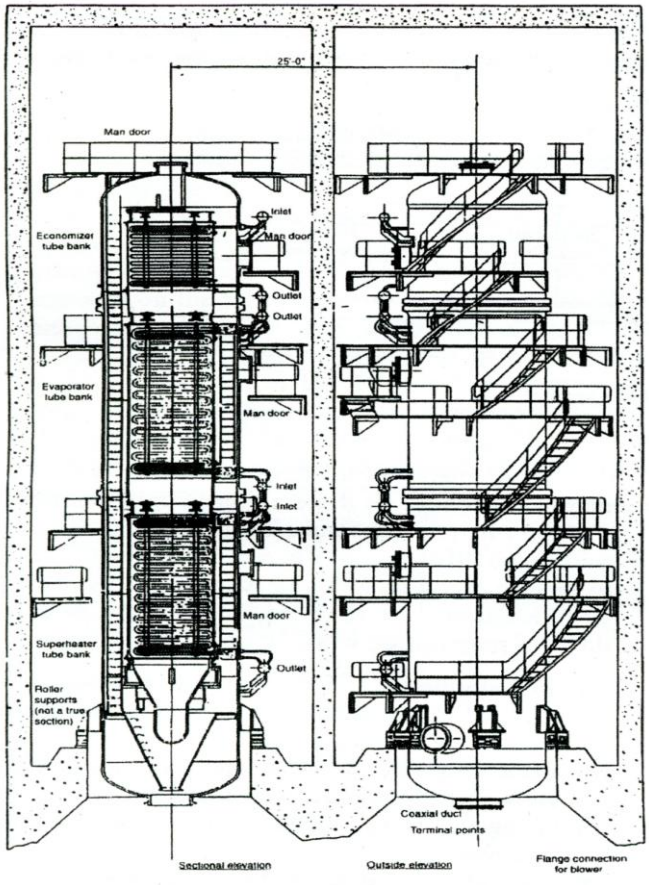

Figure 7-2 General WAGR Boiler Arrangement (reproduced from Dixon 1999)

There were three tube banks within each boiler, two sets of tubes were finned carbon steel tubing and the remaining set was plain chromium/molybdenum steel tubing. The tube banks were surrounded by mild steel ducts. During operation hot gas flowed up past the tubes inside the duct and downwards outside the duct. The duct was insulated with stainless steel plate. The tube banks and ducts weighed ~70tonne

(Crossley and Wakefield 1991). Ladders, platforms and other equipment inside the boiler shell weighed about 18tonne (Dixon 1999).

7.1.3 WAGR Boilers Radiological Issue

Contamination

The boilers were shielded from the reactor by their own biological shield and the core biological shield as shown in Figure 7-1 and were therefore not subjected to a significant neutron flux. Hence the boiler steel was not neutron-activated (Crossley and Wakefield 1991). In neutron activation material becomes radioactive due to the interaction with the neutron flux.

Part of the experimental work at the WAGR was "… the deliberate operation with failed fuel elements in the reactor core" (Crossley and Wakefield 1989). Hence nuclear fission products such as Caesium 134 and 137 (Cs134, Cs137) or activation products such as Cobalt 60 (Co60) and Manganese 54 (Mn54) from the reactor core were transported in the coolant gas flow and deposited on the gas-side surface of the boiler tubes. This resulted in 10µm to 150µm surface contamination layers on sections of the boiler tubes (Crossley and Wakefield 1991). These deposits gave estimated radiation levels in excess of 15 milli-Sieverts per hour ($mSV.h^{-1}$) at the boiler tube surfaces which was too high for manual dismantling of the boilers without either some form of decontamination or the need for requiring remote handling systems. This radiological surface contamination of the boiler's internal structures presented significant issues for decontamination, disposal or recycling of the boilers.

Chemical Decontamination

Decontamination of the boiler internals was developed as part of the combined United Kingdom Atomic Energy Authority and European Commission project to acid clean the WAGR boilers to manual handing levels (Wakefield 1988, Crossley and Wakefield 1991). The target radiation level at the tube surface was set $0.5mSV.h^{-1}$ or $0.1mSV.h^{-1}$ at the normal working distance (Wakefield 1988, Crossley and Wakefield 1989 and Crossley and Wakefield 1991). The references discuss the entrapment of the Cs134, Cs137, Co60 and Mn54 in the magnetite (Fe_3O_4) layers

that formed on the gas-side of the boiler tubes during operation and the options for chemically removing the contamination to achieve the target radiation levels.

Commercial chemical decontamination options for water reactors already existed at the time but were rejected because of secondary effluent problems for WAGR. UK laboratory scale tests showed the effectiveness of hydrochloric acid decontamination, however, the acid was not acceptable for the Sellafield active effluent treatment plant. Cleaning with dilute hydrochloric and citric acid, or nitric acid and citric acid were chosen for laboratory tests of boiler samples. Hence a nitric acid and citric acid recirculating spray system chosen for the full scale demonstration experiment on one WAGR boiler tube bank. It was concluded by Crossley and Wakefield 1991 that the demonstration test was successful as the chemicals used were effective and the method "… was simple to apply and cost effective". Further work continued to improve the decontamination system but only one of the four boilers benefited from this decontamination process.

7.2 LCA Case Study End-of-life Scenarios

The following four LCA case study scenarios investigate the end-of-life dispositions options for radioactively contaminated metal, equipment and components of the four WAGR boilers. Hence, the functional unit for these LCAs was – the removal of the WAGR boiler radioactive waste steels from the Sellafield Site. Prior to current policies, strategies and plans these would have been classed as waste and decontaminated to exempt level, if viable on economic, project and radiological grounds, but more likely disposed to authorised repositories. The drive now, however, is to divert these metals from disposal to conserve the scares national resource, by treating the metals for recycling and re-use. The following scenarios use the disposition of the WAGR boilers as a means to identify the materials and processes resulting in significant environmental burdens and benefits for each disposition option.

7.2.1 Basic Scenarios

The four LCA scenarios representing the potential end-of life disposition options for the boilers were direct disposal, packaged disposal, bulk recycling and containerised

recycling. The focus of the scenarios is to assess the merits of the end-of-life disposal versus recycling of the boiler steel. As discussed in Section 6.2 the scenarios do not include the production, use and maintenance of the boilers since they already existed in their current form, so only treatment, packaging, transport and disposal or recycling are considered. Each scenario is discussed in turn in this section.

Direct Disposal

This scenario considered the potential environmental impacts of what was actually done, i.e. the four boilers were in fact disposed whole to the LLW Repository (Drigg) (Dixon 1999, Bayliss and Langley 2003). The boilers were removed from the WAGR reactor building, taken individually by a specialised road transporter to Drigg. Once at Drigg the boilers were placed in an engineered vault. They were internally grouted with cement and encased in cement as required by the LLW Repository environmental safety case (LLWR 2011a). At the time whole disposal was unusual and authorisation was granted on the grounds of worker dose uptake was a low as reasonably practicable, disposal time could be halved and the costs reduced (Bayliss and Langley 2003). Also, at the time the recycling of large contaminated items was not a common option.

Direct disposal of large items to the LLW Repository (Drigg) is still allowed under the current UK solid LLW policy and strategy if it is shown to be the best available technique or best practicable environmental option. Hence, the direct disposal scenario was retained.

Packages Disposal

This scenario estimates the potential environmental impacts of the most common option at the time, i.e. in-situ chemical decontamination of the boilers prior to their removal from the reactor building and cut up for disposal. It was assumed that this decontamination could have been done as part of the post operation clean out of the boilers and hence was not included in the model. Decontamination is necessary to remove any activation products and/or fission products in the oxide film on boiler tubes or the shell structure and hence reduce the radiological issues for staff cutting up and handling the steel for disposal. It was also assumed that the boiler segments

were packed in half height ISO (HHISO) freight containers and transported to the LLW Repository (Drigg). It was further assumed that the HHISOs were grouted at Drigg and placed in an engineered vault.

The original estimate of HHISOs required for packaged disposal of the boilers was unknown. The planning norm for disposal in the LLW Repository Ltd websites documents is 10tonne/HHISO (e.g. Shipton and Falconer 2013). The Magnox boilers disposal from the UK radioactive waste inventory (NDA and DECC 2011) and the radioactive waste metal strategic best practicable environmental option BPEO study (Studsvik 2006b) give waste loadings from 11tonne to 22tonne per container. Hence an average loading of 15tonne/HHISO was assumed for packaged disposal, giving nominally 50 HHISOs.

Bulk Recycling

The bulk recycling scenario modelled the potential environmental impacts of what could hypothetically be done now at the Studsvik Nykoping facility. It was assumed that a transport safety case could be made for transporting the boilers whole to Sweden. This was on the basis that the thick boiler shell would effectively act as the transport package containing the internal radiological contamination during transport. This type of safety case approach is now common practice for the transport of large items to Nykoping for treatment and recycling (e.g. NDA 2014a). It was also assumed that each chemically decontaminated boiler was wrapped in a PVC tarpaulin and transported from WAGR to the nearest port using a specialised road transporter. They were then assumed to be loaded onto Studvik's specialised ship and all four boilers transported direct to Nykoping. It was further assumed that each boiler underwent size reduction, was further decontaminated as necessary and was melted into exempt level ingots for recycling into the Swedish iron and steel markets. It was finally assumed that all residual treatment wastes (i.e. blasting grit, metal swarf from cutting, crushed melt slag, protective clothing, dust filters and any other secondary materials) were packed in two HHISOs and returned to the UK for disposal at the LLW Repository (Drigg). The HHISOs were assumed to be grouted and disposed in an engineered vault.

Containerised Recycling

The containerised recycling scenario estimated the potential environment impact of current practice at any of the international treatment facilities, but Nykoping was taken as the baseline case. It forms an alternative hypothetical recycling scenario in which the chemically decontaminated boilers were removed from the reactor building and cut up at WAGR. It was assumed the boiler segments were packed in HHISOs transported by road to Hull, then by commercial ferry to Stockholm and finally by road to Nykoping. The boiler segments were assumed to be further decontaminated and size reduced for melting. The melt ingots were assumed to be sold in the Swedish iron and steel markets as before. The secondary waste was assumed to be returned, by the same route in two HHISOs, to the UK for disposal at Drigg under the same assumptions as above.

In the containerised recycling scenario nominally 50 HHISO trips were assumed for the transporting of boiler segments from the UK to Sweden for processing. It was assumed that the treated metal ingots were sold in the Swedish metals market but all residual secondary waste was returned to the UK in two HHISOs, consistent with the bulk recycling scenario.

Early development stage data and results (indexed in Table D-1) and main case study data and results (indexed in Table D-2), supporting the WAGR scenarios are summarised in the Microsoft Excel spreadsheets on the compact included in Appendix D.

7.2.2 Scenario Assumptions

The assumptions regarding the use of the Ecoincent database estimated Swiss LLW disposal facility impacts as a first order approximation to disposal at the LLW Repository (Drigg) and the weighted average transport distances for disposal and treatment for recycling were discussed previously in Section 6-4.

A number of other general assumptions were also made and are given in Box 7-1.

Box 7-1 General Windscale AGR boiler scenario assumptions

1) The environmental impacts and costs of the preparatory work to remove the boilers whole from the reactor building and associated secondary LLW (Dixon 1999) were common to all scenarios hence could be neglected when comparing scenarios,
2) All international treatment facilities were available and the chemically decontaminated WAGR boilers met the treatment facilities' conditions of acceptance.
3) A specialised road transport vehicle was available for the transport of the boilers whole from WAGR to the nearest suitable UK port for onward transport to Nykoping,
4) The environmental impacts and costs of modifications to local infrastructure needed to transport the boilers whole from WAGR to the nearest suitable UK port were negligible,
5) Stusdvik's specialised ship, or equivalent approved vessel, was available to transport the boilers whole from the UK port to the Nykoping dock
6) A suitable transport safety case could be made for the delivery of the boilers whole to Sweden for treatment and recycling,
7) The Nykoping dock crane and site transport were available and capable of handling the WAGR boilers, but that their environmental impacts were negligible,
8) Since HHISO freight containers are approved for transport there was no issues with transporting segmented boiler sections to the international treatment facilities, or the return of secondary LLW from treatment to the UK for disposal, using commercial ferries,
9) All liability/insurance cover required for international transport was available for each recycling option,
10) All melt ingots produced at Nykoping were exempt level and sold in the Swedish iron and steel market, i.e. no steel was rejected, or decayed stored at treatment facilities until recycling was possible, or returned to the consignor,
11) The cost of secondary residual waste repatriation to the UK for disposal at the LLW Repository (Drigg), or disposal in the USA, was included in the treatment contract,
12) All best practicable environmental option, best practicable means, best available techniques studies and as low as reasonably practicable radiological studies were done and their conditions complied with,
13) All International Atomic Energy Agency transport regulatory requirements and UK Environment Agency trans-frontier shipment authorisation requirements were satisfied for HHISO and whole boiler transport for disposal or recycling,
14) There were no accidents or incidents that delayed the treatment and recycling of metals.

Modelling Assumptions

In the absence of published data for WAGR boiler size reduction in the UK or at Nykoping, grit blast decontamination or equivalent at international treatment facilities, melt slag and slag crushing, engineering calculation were made based on the metal milling, dressing and sintering processes embedded in SimaPro. It was assumed that these were reasonable first order approximations for potential environmental impacts as the calculations were based on details from Dixon 1999, Crossley and Wakefield 1991 and 1989, plus Wakefield 1988. To address the

uncertainties in these estimates some additional sensitivity tests done and are reported in Section 7.5.

Waste package environmental and cost 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 average metal working impact could be used as a first level estimate for the decontamination process impact. It was assumed that 5% of the boiler steel mass would be allocated the average metal working impact to represent LLW decontamination, therefore consistent with the 95% recycling assumption as discussed in Chapter 6.

Example SimaPro Model

As discussed in Section 6.1, SimaPro models are constructed by combining assemblies and sub-assemblies with treatment and disposal processes. Different assemblies and sub-assemblies were generated for the waste metals and waste containers and linked with size reduction, decontamination, melting, slag and slag crushing processes, plus disposal to generate a life cycle.

An annotated example for the containerised recycling model is presented in

Figure **7-3**. It shows an example of the life cycle process tree for containerised recycling with the transport package assembly and container and waste sub-assemblies and processes and the waste treatment and disposal processes.

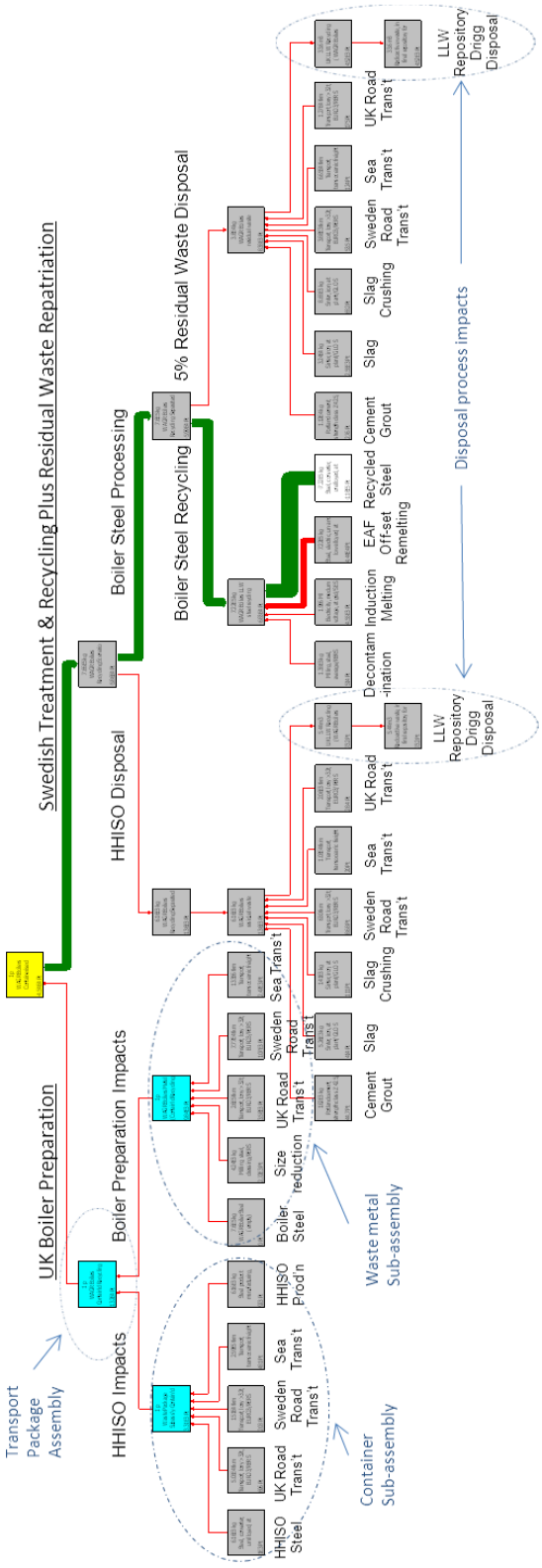

Figure 7-3 Example of SimaPro Process Tree for Containerised Recycling

The yellow box represents the net life cycle. On the left hand side of the figure are all the impacts prior to the point of recycling or disposal. The blue boxes represent the impacts of generating the waste container (HHISO) sub-assembly and the impacts of preparing the boiler steel for the waste metal sub-assembly. These sub-assemblies constitute the transport package assembly as highlighted in the figure. The associated processes in grey are the HHISO steel and production impacts plus boiler steel size reduction in the UK combined with the outward road and sea transport impacts.

The grey treatment processes on the right hand side represent boiler steel decontamination, induction melting, avoided unalloyed steel with electric arc furnace off-set for re-melting the ingots. Also included are the slag and slag crushing impacts, the repatriation of the secondary waste and HHISOs by road and sea from Sweden to the UK, and finally the cement grouting and disposal of the waste and containers at the LLW Repository (Drigg) - as highlighted in the figure. The treatment processes are split between the 95% recycled boiler steel, the 5% secondary waste and its return to the UK. The total disposal is equivalent to two HHISO with a combined disposal volume of $39m^3$. The adverse environmental impacts are shown in red and the environmental benefit (giving a negative value) of recycling the ingot is shown in green. The thickness of the red and green lines is proportional the size of the impacts.

7.3 Case Study Results

This section presents the Eco-indicator 99 (Hierarchist/Average) potential environmental impact results for the direct disposal, packaged disposal, bulk recycling and containerised recycling of the WAGR boilers including consideration of different international treatment facility impacts. A cost comparison of different disposal and treatment costs is also given.

7.3.1 Main Scenario Environmental Results

The results of the four scenarios are presented in Figure 7-4 and Figure 7-5. The Eco-indicator 99 (Hierarchist/Average) outputs for each scenario are in the form of weighted points, since these results are not in themselves meaningful and only for the

purpose of comparison the results are presented here as percentages of the total direct disposal impact (TDDI). Positive percentage values represent adverse environmental impacts; negative values represent environmental benefits.

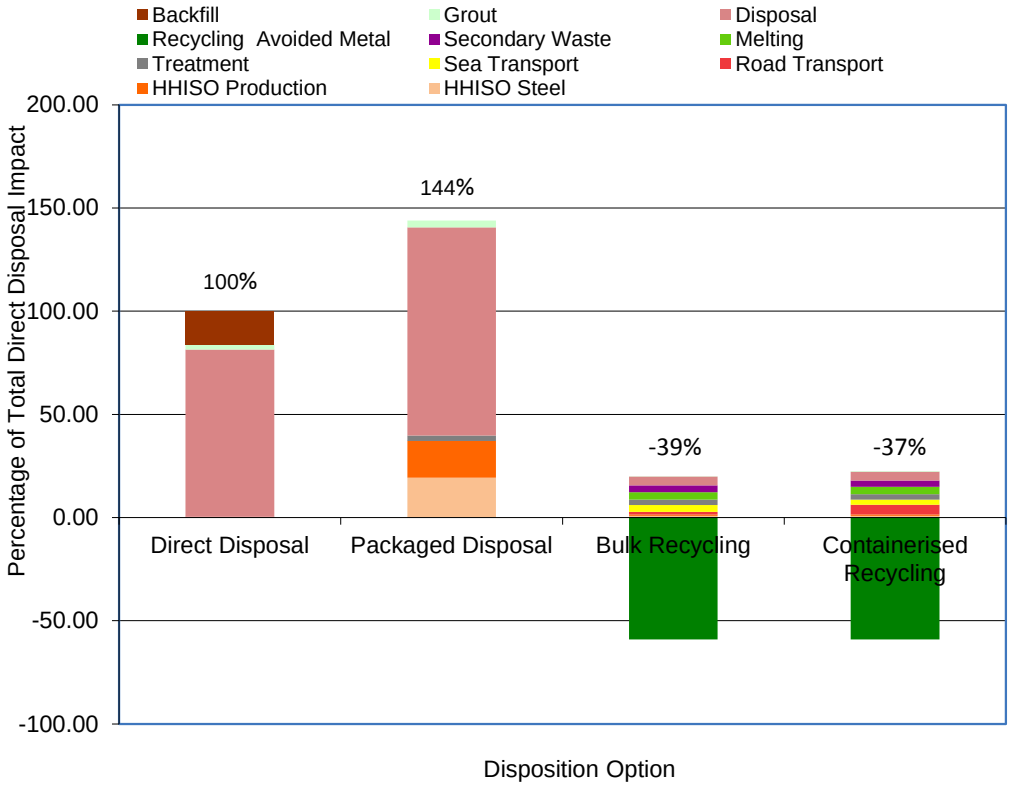

Figure 7-4 Environmental impacts of the two disposal and two recycling end-of-life options for the WAGR boilers. The results are in percentage of total direct disposal impact (TDDI) using the Eco-Indicator 99 (Hierarchist/Average) method. Direct disposal is boilers disposed whole to the LLW Repository (Drigg), Packaged disposal is boiler segments in HHISOs disposed whole to Drigg, Bulk Recycling is the boilers transported whole to Nykoping for treatment and recycling, Containerised Recycling is boiler segments sent to Nykoping in HHISOs for treatment and recycling. A negative value represents an environmental benefit. Backfill only applies to the external grouting of boilers disposed whole in the direct disposal scenario.

Figure 7-4 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 ~45% increase in disposal impact compared to direct disposal. The results from ~25% increase in disposal volume from using HHISOs, ~18% increase for HHISO metal impacts and ~19% for HHISO production impacts. About 17% of the impact for direct disposal resulted from the cement backfill and internal grouting of the 4 boilers, however this is not relevant for packaged disposal and is not included in the scenario, hence off-sets the increases from packaging. Container impacts are significantly larger than transport impacts and other process impacts across all four scenarios.

The figure also 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% TDDI.

The waste container impact for containerised recycling was insignificant as it was assumed that two HHISOs were reused for the return of secondary waste to the UK for disposal at the LLW Repository (Drigg). Recycling the melt ingots in the Swedish iron and steel industry results in a benefit equivalent to about -59% TDDI (shown as negative values in Figure 7-4) representing the avoidance of future virgin materials or other conventional scrap iron. Hence the net benefit of treatment with recycling was about -37 to -39% of TDDI.

The potential environmental impact category results for each of the four scenarios are presented in Figure 7-5.

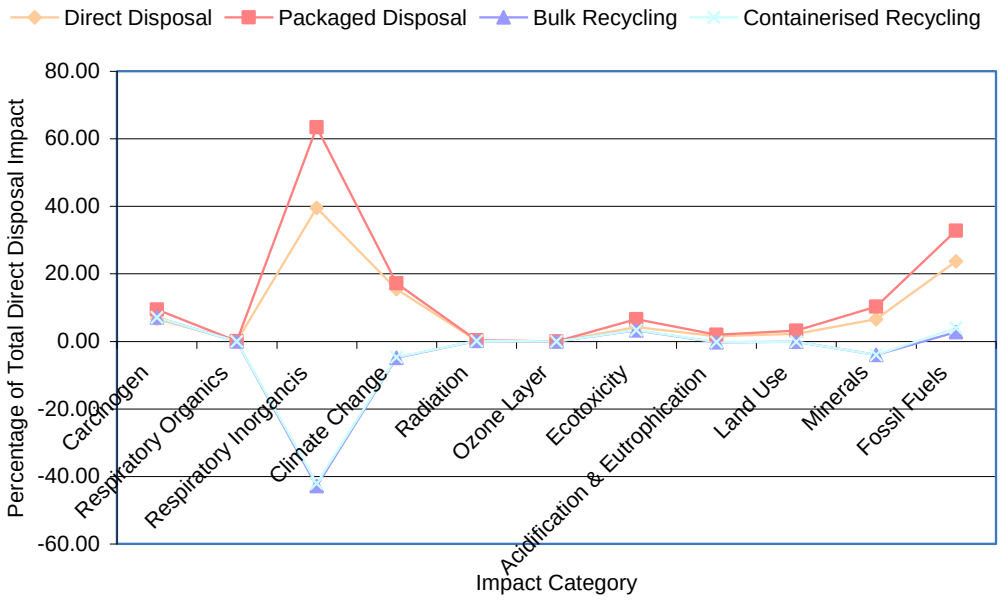

Figure 7-5 Eco-Indicator 99 (Hierarchist/Average) impact category results for each WAGR boilers disposition scenario. The results are presented in percentage of total direct disposal impact (TDDI). The environmental impact categories are as shown in Figure 3-5. Positive values represent adverse potential environmental impacts. Negative values represent potential environmental benefits.

As expected the packaged disposal impact category results are all higher than the direct disposal impacts, particularly for respiratory inorganic compounds, fossil fuels and minerals. The overall impact is dominated by the respiratory inorganic

compound impacts for carbon monoxide (CO), dust particles, nitrous oxides (e.g. NO_2 and NO_x), sulphurous oxides (e.g. SO_2 and SO_x) and ammonia (NH_3) discharged to air for iron and steel making, as discussed in Chapter 5, plus the excavation and backfilling of the LLW disposal facility. This is followed by fossil fuel impacts calculated from energy use and by climate change impacts from emissions of greenhouse gases.

The minerals, carcinogen and ecotoxicity impacts 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 (~42%), 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.

7.3.2 International Treatment

Comparison of Extant Treatment Facilities

The previous results concentrated on the treatment of the WAGR boilers at the Studsvik Nykoping facility in Sweden. However, there are four other international treatment facilities that could conceivably accept containerised boiler segments for treatment for recycling and re-use. As mentioned previously the SOCODEI facility at Marcoule in France has not yet been used by UK nuclear operators and the ECOMET-S facility near Sosnovy Bor in the Russian Federation cannot yet accept UK LLW metals. They are included here for completeness.

LCA models were generated for each facility. They assumed the same number of HHISO packages for boiler segments were sent to each facility and the treatment and melting facilities were identical. This is a simplification as arrangements and operation of each facility are slightly different, but it was deemed acceptable as a first order approximation. Where possible the electricity data (medium voltage plus imported electricity) for each country was taken from the Ecoinvent database

embedded in SimaPro as discussed in Chapter 6.. This was not possible for the Russian Federation. The OECD/IEA website suggested that Canada was probably the best proxy for Russia, unfortunately there is also no electricity data for Canada in the Ecoinvent database. Hence Germany was chosen as the proxy for the Russian Federation based on the comparison of net generation and net consumption data from the literature and websites (e.g. International Energy Agency (iea) 2012, www.iea.org/stats/electricitydata.asp?COUNTRY_CODE_RU, and www.iea.org/countries/non-membercountries/russianfederation).

The transport distances were based on standard road transport from WAGR to Liverpool and commercial sea transport to ports convenient for the treatment facilities. Marseille was chosen for the SOCODEI facility at Marcoule in France and Rotterdam for the Siempelkamp CARLA facility at Krefeld in Germany. St Petersburg was chosen for the ECOMET-S facility at Sosnovy Bor and Southampton, Virginia, for the Energy Solutions facility at Bear Creek. Standard road transport from these ports to the final treatment facility location was further assumed. The road and sea distances were taken from internet websites (e.g. www.theaa.com/route.planner, www.beyond.fr/travel/times.html, www.searates.com/reference/portdistance, www.sea-distance.org and www.mapcrow.info).

It was further assumed that secondary waste was returned to the UK for disposal from the treatment plants in France, Germany and the Russian Federation, but disposed in America as part of the Bear Creek solid waste disposal authorisation. This was due to the different melting regime at Bear Creek.

Comparison of International Treatment Facility Results

A comparison of the material and process potential environmental impacts for the five international treatment facilities is presented in Figure 7-6. The focus is on the transport, treatment and disposal impacts using the different international facilities hence the avoided metals for recycling or re-use are not included.

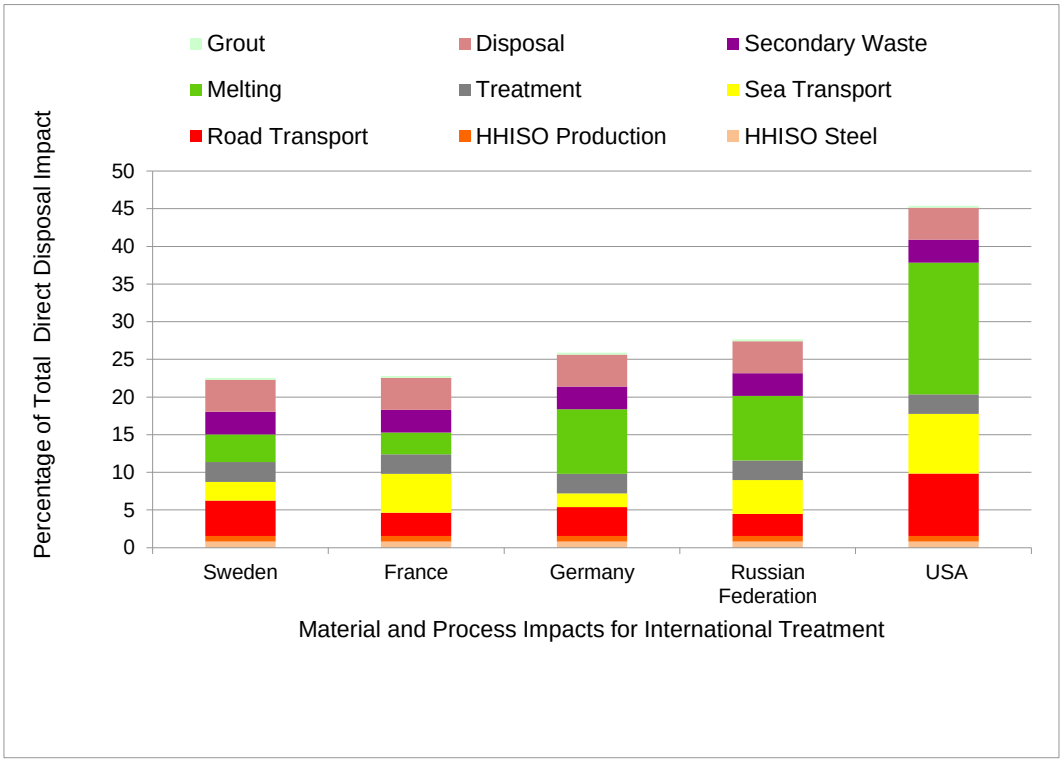

Figure 7-6 Comparison of materials and process Eco-Indicator 99 (Hierarchist/Average) potential environmental impacts for containerised recycling at existing international facilities excluding the avoided metal impacts for recycling. The avoided steel for recycling or re-use is excluded from the figure. The energy data are the Swedish, French, German and American medium voltage plus imported electricity from the Ecoinvent database. There are no equivalent Russian Federation data in Ecoinvent hence the German data is used as a proxy. The results are in percentage of total direct disposal impact (TDDI).

The highest potential environmental impact for an international facility was for America. The combined treatment, transport and disposal impacts were about 45% TDDI compared to ~23% TDDI for treatment in Sweden as discussed earlier for Figure 7-4. The French facility total impact was essentially the same as Sweden. The German and Russian Federation total impacts were 26% and 28% TDDI respectively.

The America result was high due the large sea and road transport distance to the treatment facility and road transport of secondary waste to an American LLW disposal repository. The transport impacts for the European facilities (including the Russian Federation) were between 6% and 8%TDDI, the American transport impact was ~16%TDDI. The larger American result was also due to the electricity generation technology mix which results in a large impact from the energy used in melting compared to the other countries, i.e. but 18%TDDI compared to 3%-

4%TDDI for Sweden and France and ~9% TDDI for the Russia Federation and German. Although America has a large number of nuclear power stations and hydro power stations their energy mix is dominated by fossil fuel generation. Germany also has a higher proportion of fossil fuel generation. France has about 80% nuclear electricity generation and Sweden has similar high proportion of non-fossil fuel generation from nuclear and hydro power stations.

Potential UK Melting Facility

Studsvik UK Ltd operates a metal recycling facility at Lillyhall in Cumbria. If the facility nuclear site licence and solid, liquid and gaseous radioactive waste authorisation could be revised to include radioactive metal melting then a comparison could be made with the existing international facilities. Such a comparison is presented in Figure 7-7.

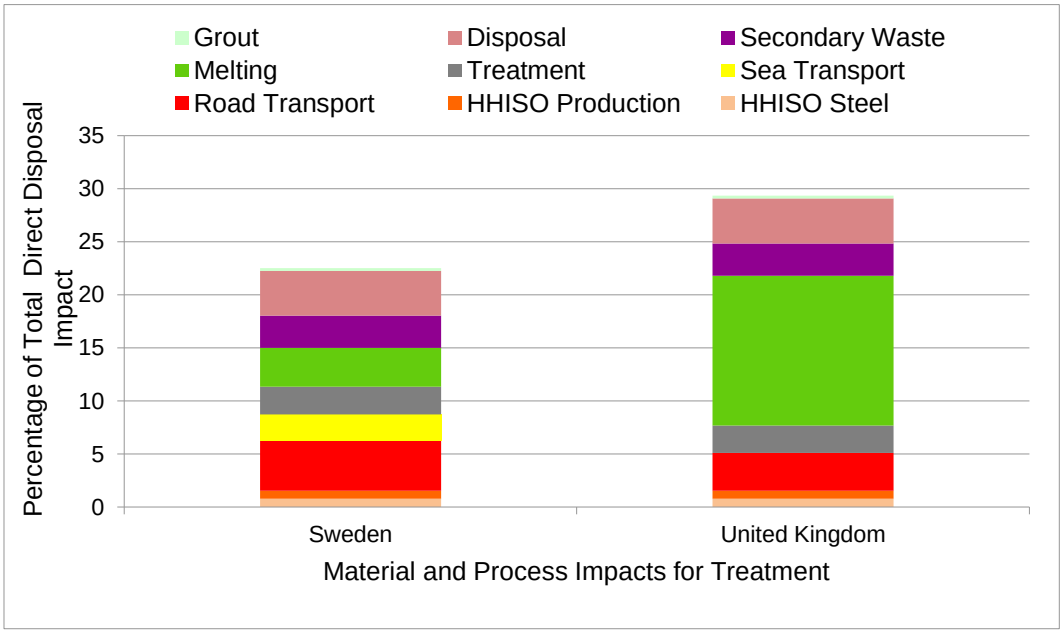

Figure 7-7 Comparison of the Eco-Indicator 99 (Hierarchist/Average) potential environmental impacts for containerised recycling at a potential UK metal melting facility and Nykoping in Sweden. The avoided steel for recycling or re-use is excluded from the figure. The Swedish and UK medium voltage plus imported electricity data are taken from the Ecoinvent database. The results are in percentage total direct disposal impact (TDDI).

Although there is no sea transport for the UK option, and road transport is essentially constant, the net materials and process potential environmental impact at Nykoping is lower than a potential UK melting facility at Lillyhall. This is due to the higher

fossil fuel electricity generation in the UK compared with Sweden (as modelled in the Ecoinvent database). Hence any future decision on developing a UK radioactive metals melting facility should depend on the cost of authorising, building, commissioning and operating a new facility and the availability of metals, plus benefits to the UK economic in terms of skills and employment rather than just environmental benefits.

7.3.3 Financial Cost Estimates for Disposal

The large range of possible waste package loading discussed in Section 7.2.1 suggested that disposal costs could vary considerably depending of the number of HHISOs needed and hence their disposal volume at the LLW Repository (Drigg). It was therefore decided to estimate the total disposal costs for the different possible packaging options as compared to whole boiler disposal. Figure 7-8 presents simple cost estimates for the two disposal options based on £2911/m^3 for the disposal volume cost at the time of calculation (LLW 2011b) and explores the sensitivity to different packing ratios (i.e 10 tonne, 15 tonne and 20 tonne per HHISO).

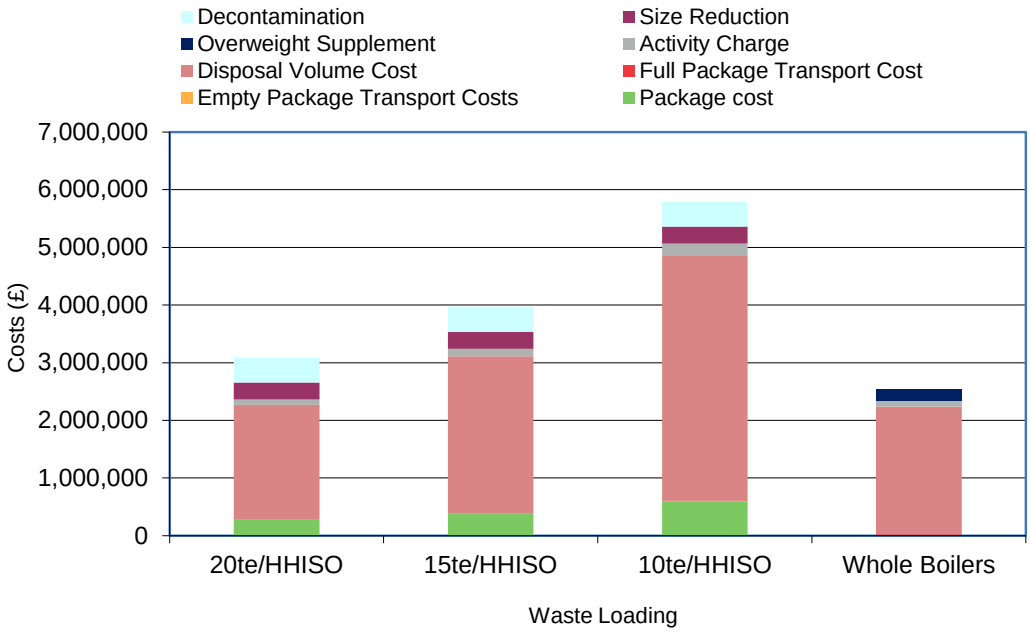

Figure 7-8 Comparison of estimated costs at 2012 for disposal options for the four WAGR boilers with disposal costs for a roughly equivalent mass of Magnox metals using planning norm values [4] of 10tonne/HHISO (i.e. 76 HHISOs), 15tonne/HHISO (i.e. 50 HHISOs) as an average waste loading and 20tonne/HHISO (i.e. 38 HHISOs) as a maximum waste loading. Whole boilers - each of the four boilers, with internal cement grout, placed together in a vault at the LLW repository near Drigg and backfilled with concrete.

Increasing the packing ratio from 10tonne/HHISO to 20tonne/HHISO requires additional decontamination and size reduction. The feasibility of higher packing ratios will depend on the level of contamination, its accessibility and the requirement to keep the radiation dose to the operators as low as reasonably practicable (ALARP). Large items disposed at the LLW Repository (Drigg) are subject to an overweight supplement charge. This was estimated as 8% of the direct disposal cost for whole boiler disposal only.

For simplicity decontamination and size reduction costs were estimated as fixed costs of £570/tonne and £385/tonne respectively from historical American studies of decontamination of redundant uranium enrichment facilities (Committee on Decontamination and Decommissioning of Uranium Enrichment Facilities (CDDUEF) 1996) and the feasibility of a dedicated steel mill for radioactive metals (United States Department of Energy (USDOE) 2001). There was good agreement of the cost estimates from both studies, but the higher cost was chosen for the calculations. The historical costs were converted to pounds Sterling and inflated to 2012 prices. In the absence of UK data these costs are considered to be a first approximation to size reduction and general decontamination costs.

The cost of disposing of boilers whole is lower than any of the packaged disposal options (Figure 7-8) 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 (European Commission – Co-ordinated Network on Decommissioning (EC-CND) 1995). The disposal volume and package costs are half that of the planning norm (i.e. 10tonne/HHISO) if the maximum loading of 20tonne/HHISO is achieved. The activity charge, taken as a percentage of the disposal volume cost, also halved. This later assumption may be too crude but is a small cost component in all cases. Transport costs are <1% in all cases.

Although the calculation methods are slightly different, the estimated disposal costs for the 10te/HHISO planning norm are in good agreement with industry estimates of

£5.89m for the disposal of 780tonne of Magnox metals in 2013/2014 (Shipton and Falconer 2013). The Magnox costs are dominated by disposal costs (75%), with container and activity costs of 11% and 13% respectively and a transport cost of 1.6%.

The WAGR boiler treatment costs were based on the estimates for the Berkeley boiler recycling contract costs of £5200 to £5800/tonne (NDA 2011) and (Studsvik 2012). This gave a treatment cost of between £4m and £4.4m for the WAGR boilers. This compares with a Magnox metals joint waste management plan treatment estimate of ~£3.56m (Shipton and Falconer 2013). The Magnox costs are dominated by the treatment costs (78%), with the container costs of 9% and transport and residual waste costs of ~6.5% each. The costs are similar, since the container, transport and disposal costs are from documents published on the LLW Repository Ltd website (www.llwrsite.com) and the treatment costs is a planning norm value of £4810/tonne rather than a published treatment contract cost. The cost of treatment for recycling or re-use is therefore approximately equivalent to that for disposal with a packing density of 15tonne/HHISO.

The results presented in Figure 7-4, Figure 7-5 and Figure 7-8 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 was 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. It also gave a lower estimated radiation dose to the process workers at WAGR and removed the need for further internal decontamination and size reduction of the boilers (EC-CND1995). For most of the UK metals inventory whole disposal of large contaminated plant at the LLW Repository (Drigg) without additional containment would not be possible. This is because it is against current policy and strategy as it would be detrimental to conserving scarce disposal capacity. However, large items can still be disposed to the LLW repository if demonstrated to be the BAT or BPEO.

7.4 Material and Process Sensitivity and Uncertainties

This section discusses the sensitivity and uncertainty in waste disposal, waste container, waste treatment and recycling factors. It also considers possible variations in avoided metals assumed in recycling, the recycling percentage and transport factors.

7.4.1 Disposal Factors

As mentioned previously in Chapter 6 there are no data for UK radioactive waste disposal in the Ecoinvent database embedded in SimaPro, hence Ecoinvent data for the Swiss LLW disposal impacts were used. A preliminary comparison of the direct disposal of the boilers including road transport, backfill and disposal for various Swiss radioactive waste disposal impacts showed that the transport and backfill impacts were trivial compared to the disposal impact. It also showed that:

a) Assuming Swiss HAW disposal gave a net impact of ~1.21E6Pt
b) Assuming Swiss LLW disposal gave a net impact of ~1.24E5Pt
c) Assuming Swiss Landfill disposal gave a net impact of ~2.20E4Pt

Where, a point (Pt) is the Eco-indicator-99 score discussed previously.

In other words there was about an order of magnitude between the LLW and HAW disposal impacts and about a further half an order of magnitude between LLW and landfill disposal impacts. Hence, in the absence of UK data the Swiss LLW disposal impact appears to be the most appropriate choice as the HAW disposal impact would be too high to be representative and the landfill disposal impact would not be an appropriate choice for radioactive waste disposal.

7.4.2 Waste Container Factors

A review of the HHISO impacts showed that production, estimated from the Ecoinvent average metal working process impact, was ~92% of the container metal impact. The HHISO steel and production impacts were ~19% and ~18% TDDI respectively for the packaged disposal scenario shown in Figure 7-4. Only two HHISO were needed for the disposal of the residual wastes from the bulk and

containerised recycling scenarios in Figure 7-4, hence the container impacts were reduced to less than 1% TDDI for each recycling option.

The transport impact for empty HHISOs to WAGR from Drigg was ~0.5% of the HHISO metal impact. If the HHISOs were assumed to be supplied direct from the manufacturer (e.g. Yorkshire Marine Containers on Humberside) the empty HHISO transport impact increased to ~3.3% of the HHISO metal impact.

In summary, container metal and production impacts are significant for the disposal scenarios but negligible for treatment and recycling scenarios and too small to be plotted in Figure 7-4. Transport impact for the supply of the containers was negligible compared to the container metal and production impacts.

7.4.3 Treatment and Recycling Factors

As discussed previously, the treatment processes included boiler size reduction and decontamination, metal melting, slag production and slag crushing. Engineering calculation, based on WAGR boiler data from Dixon 1999 and Crossley and Wakefield 1991 were used to estimate the process material flows. The impacts of the secondary wastes associated with each processes, and the dust and gases captured in the filtration system, should also be included. However, they were assumed to be negligible compared to the processes modelled and are not included here. Figure 7-4 shows that the total impact from all the treatment process for the bulk and containerised recycling scenarios was about 20 to 23% TDDI, hence the contribution from each process was small.

Size reduction and Decontamination Impacts

Size reduction and decontamination were based on engineering calculation of cutting up the boilers and skimming off the surface contamination using the milling and dressing data from the Ecoinvent database. A comparison with the size reduction estimates for the ECOMET-S facility in the Russian Federation (Gelbutovsky et al. 2006 and Gelbutovski et al. 2009) showed that the original engineering estimates could be a factor of 3 to 3.5 too small. Assuming a factor of 4 increase in the original engineering estimates increase the size reduction impact by ~17% and

decontamination impacts by ~3%. Together these changes would increase the total treatment process impact to about 24% to 28% TDDI.

Melting Impacts

The melting impact was based on the calculated electrical energy needed for the induction furnace only. It did not include energy required to operate the treatment facility ventilation plant or the water consumption need to cool the induction coils. If it was pessimistically assumed that the melting energy requirement was doubled the total treatment process impacts for bulk and containerised recycling in Figure 7-4 would increase by about 2%, i.e. to ~22% to ~25% TDDI.

The Studsvik Nykoping induction furnace electricity supply was taken as the Swedish medium voltage supply plus imported electricity. The high and low voltage supply including imported electricity would give a 0.4% and 3.5% treatment process rise respectively.

These changes for Sweden would not particularly large and would be about the same for the French treatment facility as discussed for Figure 7-6. The voltage changes would be more significant for the current German, Russian Federation and American treatment facilities, and a potential UK melting facility as discussed for Figure 7-6 and Figure 7-7.

Slag and Slag Crushing Impacts

The sintering data in the Ecoinvent database was used to represent the top slag produced during melting and the slag crushing to reduce the residual waste volume. Assuming the original engineering estimates for the slag and crushing had to be increased by a factor 4 the slag impacts would increase by about 18.5% and crushing by ~5.3%. These factors increase the total treatment process impacts to ~25% to ~28% TDDI.

7.4.4 Avoided Metal Factors

The changes in the treatment factors above are small because the process impacts are small compared to the total direct disposal impact. However, as seen in Figure 7-4

recycling the melt ingots gives a substantial benefit. This is because all of the melt ingots from the treatment process negate the need for the extraction, transport and production of steel from virgin ores, or save the recycling of conventional scrap metal, as discussed in Chapter 5. This is modelled in SimaPro by allocating an avoided material, which in this case could be pig iron, unalloyed, low or high alloyed steel or cast iron.

An example of different avoided metals is shown in Figure 7-9. The treatment processes impacts are constant at just over 20%TDDI and the avoided metal benefit is much larger at 60% to 90% TDDI. There is very little difference between avoided pig iron and unalloyed steel. The avoided metal will have to be re-melted, probably in an electric arc furnace with other scrap metals, alloying elements and fluxes to produce new steel. Hence, the reduced benefit caused by re-melting is shown in the "Unalloyed Steel with Electric Steel Off-set" example in Figure 7-9.

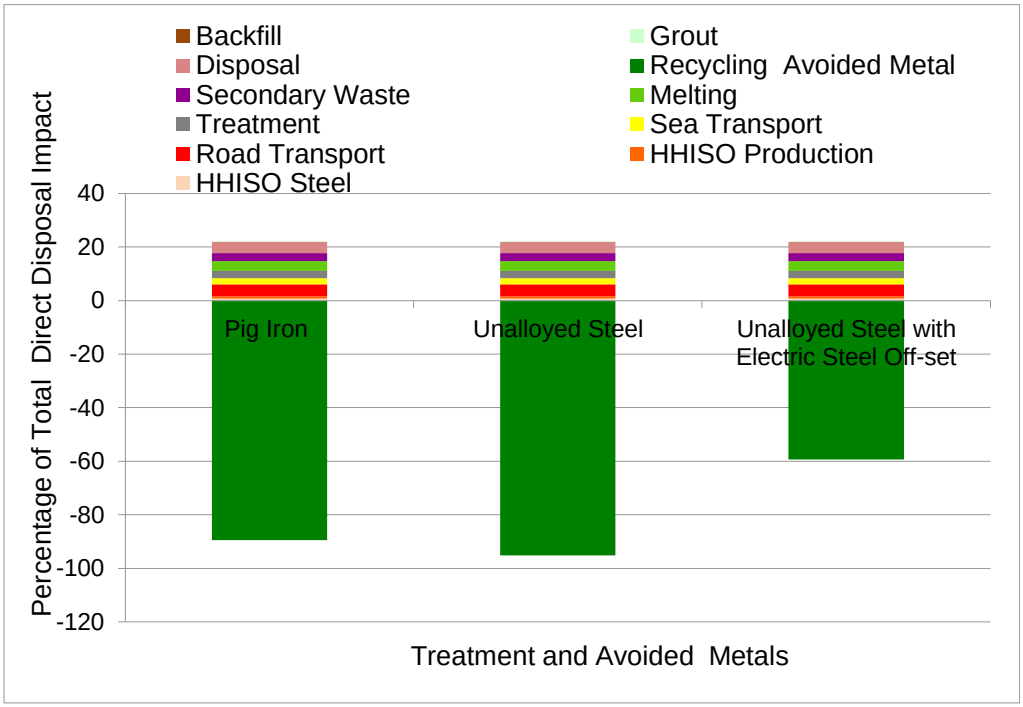

Figure 7-9 Comparison of containerised recycling with different avoided metals (Eco-Indicator 99 Hierarchist/Average). A positive value represents an adverse environmental impact a negative value represents an environmental benefit. The results are percentages of total direct disposal impacts (TDDI). The pig iron and unalloyed steel were assumed to be produced from predominately primary iron in a blast furnace as discussed in Chapter 5. However, secondary steel is also produced from nearly 100% scrap in an electric arc furnace. It was assumed that to re-use the treated ingots it is necessary to re-melt them in an electric arc furnace and hence the detriment of re-melting the ingots reduces the benefit of the avoided metal. This is shown as the unalloyed steel with an off-set for electric steel melting.

The SimaPro results show that the impact of the European average electric arc furnace re-melting of the ingots was just under an order of magnitude greater that the equivalent Swedish induction melting. This is due the large non-fossil fuel generation in Sweden compared to European countries as discussed previously. There may also be a difference between a small clean melting process and the impacts of a heavy industrial smelting process.

To conclude, the sensitivity of the impact score to avoided metals and re-melting is much larger than any variation caused by individual treatment processes.

7.4.5 Changes in Recycling Percentage

The assumed boiler steel recycling rate (i.e. the percentage of recyclable material) is 95% as previously stated. As the recycling rate decreases it is expected that the avoided metals impact will decrease and secondary waste (i.e. metal that could not be released and processin waste), HHISOs and disposal volume impacts will increase proportionally. This will lead to a higher number of HHISO and hence an associated increase in steel, production, transport, grouting and disposal impacts. This is shown in Figure 7-10.

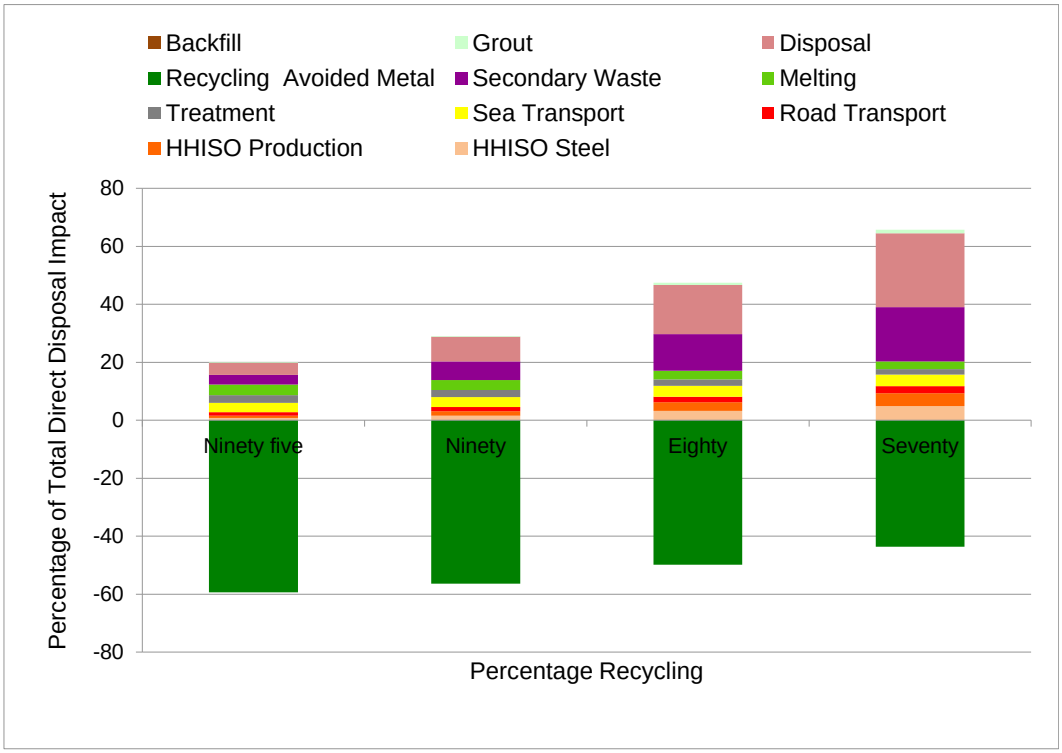

Figure 7-10 Comparison of Eco-Indicator 99 (Hierarchist/Average) potential environmental impacts for the treatment processes and recycling for the containerised recycling scenario at Nykoping in Sweden The values shown are the percentage recycling rate changes as a percentage of total direct disposal impact.

The figure shows that as the recycling rate drops the benefit of the avoided metal decreases linearly the treatment and secondary waste impacts increase linearly. Interpolating from the data in Figure 7-10 is appears that at about 75% recycling the benefits and detriments of the processes will be equally. A current recycling project for fifteen 310tonne Magnox boilers from Berkeley power station achieved a 97% recycling rate (NDA 2014a), this supports the 95% recycling rate assumed for the research discussed in Section 7.3.2.

7.4.6 Transport Factors

The transport impact for direct disposal and packaged disposal in the UK were between ~0.1% and ~0.4% TDDI from Figure 7-4. The transport for bulk recycling and containerised recycling in Sweden was less than 0.1% TDDI. However, this is because the impacts are off-set by the benefits of recycling the boiler steel. If these are ignored the bulk and containerised transport impact becomes about 7% TDDI, and as high as 16% TDDI for international treatment in the USA as discussed previously for Figure 7-6.

7.5 Modelling Uncertainty

This sub-section presents a comparison of the overall life cycle impacts for the four disposition options for the hierarchist, egalitarian and individualist perspectives outlined in Table 6-1 for both the average and cultural group normalisation and weighting in Eco-Indicator 99. Hence, there are six potential variants of the results for each of the four disposition scenarios.

Also given is a comparison of the net life cycle impact for the Hierarchist/Average perspective for Eco-Indicator 99 and another life cycle impacts analysis method ReCiPe 2008. ReCiPe 2008 is an updated version of Eco-Indicator 99 combining some of the thinking and methods from a contemporary life cycle impact analysis method CML 2001.

Eco-Indicator 99 Total Potential Environmental Impact Results for all cultural perspectives

A comparison of the Eco-indicator 99 net life cycle impacts results for each cultural perspective with average and prespective normalisation and weighting, and using European data, are shown in Figure 7-11. The figure shows the results for the six potential cultural perspective options discussed in Section 6.1.1.

Figure 7-11 shows that perspectives can be split into two groups with very similar outcomes. First, the Egalitarian and Hierarchist results, regardless of the weighting adopted, are all very similar. By comparison, again regardless of the weighting method adopted, the Individualist assumptions results in significantly over estimate both of the environmental detriment (positive values) associated with disposal and of the environmental benefits (negative results) of treatment and recycling.

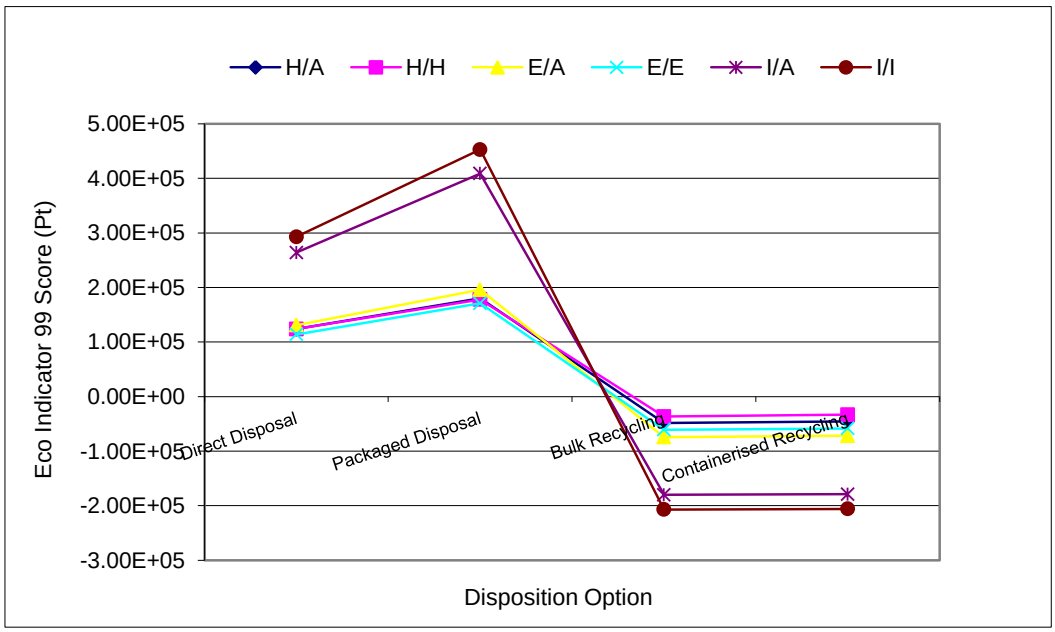

Figure 7-11 Variation of Eco-Indicator 99 total life cycle potential environmental impact results for the four disposition options for each cultural perspective. **H/A** is Hierarchist/Average normalisation and weighting, **H/H** - Hierarchist/Hierarchist normalisation and weighting, **E/A** - Egalitarian/Average normalisation and weighting, **E/E** - Egalitarian/Egalitarian normalisation and weighting, **I/A** - Individualist/Average weighting, **I/I** - Individualist/Individualist normalisation and weighting. A negative value is an environmental benefit. The results are in Eco-Indicator 99 scores (Pt), a negative value represents an environmental benefit.

The differences arise from two factors, first the modelling assumptions for each cultural perspective, summarised in Table 6-1. Secondly, the normalisation factors giving the relative magnitude of the damages to Human Health, Ecosystem Quality and Resources are very similar for the Egalitarian and Hierarchist perspectives, giving very similar results, where as those for the individualist perspective are substantially lower giving higher results. Hence, choosing the Hierarchist/Average perspective is a conservative assumption in that it minimises the difference between scenarios.

Eco-Indicator 99 and ReCiPe 2008 Hierarchist/Average results

The results for the net life cycle impacts for the four disposition scenarios from Eco-indicator 99 and ReCiPe 2008 life cycle impact assessment methods are presented in Figure 7-12. The results are for the European Hierarchist/Averaged normalisation and weighting options for both life cycle impact analysis methods. Figure 7-12 shows that there is not a large difference in the results from the two methods and the basic trend of results remains unaltered.

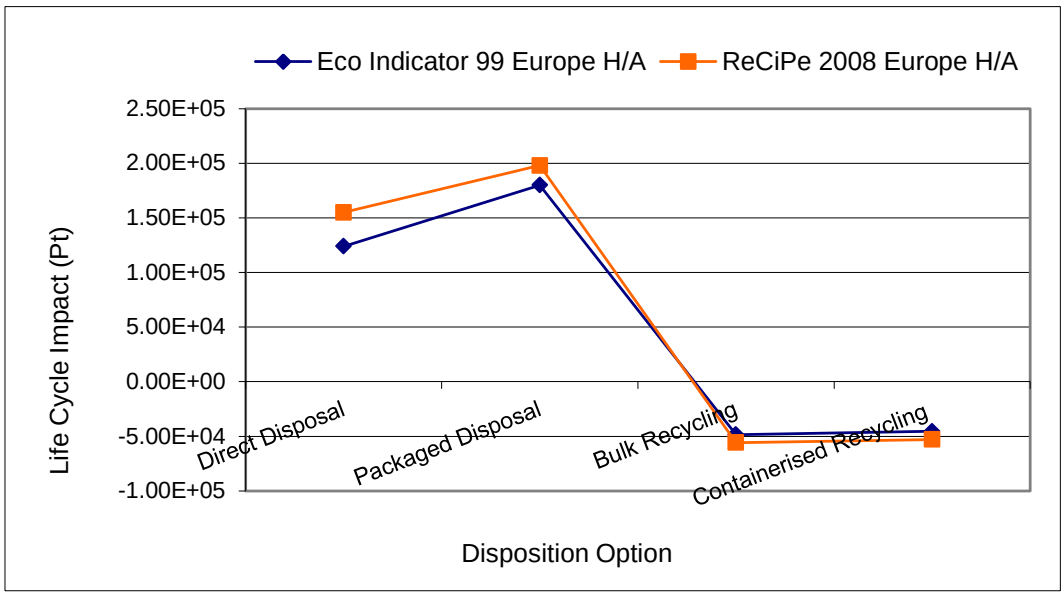

Figure 7-12 Comparison of the four disposition options total potential environmental impacts using Eco-Indicator 99 and ReCiPe 2008 European Hierarchist/Average (H/A) cultural perspective. The results are in Eco-Indictor 99 and ReCiPe 2008 scores (Pt). A negative value represents and environmental benefit.

The comparison in the last two figures are intended to help put the Eco-Indicator 99 Hierarchist/Average results reported previously in the chapter into perspective and give an indication of the inherent uncertainties involved in using these cultural perspective based environmental damage assessment methods. The results in both figures give confidence in the overall results generated in this case study and hence the Eco-indicator 99 (Hierarchist/Average) life cycle impact assessment method will be used in the remain two sections of the research.

Weighting Eco-indictor 99 results

It is possible to change the weighting factors from Human Health, Eco-system Quality and Resource damages in SimaPro. However, there was no clear logic as to how a different set to weighting factors could be derived and justified. A test of modifying the weighting for an early modelling example of the four disposition scenarios was produced using an Excel spreadsheet of the total life cycle potential environmental impacts.

The eleven environmental impact category indicators (as shown in Figure 7-5) are grouped in three damage categories; Human Health, Eco-system Quality and Resources consistent with the current ISO14040 2006 standard requirements

discussed in Chapter 3. The relationship of the impact categories, damage categories and Eco-Indicator 99 embedded Hierarchist/Average weightings are shown in Table 7-1.

Table 7-1 Eco-Indicator 99 Impact Category and Damage Category Weighting Factors from SimaPro 7.3.3

Impact Category	Damage Category	Damage Category Units	Hierarchist Average Weighting Factors
Carcinogens	Human Health	DALY	400
Respiratory Organics	Human Health	DALY	400
Respiratory Inorganics	Human Health	DALY	400
Climate Change	Human Health	DALY	400
Radiation	Human Health	DALY	400
Ozone Layer	Human Health	DALY	400
Eco-toxicity	Eco-system Quality	$PDF.m^2.yr$	400
Acidification & Eutrophication	Eco-system Quality	$PDF.m^2.yr$	400
Land Use	Eco-system Quality	$PDF.m^2.yr$	400
Minerals	Resources	Surplus MJ	200
Fossil Fuels	Resources	Surplus MJ	200

The weighting factors were taken from the Eco-indicator 99 method's data in SimaPro 7.3.3. DALY represents the Disability Adjusted Life Years impact on health, PDF represents the Potentially Disappeared Fraction of plant species, MJ represents the Mega Joule surplus energy required to excavate dwindling mineral and fossil fuel resources as discussed in Section 6.1.1. Weighting is discussed in Section 3.2.3 and Section 6.1.1.

To investigate the effect on total life cycle impact of changing the Eco-Indicator 99 (Hierarchist/Average) weighting factors in Table 7-1 a uniform weighting of 333 was applied to a set to preliminary normalised impact results. Since Eco-indicator 99 is a linear assessment method this increased the resources (minerals and fossil fuels) weighting by ~67% but decreased the human health and ecosystem quality weighting by ~17%. The results of the change are shown in Figure 7-13.

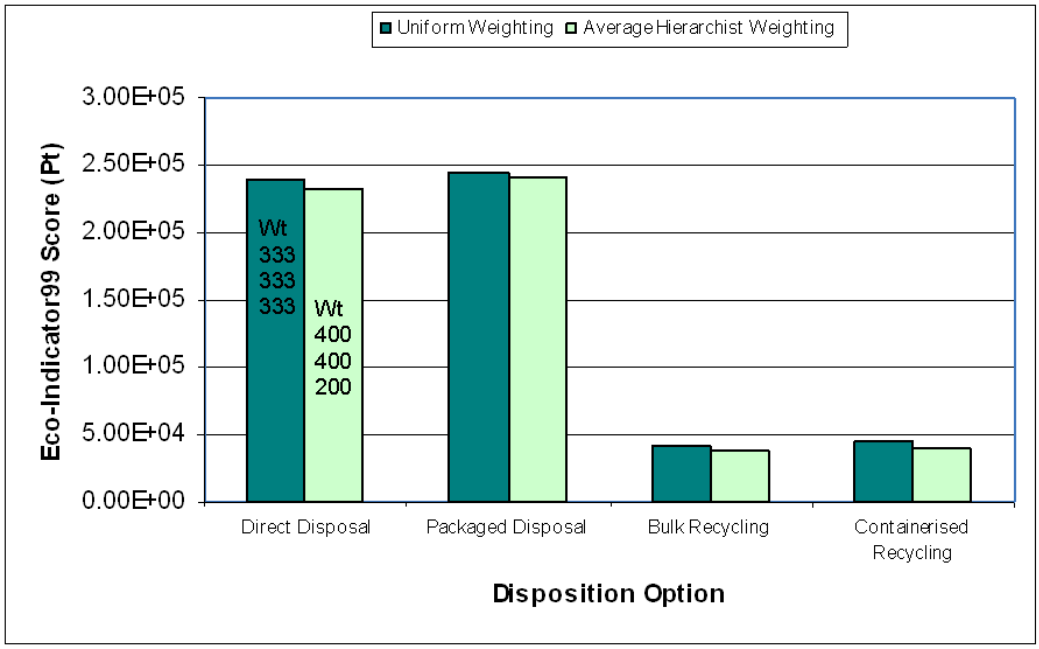

Figure 7-13 Comparison of total life cycle potential environmental impacts for the four scenarios using the Eco-indicator 99 (Hierarchist/Average) weighting (wt) factors (400 for human health and eco-system quality and 200 for resources damage categories) and a uniform 333 weighting (wt) factor.

The results showed that changing to a uniform weighting factor rather than the embedded cultural perspective weighting factors only marginally increased the scenario total life cycle impact results. Given that the change in weighting produced a negligible overall effect in these four scenarios it was decided to retain the Eco-Indicator 99 Hierarchist/Average method for the remaining research.

7.6 Case Study Summary

Overall, the environmental impacts and costs are dominated by LLW disposal as shown in Figure 7-9. LLW metal environmental impacts and costs can be reduced by increasing waste container loading for disposal and by treatment to minimise disposal as shown in Figure7-4. Recycling LLW metals, thus avoiding future virgin materials or conventional scrap for new products, offer further substantial improvements as shown in Figure 7-4. These points are summarised pictorially in Table 7-2 to show the relationship between potential environmental impacts and estimated costs.

Table 7-2 Summary of potential environmental impacts and financial costs for WAGR boiler disposition options

	Direct Disposal	Packaged Disposal	Bulk Recycling	Containerised Recycling
Potential Environmental Impact	◉	●	⊙	⊙
Direct Financial Cost	⊙	●	○	○

Key: Lowest ⊙, Low ◉, Medium ○, High ◉, Highest ●

Direct disposal gave a high environmental impact but the lowest estimated cost (~£2.5m from Figure 7-8). Packaged disposal gave the highest environmental impact at ~144% TDDI (Figure 7-4) and the highest costs (~£5.9m from Figure 7-8). Both recycling scenarios gave the lowest environmental impact (~22% TDDI without recycling, -38% TDDI with recycling from Figure 7-4) and medium costs (~£4m from Figure 7-8). Improving the waste disposal loading from 10tonne/HHISO to 15tonne/HHISO and 20tonne/HHISO resulted medium and low cost of ~£4m and ~£3m respectively from Figure 7-8.

Having demonstrated the potential of the approach for the WAGR boiler case study attention will now turn to considering the UK radioactive metals inventory as a whole.

8 UK Radioactive Waste Metals Inventory Analysis

The previous chapter showed that LCA models of low level waste (LLW) metals disposal and treatment for recycling or re-use could be constructed. It also showed that radioactive metals recycling and re-use could significantly reduce the waste volume for disposal at the LLW Repository near Drigg in Cumbria. The results are consistent with the current UK solid LLW policy (DEFRA et al. 2007), strategy (NDA 2010a) and plans (e.g. Shipton and Falconer 2013) which helps conserve the scarce disposal capacity at the LLW Repository. This chapter uses the experience gained from Chapter 7 to model the disposal, treatment, recycling or re-use of all metals in the 2010 UK radioactive waste inventory (NDA and DECC 2011).

8.1 UK Radioactive Waste Inventory

This section discussed the UK radioactive waste inventory background and the radioactive waste metals data used in the SimaPro models.

8.1.1 Inventory Background

The 2010 inventory was primarily used for this research (NDA and DECC 2011). Estimates for some metals were extrapolated from previous inventories as they are no longer reported as individual values in the 2010 inventory.

8.1.2 SimaPro Metal Inventory Data

The carbon steel, stainless steel and aluminium masses were taken from the 2010 UK radioactive waste inventory. The metals reported in the UK radioactive waste inventory changed after 2001. Copper, lead, nickel and zinc are no longer individually reported and estimates were extrapolated from the 2001 inventory (DEFRA and Nirex 2002) to provide data for the SimaPro models. The change in UK solid LLW policy and strategy resulted in high volume very low level waste (VLLW) being reported as a subset of LLW from 2010 as discussed in Section 2.1.1. VLLW metals were not reported separately in the 2010 inventory hence they also

had to be estimated here. The VLLW metals masses were based on Springfields historical data as they were the only site that regularly reported VLLW. LLW and VLLW are commonly grouped together as low activity waste (LAW) as discussed in Section 2.1.1 and Box1-1.

The radioactive metal masses used in SimaPro are presented in Table 8-1 and show which UK radioactive waste inventory was used for each metal.
The data are taken from the 2001 and 2010 inventories but the total ILW and LAW masses are consistent with the 2010 metals inventory. The waste packaging sub-assemblies and treatment, disposal and recycling end-of-life processes were linked to these VLLW, LLW and ILW metal masses.

A new inventory was produced for 2013 (NDA and DECC 2014a et seq). This showed that the ILW metals estimate remained unchanged but the LLW metals estimates decreased by about 6.4%. Hence the current results may be slightly pessimistic but still a representative first order model for disposal and recycling.

Table 8-1 Radioactive metals inventory data used in SimaPro for disposal, treatment and recycling

Metal	**Waste Type**	**Mass (tonne)**	**Source**
Aluminium	ILW	1,200	UKRWI 2010
Aluminium	LLW	2,353	UKRWI 2010
Aluminium	VLLW	14,647	Springfields
Copper	ILW	405	UKRWI 2001
Copper	LLW	3,875	UKRWI 2001
Copper	VLLW	1,869	Springfields
Unspecified	ILW	7,423	UKRWI 2010
Unspecified	LLW	171,561	UKRWI 2010
Unspecified	VLLW	0	
Lead ILW	ILW	822	UKRWI 2001
Lead LLW	LLW	6,486	UKRWI 2001
Lead	VLLW	3,694	Springfields
Nickel	ILW	3,102	UKRWI 2001
Nickel	LLW	7,650	UKRWI 2001
Nickel	VLLW	0	
Stainless Steel	ILW	40,000	UKRWI 2010
Stainless Steel Surface Contaminated	LLW	59,402	UKRWI 2010
Stainless Steel ‘Activated’	LLW	24,000	UKRWI 2010

Metal	Waste Type	Mass (tonne)	Source
Stainless Steel	VLLW	36,598	Springfields
Mild Steel	ILW	49000	UKRWI 2010
Mild Steel Surface Contaminated	LLW	270233	UKRWI 2010
Mild Steel 'Activated'	LLW	86,000	UKRWI 2010
Mild Steel	VLLW	73,767	Springfields
Zinc	ILW	48	UKRWI 2001
Zinc	LLW	353	UKRWI 2001
Zinc	VLLW	14,647	Springfields
Sub-total ILW		102,000	
Sub-total LLW		631,913	
Sub-total VLLW		145,222	
Sub-total LAW		777,135	

Key
"UKWRI" - UK radioactive waste inventory. "Unspecified "- metals that have not yet been individually reported in the inventory and hence could be a mixture of all the above metals and other metals such as tin, bronze, platinum. "Activated" - steel with a tenacious oxide film containing activation products, or evidence of limited neutron activation of steels outside the reactor biological shield. Copper, lead, nickel and zinc data are extrapolated from the 2001 UKRWI inventory data used on the assumption that none had been disposed. Springfields VLLW wastes metals from the 2010 UKRWI were taken as a starting point for VLLW metals in SimaPro (see Rad Metals for SimaPro in Table D-1).

8.2 End-of-life Disposition Scenarios

This section gives an outline of the four disposition scenarios: baseline disposal, improved packaging and international treatment without recycling and with recycling. It also discusses some scenario assumptions.

8.2.1 Scenarios

Four LCA scenarios were considered for higher volume VLLW metals, mixed VLLW/LLW metals, LLW metals and ILW metals as discussed in Section 6.1.1. Hence, the functional unit for these LCAs was – the removal of the current estimated radioactive waste metals from UK legacy nuclear facilities. Each scenario is now discussed in turn.

Baseline Disposal

The first scenario was a baseline case that assumed no segregation of VLLW and LLW metals. Hence both wastes were disposed in cement grouted half height ISO (HHISO) freight containers to the LLW Repository near Drigg with a nominal waste

loading of 10tonne/HHISO (Shipton and Falconer Ltd 2013 and LLWR 2011b). This reflected the UK position prior to the current solid LLW policy and strategy. All ILW metals were assumed to be disposed to a future geological disposal facility (GDF) based nominally at the waste mass weighted average distance from each nuclear site to Sellafield as discussed in Section 6.5. The waste was disposed in cement grouted stainless steel 4m boxes with internal concrete shielding to reduce the external radiation levels for handling and transport. The waste loading was assumed to be 10tonne/4m box.

Improved Disposal

The second scenario postulated an improved waste packaging approach. It assumed that 20% of the LLW metals could be characterised and segregated as VLLW metal and disposed to a licensed VLLW landfill site in ungrouted 210litre drums rather than grouted HHISOs. The remaining LLW metals were assumed to be disposed to the LLW Repository (Drigg) with an improved average waste loading of 15tonne/HHISO. ILW that could decay or be decontaminated to LLW is ~3% of the ILW inventory (NDA and DECC 2010). This value increases to ~9% if it is assumed to apply only to ILW metals. Hence ~9% of the ILW metals were assumed to decay or be decontaminated to LLW and disposed to the LLW Repository (Drigg) rather than a future GDF. The radioactivity levels and handling requirements of the original ILW metals makes a 15tonne/4m box unlikely, hence a target 12tonne/4m box was assumed for disposal of the decayed/decontaminated metals to Drigg in 4m boxes. It was further assumed that since they were destined for LLW disposal the 4m boxes could be made of carbon steel like the HHISOs therefore saving on stainless steel costs. The remaining ILW metals were assumed to be disposed at 12tonne/4m box to a GDF.

The improved packaging is suggested on the basis that consigners will want to reduce their disposal costs and therefore make every effort to optimise the waste loading to meet the handling, transport and disposal requirements. Hence they are intended as indicative of the waste loadings that may be achieved for the majority of the ILW metals.

International Treatment without Recycling

The third scenario assumed that the benefits of radioactive metals treatment for recycling and re-use could be applied to the majority of the LLW metals and the ILW metals that could decay to be decontaminated to LLW. This is consistent with the current UK solid LLW policy, strategy and plans.

It was conceivable that some LAW metals might require too much effort or expense to treat and recycle and hence the business case is disposal. Alternatively, the excepted radiation dose to the process staff at UK nuclear sites or international treatment facilities may not be as low as reasonably practicable and hence a radiological case cannot be made for treatment. In the absence of data on how much of the LAW would fall into this category 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 assumed to be treated for recycling but generated 5% secondary waste for disposal. It was further assumed that all VLLW metals and half of the surface contaminated LLW metals were decontaminated to exempt level in the UK without melting. The remaining half of the surface contaminated LLW metals were transported to Sweden for treatment and melting. All activated LLW metals and ILW metals decayed/decontaminated to LLW were assumed to be sent to Sweden for melting for waste volume reduction. All secondary wastes from international treatment and melting were assumed to be returned to the UK for disposal unless otherwise stated.

It was further assumed that ~14% of the LLW metals were lightly neutron activated or had an oxide film containing activation products (Nieves et al. 1995) as previously discussed. It was therefore assumed that these activated LLW metals and ILW metals that could decay, or be decontaminated, to LLW would be melted for volume reduction only. A waste volume reduction factor of 20 (IRSN 2004, Buckley et al 2004, SOCODEI 2008) was assumed for these metals to match the 5% disposal. This is rather optimistic and warrants further research as radiation doses to power station staff and treatment facility staff plus the final ingot assay levels will be important.

The melt ingots produced from these metals will exceed exempt levels hence they will have to be returned to the consigner for disposal. Care must be taken not to increase the radiation level (Bq/m^3) of melt ingots to above LLW levels or they will have to be transported and disposed as ILW, thus negating the benefit of volume reduction.

This scenario assumes that the system boundary for the LCA stops at the treatment facility. Hence no benefit is claimed for recycling the melt ingots in open metals markets or avoiding virgin materials by making shielding block, cylinders or ILW waste packages etc. However, it does include the size reduction, decontamination, transport to UK and/or international treatment facilities, metal melting and the repatriation and disposal of residual wastes to the UK.

International Treatment with Recycling

This scenario has the same basic assumptions, materials and processes as the previous scenario but assumed that the benefit of recycling or re-using the melt ingots can be included in the LCA. Unless a product destined for re-use can be cast directly from the melting process as for example shielding blocks in America or cylinders in France, then some reheat or re-melting would be required.

Early development stage data and results (indexed in Table D-1) and main case study data and results (indexed in Table D-2), supporting the scenarios are summarised in the MicroSoft Excel spreadsheets on the compact included in Appendix D.

8.2.2 Scenario Assumptions

Additional Modelling and Data Assumptions

The general modelling assumptions were common with the WAGR case study in Chapter 7 and discussed previously in Chapter 6. However, separate sub-assemblies were generated for the waste metals and the stainless steel 4m boxes for ILW metals, carbon steel HHISOs for LLW metals and carbon steel 210 litre drums for VLLW metals. These sub-assemblies included the Ecoinvent average metal working impact as a proxy for container production and empty package transport to site.

Waste treatment processes were generated for each of ILW, LLW and VLLW metals. Size reduction, decontamination, melting, cement grouting of waste packages plus full package transport and disposal at the appropriate disposal facility were included consistent with the initial WAGR case study in Chapter 7. Where possible, Ecoinvent data were used for each metal, but where data were not available unalloyed steel data was assumed. Also, since the constituent metals for the "Unspecified metals" in Table 8-1were unknown they were assumed to be steels as these are about 71% of LLW metals and 87% of ILW metals. Although the 4m boxes for ILW disposal were presumed to have 100mm or 200m thick internal concrete shielding in the model this was included as part of the packaging grouting cement impact.

The postulated melting of the WAGR boiler steel in Chapter 7 assumed the Swedish medium voltage with imported supply. As stated previously the induction melting impact was about $1/10^{th}$ of the impact of melting the equivalent scrap steel mass in an electric arc furnace. Hence a 10% electric arc furnace impact was assumed for melting the metals in this scenario. Studsvik's Swedish treatment facility at Nykoping was taken as a reference European facility as discussed in Chapter 6.

In Chapter 7, WAGR size reduction data were estimated from cutting up the boilers to meet the induction furnace loading requirements. This approach was not practical for the UK metals inventory investigation hence size reduction was estimated from data in the Wallbridge et al 2012a UK decommissioning LCA.

8.3 Metals Inventory Analysis Results

This section presents the potential environmental impact results for the baseline and improved disposal scenarios and the international treatment scenarios without and with the benefit claimed for recycling. It also presents the estimated financial costs and savings for each scenario.

8.3.1 Potential Environmental Impact Results for Each Scenario

The results of the baseline disposal model (based on 10tonne waste/container) are shown in Figure 8-1. The results are split in to the impacts associated with containing

the waste metals and the impacts for disposing of the waste to VLLW, LLW and ILW repositories.

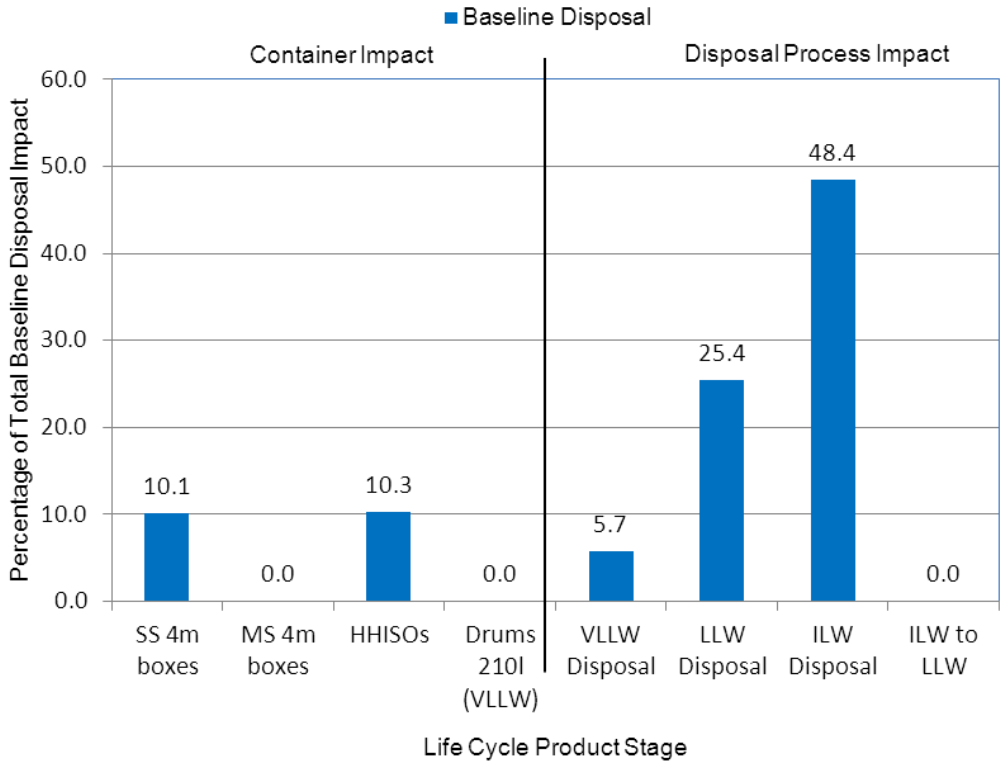

Figure 8-1 Baseline disposal impacts for the 2010 UK radioactive metals inventory (from Table 8-1) for waste container impacts and waste disposal impacts. The results are in percentage of Total Baseline Disposal Impact (TBDI) from Eco-Indicator 99 (Hierarchist/Average). SS - stainless steel. MS - mild steel.

The disposal process accounts for about 80% of the Total Baseline Disposal Impact (TBDI) and is dominated by the ILW disposal, followed by LLW disposal. VLLW disposal results in a significantly lower impact because of the low repository impacts. The container metal, container production and transport account for about 20% TBDI, arising equally from packaging the ILW metals in 4m stainless steel boxes and the LLW plus VLLW metals in HHISOs. Far fewer stainless steel 4m boxes are needed for ILW metals disposal compared with HHISOs used for LLW/VLLW metals disposal. The material and production impacts for the stainless steel 4m boxes are significantly higher than those for mild steel HHISOs, however the container impacts are essentially equal.

There are no mild steel 4m boxes for ILW or 210 litre drums for VLLW disposal in this model. No ILW metals are assumed to decay to LLW. The impacts for these latter aspects are therefore zero in the figure.

Figure 8-2 shows a comparison of the results for the two disposal model scenarios: the baseline scenario and the improved packaging scenario. The improved packaging scenario is still dominated by ILW and LLW disposal, but both the disposal and container impacts are significantly reduced.

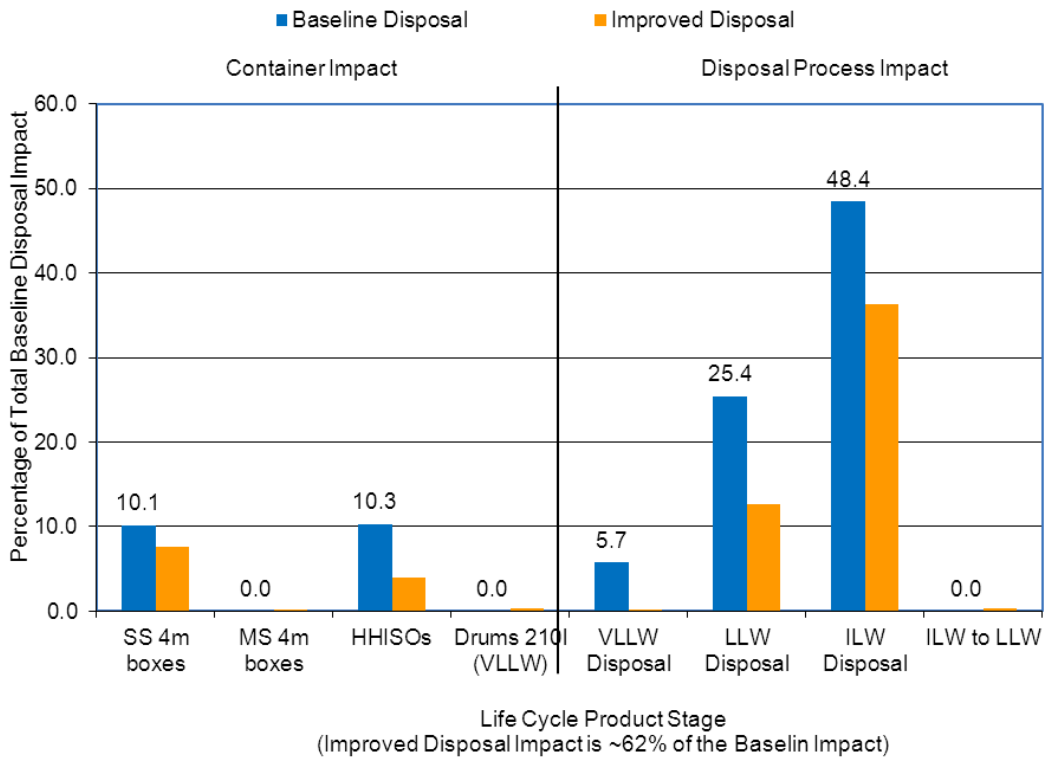

Figure 8-2 Comparison of baseline disposal and improved packaging disposal scenarios for the 2010 UK radioactive metals inventory (from Table 8-1) for waste container impacts and disposal impacts. The results are in percentage of Total Baseline Disposal Impact (TBDI) from Eco-Indicator 99 (Hierarchist/Average). SS – stainless steel. MS – mild steel.

The improved packaging disposal scenario gives about a 38% reduction in environmental impact compared to the total TBDI: 13% from increasing the container load from 10tonne/HHISO to 15tonne/HHISO for LLW and 12% from 12tonne/4m box for ILW. A further 5% is from VLLW disposal to specified landfill, with 6% from changing the VLLW containers from grouted HHISOs to ungrouted 200 litre drums. There is also about a 2.5% improvement for packaging the ILW metals that decay, or can be decontaminated, to LLW in mild steel 4m boxes. This is off-set slightly by ~1% increase in LLW disposal impact.

Figure 8-3 compares the environmental impacts of disposal with those for treatment with and without 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 TBDI. Including recycling reduces the impact further to ~38% of total TBDI. Hence, international treatment without and with recycling represent a 15 % to 24% reduction in environmental impact compared to the improved disposal.

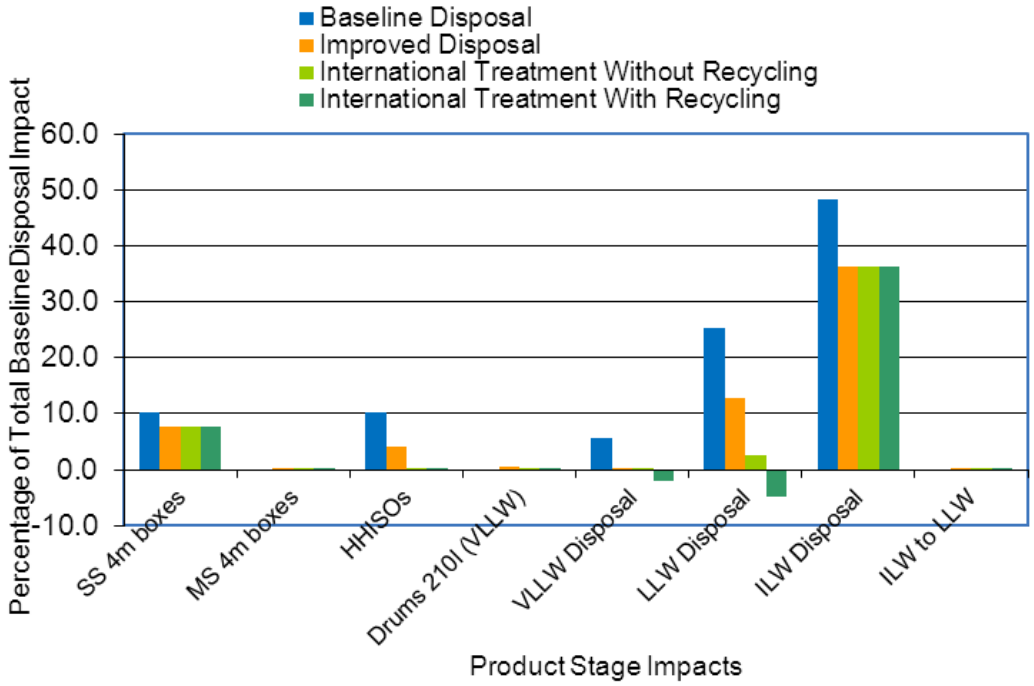

Figure 8-3 Comparison of disposal scenarios and international treatment scenarios with and without recycling of the 2010 UK radioactive metals inventory (from Table 8-1). The results are in percentage of Total Baseline Disposal Impact (TBDI) from Eco-Indicator 99 (Hierarchist/Average). A negative value indicates an environmental benefit. SS – stainless steel. MS – mild steel.

Analysis of Figure 8-3 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 200 litre drums respectively. Recycling surface contaminated LLW metals, thus avoiding future metals made from virgin ores, gave an additional benefit of ~5% of total TBDI, with a further 2% TBDI for recycling VLLW metals.

These results demonstrate the environmental benefits achieved by the introduction of the 2007 UK LLW policy that requires the treatment and recycling of LLW and VLLW metals. The results also imply that significant benefits could be obtained if the UK applied similar logic to ILW metals. Some ILW metals have already been decontaminated to LLW (e.g. Hinkley Point A used fuel skips) and then treated at international facilities for recycling or re-use in the nuclear industry. However, examples of ILW metals decayed to LLW and treated for recycling or re-use are unknown. This type of metal is likely to be highly activated (i.e. volumetrically contaminated) and may require a modified treatment process with additional shielding to protect the power station and treatment facility staff from the higher radiation levels. Transport of this type of metal may also be an issue and may require extra shielding of the container to comply with international transport regulatory limits. This is an area for further research and ILW metal treatment is anticipated to be significantly more expensive than current LLW treatment.

Figure 8-4 shows the impacts split by category for each of the four scenarios. Respiratory inorganic and fossil fuel impacts still dominate as seen in the WAGR boiler case study in Chapter 7.

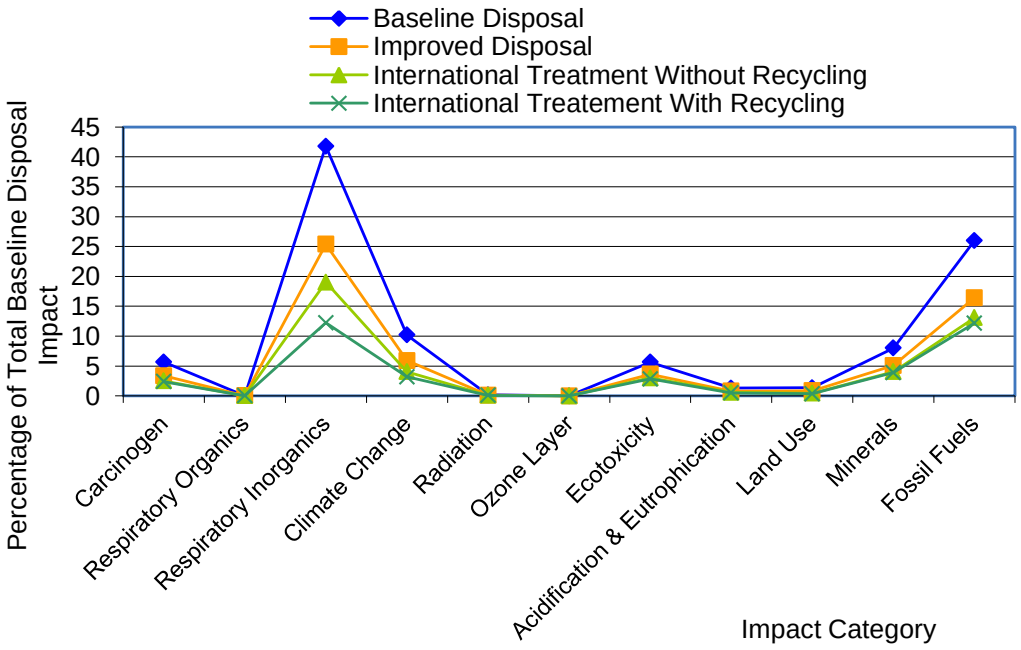

Figure 8-4 Environmental impact category results for baseline and improved disposal scenarios and international treatment scenarios without and with recycling for the 2010 UK radioactive metals inventory (from Table 8-1). The results are in percentage of the Total Baseline Disposal Impact (TBDI) from Eco-Indicator 99 (Hierarchist/Average).

The reduction in impacts from the baseline to the improved disposal is a result of the reduced waste disposal containers, and hence waste disposal volume, from increased waste package loading, segregation and disposal of VLLW metals and the LLW disposal of decayed or decontaminated ILW metals. The largest reduction from the improved disposal to international treatment is for respiratory inorganic compounds. These show ~6.5% reduction without claiming the benefit of recycling the melt ingots and ~13% when this is also taken into account. There is ~3% reduction in fossil fuel impacts without recycling the treatment ingots and 4% when they are included. The improvements in the other impact categories are minor at between 1% and 3%.

The figure shows a progressive reduction in impacts from baseline disposal to improved disposal through to treatment with recycling. The addition of recycling, rather than just treatment, produces a further improvement to respiratory inorganic, climate change, mineral and fossil fuel plus carcinogens and ecotoxicity impacts due to the avoidance of the extraction and processing of metal ores including waste, energy and transport impacts.

8.3.2 Financial Cost Estimates

As stated previously obtaining LLW disposal container, transport and disposal costs from the LLW Repository Ltd website documents is straightforward (e.g. LLWR Ltd 2014). However, obtaining equivalent data for ILW packages and disposal to a future GDF proved more difficult. Container, treatment, disposal and transport costs are summerised in Section A.2 and A.3.

Container Costs

The cost of the 210 litre carbon steel drums for VLLW disposal was not stated in the LLW Repository Ltd website documents in 2011. However, these carbon steel drums are readily available and cost between £25 and £38 according to internet sources. The planning norm price for these drums was confirmed as £35 in LLWR Ltd 2014.

The disposal costs for reactor core graphite presented in NDA 2012a show a current production cost of £60,000 for a 4m stainless steel box, but it is recognised that the cost may drop as the production process is refined and demand for 4m boxes increases. Extrapolating the carbon steel HHISO cost of £8,000 by 4m box mass and assuming a price increase factor of 2 to 4 for stainless steel gave a cost of £16,000 to £32,000 per 4m box. The cost of a carbon steel 4m box is estimated as £11,000 based on extrapolating the HHISO cost of £8000 (LLWR Ltd 2014) by the ratio of the package masses.

Transport Costs

The LLW transport costs were assumed to be £600/trip from the 2011 version of the LLW Repository Service Price – Overview. It was assumed that there would be eight 210 litre drums per pallet and that 4 pallets was equivalent to a HHISO for VLLW transport costs. The current joint waste management plans (e.g. Shipton and Falconer 2013) indicate that LLW transport costs are about 2% of the total LLW disposal cost.

The location of a future GDF is unknown hence NDA 2012a assumes a generic ILW transport cost of £1,250/m^3. Hence this was assumed for the transport of 4m boxes of ILW metals to a future GDF.

Disposal Costs

The LLW Repository Ltd website shows that the LLW disposal volume cost norm has increased from £2,911/m^3, used in the WAGR case study, to £2,990/m^3 (LLWR Ltd 2014). By comparison, the VLLW disposal cost has remained constant at a planning norm of £500/m^3 in LLWR Ltd 2014 but was as high as £500 to £2000/tonne in LLWR and Entec 2009b.

A generic design currently exists for a UK GDF. The final GDF design will depend on the location and its geology. HAW disposal costs are currently speculative and therefore highly uncertain but several options exist, namely:

a) A historic conditioned waste volume cost of £4,000/m^3 is presented in the macro-economic study NDA 2007. When converted to a packaged volume

cost and inflated to current prices it gives a disposal volume cost of £5,000 to £6,240/m^3

b) A cost of £15,000 to £20,400/m^3 was estimated for the disposal of ILW from the proposed new nuclear power station programme (DECC 2009)

c) A disposal volume cost as £3840/m^3 for shielded ILW disposal packages (e.g. 4m stainless steel boxes with internal concrete shielding) and £10,260/m^3 for unshielded disposal packages (e.g. stainless steel drums 3m^3 boxes or drums) as presented in NDA 2012a. A baseline cost of £8,380/m^3 for mixed shielded and unshielded disposal packages was also given in NDA 2012a.

Since the ILW packages for long-term storage of ILW that could decay to LLW and ILW metal disposal are assumed to be 4m boxes, the £3840/m^3 for shielded ILW packages was assumed. Given the uncertainty in these costs the NDA 2012a baseline cost of £8,380/m^3 was also included. These assumptions give a lower and upper limit to the ILW metals disposal costs.

Size Reduction and Decontamination Costs

The NDA and LLW Repository Ltd costs only include the package costs, transport costs and disposal costs. They do not make any estimate of the size reduction and decontamination costs required at nuclear sites to enable radioactive metals to be packaged for disposal. These data are not readily available from the literature. Hence these costs were based on the two historical American studies as discussed for the WAGR case study in Chapter 7, i.e. £570/tonne for decontamination and £385/tonne for size reduction based on CDDUEF 1996 and USDOE 2001.

Total Costs

Estimated total costs for all the disposal and recycling scenarios are presented in Table 8-2 alongside the costs for each category of radioactive metals. They are rounded to the nearest £10m because of the large uncertainties in the data. Costs for the treatment and recycling, including the benefit of the resale of the melt ingots are included.

Table 8-2 Comparison of UK disposal and UK and International treatment/recycling costs at 2013

Scenario	**ILW metals**	**ILW Metals decayed to LLW**	**LLW Metals**	**VLLW Metals**	**Estimated benefit from scrap values**	**Total Costs**
Baseline Disposal	£1320m to £2480m	N/A	£5580m to £5940m	N/A	N/A	**£6900m to £8420m**
Improved Disposal	£1000 to £1860m	£60m to £80m	£2280m to £2640m	£110m to £160m	N/A	**£3450m to £4740m**
Treatment	£1000 to £1860m	£40m to £50m	£1810m to £2140m	£90m	N/A	**£2940m to £4140m**
Treatment with recycling	£1000 to £1860m	£40m to £50m	£1810m to £2140m	£90m	-£160 to -£290[1,2]	**£2780 to £3850**

Notes
1 The scrap metal values shown are based on the estimates in Table 8-3 and Table 8-4 at 2015 prices
2 The treatment and recycling costs do not account for the savings in products made from melt ingots for the nuclear industry.
N/A Not Applicable

The LLW disposal costs dominate as there is a total of nearly 800,000tonne of LAW metals but only ~100,000tonne of ILW metals. Also, the disposal volume costs for shielded ILW 4m boxes packages to a GDF (NDA 2012a) is only a factor of ~1.3 higher than the current LLW disposal volume costs, and the ILW baseline disposal volume cost a factor of ~2.8 higher. Hence, the near eight fold difference in LAW metal mass for disposal outweighs the higher ILW disposal cost.

The results in Table 8-2 show that the improved disposal saves £3.5b - £3.7b by assuming increased ILW and LLW metal disposal loading, LLW disposal of 9% of the ILW metal decayed, or decontaminated, to LLW and the segregation of VLLW metal for significantly cheaper disposal. The benefit of treatment for recycling of LAW metals saves a further £0.5b - £0.6b. ILW containment and disposal costs are not affected by treatment and recycling hence they represent a high fixed cost.

Cost Recovery from metals treatment

The American steel feasibility study (USDOE 2001) assumed that 3% of the treatment cost could be used for the sale value of the treated metals, thus off-setting the costs. Adopting this value results in an alternative saving of £57m to £67m, but would depend on the prevailing metal market prices. This can be compared with the

estimated benefits from the resale of melt ingots in the open metal markets shown in Table 8-3 and Table 8-4.

The data in Table 8-3 give an indication of the wide range of recent scrap metal prices published on the identified web pages. They depend on the type and grade of scrap that is typically traded in the metals markets and are intended as indicative prices only.

Table 8-3 UK Scrap metal price range at March 2015

Metal	**Relative Price[1]**	**WRAP prices £/tonne[2]**	**Greengate £/tonne[3]**	**Prices for Scrap £/tonne[4]**	**Lets Recycle £/tonne[5]**	**Price for Calculation £/tonne[7]**
Aluminium	300-400	750 – 850	400 – 650	250 - 400	550 - 900	400 – 900
Copper	400	3600	600 – 3300	3300 - 4200	2300 - 3400	600 – 4200
Iron	70-90	N/A	70	40	60 - 80	40-80
Lead	200-500	350 – 400	900	700 - 1000	850 - 900	350 – 1000
Nickel	20000	N/A	N/A	N/A	N/A	9100 - 9800[6]
Zinc	350 - 400	N/A	N/A	100	500	100 – 500
Carbon Steel	100	80 – 125,		50	100 - 125	80 – 125
Stainless Steel	600	N/A	600 – 900	300 - 400	N/A	300 – 900

Notes – N/A not available
1 – Metal values relative to carbon steel/mild steel (Table 5.1 of Chapter 5)
2 – From http://www.wrap.org.uk/content/ by metal type, 5/3/15
3 – From http://www.greengatemetal.co.uk/scrapmetalprices guide values for January and February 2015, 5/3/15
4 – From http://www.pricesforscrapmetal.co.uk/prices.html at 5/3/15
5 – From http://www.letsrecycle.com/prices/metal by metal type for January and February 2015, 3/5/15
6 – From http://www.recycleinme.com/scrapresources/LME%20Cash%20Prices-Nickel_details. aspx. It was difficult to get scrap nickel prices. Only the London Metal Exchange (LME) gave prices. The lower nickel price was taken from a graph of Opening, High, Low and Closing (OHLC) prices. The high value was taken from graph showing the trend in nickel prices from 12/5/14 to 1/1/15. The price was $14,900/tonne on 5/3/15 and US dollar to pounds Sterling exchange rate on the day was 1.52, and gave £9800/tonne.
7 The lowest and highest prices from the identified web sites listed above.

The data in column 2 are the relative prices of metal per tonne compared to carbon steel from Ashby and Jones 2006 (as presented in Table 5-1 of Chapter 5). They are presented here to give a perspective to the scrap metals data in columns 3 to 6, which were taken from the web sites identified on 5/3/15. The data in column 7 give the lowest and highest scarp values from columns 3-6 as a range of values that might be obtained by selling treated melt ingots back into the UK markets.

These data can then be applied to the UK radioactive metals inventory data to estimate the value of the treated ingots for each metal. The results are summarised in Table 8-4, and again these data should be regarded as indicative only.

Table 8-4 Estimated cost recovery for selling treated UK radioactive metals into the open metal markets at March 2015

Metal	**Amount (tonne (te))**	**Recovered metal (te)**	**Low Scrap Price (£/te)**	**Low Value Recovered Metals (£m)**	**High Scrap Price (£/te)**	**High Value Recovered Metals (£m)**
Aluminium ILW	1200	108	400	0.04	900	0.10
Aluminium LLW	2353	2117	400	0.85	900	1.91
Aluminium VLLW	14647	13183	400	5.27	900	11.86
Copper ILW	405	36	600	0.02	4200	0.15
Copper LLW	3875	3488	600	2.09	4200	14.65
Copper VLLW	1869	1682	600	1.01	4200	7.06
Unspecified ILW	7423	668	80	0.05	125	0.08
Unspecified LLW	171561	154405	80	12.35	125	19.30
Unspecified VLLW	0	0	80	0.00	125	0.00
Lead ILW	822	74	350	0.03	1000	0.07
Lead LLW	6486	5837	350	2.04	1000	5.84
Lead VLLW	3694	3325	350	1.16	1000	3.32
Nickel ILW	3102	279	9100	2.54	9800	2.74
Nickel LLW	7650	6885	9100	62.65	9800	67.47
Nickel VLLW	0	0	9100	0.00	9800	0.00
Stainless Steel ILW	40000	3600	300	1.08	900	3.24
Stainless Steel Surface Contam'd LLW	59402	53462	300	16.04	900	48.12
Stainless Steel Activated LLW	24000	21600	300	6.48	900	19.44
Stainless Steel VLLW	36598	32938	300	9.88	900	29.64
Mild Steel ILW	49000	4410	80	0.35	125	0.55
Mild Steel Surface Contam'd LLW	270233	243210	80	19.46	125	30.40
Mild Steel Activated LLW	86000	77400	80	6.19	125	9.68
Mild Steel		66390	80	5.31	125	8.30

Metal	Amount (tonne (te))	Recovered metal (te)	Low Scrap Price (£/te)	Low Value Recovered Metals (£m)	High Scrap Price (£/te)	High Value Recovered Metals (£m)
VLLW	73767					
Zinc ILW	48	4	100	0.00	500	0.00
Zinc LLW	353	318	100	0.03	500	0.16
Zinc VLLW	14647	13183	100	1.32	500	6.59
Total	**879135**	**708602**		**156.26**		**290.68**

Key – Contam'd is contaminated

The unspecified metals are taken to be mostly iron and steel, hence the carbon steel price is assumed. Only 9% of the ILW metal is assumed to be recoverable for treatment and recycling. About 5% LLW and VLLW metals were assumed to be not financially viable to sort for treatment and a further 5% were assumed to be residual waste from the treatment process. Hence the LLW and VLLW metals recovery is taken to be 90%.

Although nickel masses are low their scrap value dominates the Table 8-4. This is because of the high scrap value allocated to nickel. However, given that it was difficult to get scrap prices for nickel the value may be significantly inflated compared to the other metals. Hence there is a high uncertainty in the value estimated.

There is about 4 to 5 times as much carbon steel as stainless steel, however, the scrap value of stainless steel is significantly higher than carbon steel. Hence the scrap values are very similar. Both aluminium and copper appear to give a significant return, but lead and zinc give relative small returns.

Overall, there is a potentially significant financial return from selling treated radioactive metals into the scrap markets. The results in Table 8-4 broadly support the America feasibility study. The sale of the treated metals could recover a small percentage of the treatment cost.

8.4 Analysis Review

The results above demonstrate the environmental benefits that can be achieved through applying the current UK LLW policy, strategy and plans for low activity metals. The results also imply that significant benefits could be obtained if the UK applied similar logic to the ILW metals that decay, or can be decontaminated, to LLW levels.

The UK LLW policy (DEFRA et al. 2007) and the strategic best practicable environmental option (BPEO) for radioactive metals (Studsvik 2006a and 2006b) support using international recycling facilities in the short term but state that a UK facility should be considered in the longer term. Currently the waste problem is transferred to another country and they accrue the major benefits in employment and experience. However, should availability of international treatment facilities change in the future due to the non-acceptance of UK metals or the non-availability of the facilities through accident or national policy, then the current option could be at risk. As previously mentioned there is no radioactive metal melting facility in the UK. Hence loss of access to one treatment facility could only be met by negotiating new contracts with remaining facilities or the construction a UK melting facility. This is discussed in more detail in Chapter 10.

Although the strategic BPEO for VLLW (LLWR and Entec 2009a and 2009b) includes statements on VLLW metals complying with current policy and strategy the low cost of VLLW disposal as between £300 and £700/m^3 suggests that it is likely to be the preferred business case. The estimated cost of £385/tonne for size reduction alone significantly increases the VLLW disposal costs. Hence, there is a dilemma in not meeting policy requirements for VLLW on the basis of cost. This could be overcome by financial incentives promoting treatment and recycling of VLLW rather than disposal. One option would be to increase VLLW metals disposal cost thus promoting treatment. Another option would be to allow the mixing of VLLW metals with LLW metals approaching the upper LLW radiological limits when melting. This could help ensure that the

melt ingots meet exempt limits. This could also be applied to melting activated LLW metals or decayed ILW metals. This is a potential area for further research.

8.5 Chapter Summary

The overall environmental impacts and are dominated by ILW disposal, disposal package metals and package grouting. The overall costs are dominated by LLW disposal rather than ILW disposal because of the much larger quantity of LLW metals. ILW, LLW and VLLW metal environmental impacts and financial costs can be reduced significantly by increasing waste container loading for disposal and by treatment of LAW metals to minimise disposal. Recycling LLW and VLLW metals, thus avoiding future metals for new products for the nuclear industry, or as scrap for general recycling, offer very substantial environmental improvements compared to disposal. Segregating VLLW metals and treating ILW metals that can decay, or be decontaminated, to LLW lead to further marginal costs savings and environmental benefits. These points are summarised pictorially in Table 8-5 to show the relationship between environmental impact and estimated costs.

Table 8-5 Summary of potential environmental impacts and financial costs for UK radioactive metals inventory disposition options

	Baseline Disposal	Improved Disposal	Treatment without recycling	Treatment with recycling
Potential Environmental Impact ILW	●	○	○	○
Potential Environmental Impact LLW	◉	○	◉	⊙
Potential Environmental Impact VLLW	◉	⊙	⊙	⊙
Direct Financial Cost ILW	◉	○	○	○
Direct Financial Cost LLW	●	◉	◉	⊙
Direct Financial Cost VLLW	○	⊙	⊙	⊙

Key: Lowest ⊙, Low ◉, Medium ○, High ◉, Highest ●

Baseline disposal gives the highest potential environmental impact. Improved disposal is about 62% of the baseline, international treatment without recycling and with recycling are about 47% and 38% of the baseline respectively (from Figure 8-4). The Baseline disposal gives the highest cost of £6.9b - £8.4b, improved disposal a gives a medium cost of £3.5b – £4.7b (from Table 8-2). International treatment with and without metals recycling give the lowest costs at £2.8b - £4.1b (from Table 8-2).

Having demonstrated the approach for the UK radioactive waste metals inventory attention will now turn to considering the assessment of the long term storage and eventual disposal of depleted, natural and low enriched uranium.

9 UK Depleted Uranium Storage and Disposal Analysis

Chapters 7 and 8 showed the potential opportunity to conserve scarce low and intermediate level radioactive waste (LLW and ILW) disposal volume by treating radioactive metals for recycling and/or re-use in the nuclear industry. Chapter 8 also showed the dominance of ILW disposal impacts for a future UK geological disposal facility (GDF) and the savings in steel and cement grout from improved container utilization in addition to the metals treatment. This chapter builds on these latter themes by applying the LCA approach to a potentially significant future UK radioactive waste management issue; namely the long-term interim storage and disposal of depleted, natural and low enriched uranium (DNLEU).

The UK currently considers DNLEU as a "… zero-value asset radioactive material…" rather than waste (NDA 2013b). If DNLEU is designated as waste in the future for UK nuclear policy, commercial or environmental reasons it would represent "… a significant fraction (~17%) of the total volume of UK higher activity materials…" (NDA 2013b). Hence its disposal to a GDF or a near surface disposal facility would be a significant issue.

9.1 Overview of UK Civil Depleted, Natural and Low Enriched Uranium

This section discusses the background to the UK civil DNLEU, the current inventory (at 2015) and location of the main stores of DNLEU and its potential end-of-life uses or disposition options.

9.1.1 Background to Depleted, Natural and Low Enriched Uranium

Historically, the UK nuclear fuel cycle started with the conversion of uranium concentrate (yellow cake) at Springfields (i.e. Stage 3, Figure 1-1 Chapter 1). Yellow cake is a mixture of uranium dioxide (UO_2), uranium trioxide (UO_3) and triuranium octoxide (U_3O_8) and the quality can vary from 75% to 85% uranium concentrate depending on the residual impurities from milling and initial refining (Sovacool 2008, Norgate et al 2013, www.wise-uranium.org/Rup.html). Springfields converted

the yellow cake to uranium tetraflouride (UF_4) or uranium hexafluoride (UF_6). The UF_4 was made into un-enriched uranium metal for Magnox fuel elements and the UF_6 sent to Capenhurst for enrichment. Conversion is now predominately carried out abroad and Magnox fuel is no longer required. Once enriched, the UF_6 is returned to Springfields for reconversion to UF_4 and thence to UO_2 ceramic fuel pellets for AGR fuel elements made at Springfields and civil PWR fuel elements made abroad.

A by-product of the enrichment process is depleted UF_6, which is retained for future re-enrichment. The grade of depleted UF_6 produced depends on the purity of the input product and the desired enrichment level. The input product maybe un-irradiated UF_6 from Springfields or UF_6 produced from reprocessed nuclear fuel at Sellafield. The latter will contain fission product contamination from the initial irradiated fuel. It will also contain U232 which emits high energy gamma rays and presents a significant radiation problem (Butler 2015 pers. comm.). These may cause radiological problems or neutron absorption problems. Existing depleted UF_6 can also be blended with the gas stream to achieve the required enrichment. For civil nuclear power stations the enrichment in normally 3.75% to 5% uranium 235. One kg of enriched UF_6 results in ~5kg of depleted UF_6 for a 3.57% enrichment level, according to the Urenco UK Ltd website (Urenco.com, 2014). However the quantity of depleted UF_6 can vary from 5.7 to 7kg according to Sovacool 2008 and Fthenakis and Kim 2007 or as high as 8 to 9kg for 5% enriched UF_6 according to WNA 2013. Hence a considerable amount of depleted UF_6 by-product arises from the enrichment process.

Magnox and AGR spent nuclear fuel reprocessing at the backend of the nuclear fuel cycle (i.e. Stages 7 and 8, Figure 1-1, Chapter 1) results in the recovery of plutonium and uranium. The uranium materials recovered are usually un-enriched or slightly enriched UO_2 or UO_3. These are stored as Magnox depleted uranium at Capenhurst and Thermal Oxide Reprocessing Plant (THORP) product uranium at Sellafield. These materials are stored for re-enrichment for fuel fabrication for future nuclear power stations.

9.1.2 Depleted Natural and Low Enriched Uranium Inventory

DNLEU has a very low activity and could not meet the conditions of acceptance of the LLW Repository near Drigg in Cumbria if it had to be disposed. This is due to the exceptionally long half-life of the Uranium-234, 235, 236 and 238 isotopes and their decay products such asThorium-230. These radionuclides actually result in an in-growth of radioactivity with time (NDA and DECC 2011). Hence DNLEU, unlike other materials, actually increases in activity in the very long term and raises a problem for disposal. If most of the DNLEU is eventually classed as waste it will need to be disposed as higher activity waste (HAW) at a future GDF or near surface disposal facility. In the interim it is kept in approved long-term interim storage packages at a number of sites pending re-enrichment.

DNLEU was first included in the 2007 UK radioactive waste inventory. Estimates for a Derived Inventory and Upper Inventory for the proposed future GDF are presented in the original HAW policy (DEFRA et al. 2008) and Nuclear Decommissioning Authority (NDA) discussions with waste packagers (NDA 2013a).

The original UK HAW policy (DEFRA et al 2008), strategy (NDA 2013b) and option analyses (NDA 2007, Hickford et al 2012 and Wilson et al 2012) build on international concepts for the management of depleted uranium (OECD/NEA and IAEA 2001). The overall principle in each case is to minimise the disposal of DNLEU as far as practicable.

The current stock (at 2015) of civil DNLEU is estimated as 160,000tU. The quantities currently retained at Urenco UK Ltd (UUK) and Capenhurst Nuclear Services at Capenhurst, Sellafield, Dounreay, Springfields, Harwell and Winfrith are shown in Table 9-1. The Urenco UK Ltd depleted UF_6 tails dominates at 100,000 tU with the Magnox depleted uranium and Capenhurst Nuclear Services depleted UF_6 tails representing the bulk of the remainder at 55,000tU. In addition, NDA 2013b notes "…UUK has indicated that it expects its future nuclear fuel production operations at Capenhurst to generate approximately 5,000tU per annum of depleted UF_6 tails, which suggest that the total future stockpile could be very large". Hence

the Urenco UK Ltd depleted uranium is likely to continue to dominate the DNLEU stock into the future.

Table 9-1 UK Civil DNLEU Owners, Location and Quantities at 2015

Material	Owner	Current Location	Total Mass (tU))
Depleted UF_6 tails	Urenco UK Ltd	Capenhurst	100,000
Depleted UF_6 tails	NDA	Capenhurst	25,000
Magnox depleted uranium	NDA	Capenhurst	30,000
THORP product uranium	NDA/EDF Energy	Sellafield (Low enriched uranium)	5,000
Miscellaneous DNLEU	NDA	Various Sites	2,000
Total			162,000[1,2]

[1] NDA 2013b notes that the 1998 Defence Review estimated that the Ministry of Defence (MoD) DNLEU was about 15,000 tU hence makes up the balance of the ~180,000 tU (tU= metric ton or tonne of uranium). These values are significantly higher than the NDA 2007, Table 1 (p4).
[2] Hickford et al, 2012, assumes 160,000 tHeavy Metal (i.e. U), hence only ~1.2% difference.

Depleted UF_6 is currently stored in either internationally approved carbon steel Type 48 transport cylinders or historic Type 0236 cylinders, Magnox depleted uranium and miscellaneous UO_3 are stored in a mixture of carbon steel and stainless steel 200 litre drums and THORP product uranium in stainless steel 50 litre kegs with an outer stainless drum if slightly enriched (Jones pers. comm. 16/5/14). The estimated number of containers and their storage volume are shown in Table 9-2.

Table 9-2 Current (at 2015) UK Civil DNLEU storage container type, mass and volume per location

DNLEU Masses and container numbers	Urenco UK Ltd	Capenhurst Nuclear Services	Sellafield	Misc. Locations
UF_6 Mass (tU)	100,000	25,000	0	0
No. of Type 48 Cylinders	8000	1,500	0	0
Total Mass of Type 48 Cylinders (tonne)	18,870	3540	0	0
Total Volume of Type 48 Cylinders (m^3)	32,230	6060	0	0
No. of Type 0236 Cylinders	0	10,000	0	0

DNLEU Masses and container numbers	Urenco UK Ltd	Capenhurst Nuclear Services	Sellafield	Misc. Locations
Total Mass of Type 0236 Cylinders (tonne)	0	2,500 (pers. comm. 16/5/14)	0	0
Volume of Type 0236 Cylinders (m^3)	0	72,00	0	0
Uranium Oxide (UO_3) Mass (tU)	0	30,000	5,000	2,000
No. of 50 Litre Drums	0	0	25,000 (with assumed 12,500 over-packed)	0
No. of 200 litre Drums	0	60,000	0	4,000
Total Mass of 50 litre (@8kg*) (tonne)	0	0	200, with 100 for overpack	0
Mass of 200 litre Drums (@ 18.6kg) (tonne)	0	1,120	0	75
Volume of 50 litre Drums (m^3)	0	0	1660 (with 50% overpack) - 3470	0
Volume of 200 litre Drums	0	17,760	0	1,150

Additional data on DNLEU containers is given in Table A-9 to Table A-21.

The UF_6 cylinders are stored externally at Capenhurst, hence subject to corrosion. The Magnox depleted uranium 200 litre drums are stored internally in a refurbished building at Capenhurst. The THORP product uranium 50litre kegs are stored in a bespoke store at Sellafield. The storage arrangements of the miscellaneous UO_3 at various sites are unknown. Hence, there may be some issues with the acceptability of current stores to meet the requirement for long-term interim storage for up to 100 years. The total current storage volume is estimated at about 67,900m^3, excluding the annual Urenco UK Ltd UF_6 production mentioned above.

UF_6 is not the preferred waste form for long-term interim storage (LTIS) because of the chemical toxicity issues in an accident (OECD/NEA and IAEA 2001, NDA 2013a). Nor is it an acceptable waste form for disposal at a future GDF (NDA 2012d) as it is not immobilised. Urenco UK Ltd is therefore currently building a Tails Management Facility at Capenhurst to deconvert (de-fluorinate) its UF_6 to more passively safe U_3O_8 powder and recover Hydrogen Fluoride as a co-product for sale in the open market. U_3O_8 powder is conventionally stored in Areva DV70 $3m^3$ 'green boxes' in modular steel and concrete stores (as for example at the COVRA VOG facility in the Netherlands) or in a modified mine (e.g. Bessines in France (Capus and Durante 2007). The Urenco Tails Management Facility will incorporate a uranium oxide store to house the DV70 boxes, but its design specification is unknown.

These developments significantly reduced the chemical toxicity hazard for the site and allow Type 48 cylinders to be cleaned and reused, thus reducing the need for new cylinders. These have been significant local stakeholder and regulator concerns for some time. However, it is costly to do - will incur significant re-conversion costs (i.e. an extra £3.5k/te more than using UF6) – requires large structures for operating plant and stores with associated steel and concrete inventory – requires additional energy for building, operating, maintaining and decommissioning – requires new and additional long-term interim storage containers. This is potentially a very interesting area for future research.

The Tails Management Facility will also include a cleaning and re-certification facility for Type 48 cylinders to return as many as possible to service. The Type 48 cylinders that cannot be re-used or re-sold will be disposed or treated for recycling along with the redundant Type 0236 cylinders. The U_3O_8 can be readily reconverted to UF_6, and the DV70s have a proven pedigree in France and the Netherlands. Hence, the long-term storage of U_3O_8 seems a practical business option for Urenco UK Ltd. The company can safely retain the U_3O_8 for reconversion for enrichment when needed, or as the quality of uranium ore deposits degrade over the next 50

years (as forecast in Fthenakis and Kim 2007, Sovacool 2008, Norgate et al 2013 and Warner and Heath 2012) or for disposal if eventually classed as waste.

The legacy UF_6 managed by Capenhurst Nuclear Services may also be deconverted at the Tails Management Facility (ONR 2014). However, this is likely to be from 2020, i.e. after Urenco has deconverted its British, Dutch and German UF_6.

9.1.3 End-of-life Disposition Options

DNLEU is currently a valuable resource rather than a waste. It has several possible uses within the nuclear industry such as re-enrichment, down-blending high enrichment uranium from weapons programmes for civil power station fuel, mixed oxide (MOX) and fast reactor fuel or high density concrete for shielding, waste packages or repository backfill (OECD/NEA & IAEA 2001 and NDA 2013b). Depleted uranium metal also has potential uses in the aviation industry, the oil industry and for munitions (OECD/NEA and IAEA 2001). Hence, the exact end-of –life disposition options for UK DNLEU are not completely clear. However, it is presumed that these alternative use options will only require a small proportion of the DNLEU. Hence the options are not explored in this research, rather it is postulated that as a worst case all the current stock of civil DNLEU will be stored and then disposed.

This research assumes that the DNLEU will be stored as either a mixture of U_3O_8 and UO_3 in a variety of containers as shown in Table 9-3, or uniformly as U_3O_8 in DV70 boxes, pending disposal to a future GDF or near surface disposal facility.

Table 9-3 Package and Storage Volume for Proposed Long-Term Interim Storage of Civil DNLEU at 2015

DNLEU masses and container numbers	**Urenco UK Ltd Stock**	**Capenhurst, Sellafield and other sites**
Total tU_3O_8 Mass (tonne)	118,000	29,500
No. of DV70 Boxes	11,240	2,810
Mass of DV70 Boxes (tonne)	8430	2110
Volume of DV70 Boxes (m^3)	41,590	10,400
Total tUO_3 Mass (tonne)	0	37,000
No of 200 litre Drums	0	64,000
Mass of 200 litre Drums	0	1,195

DNLEU masses and container numbers	Urenco UK Ltd Stock	Capenhurst, Sellafield and other sites
Volume of 200 litre Drum (m^3)	0	18,910
No. of 50 litre Kegs	0	25,000
Mass of 50 litre kegs (tonne)	0	200
Volume of 50 litre kegs (m^3)	0	3470 (with overpacks)

The table indicates that the estimated total storage volume for mixed long-term storage packages is 74,370m^3 i.e. about 9.5% higher than the current storage volume above. However, if all the current DNLEU stock was converted to U_3O_8 and stored in DV70 boxes the estimated storage volume is about 67,700m^3, i.e. virtually the same as the current storage volume estimated above.

It is anticipated that the DNLEU will be removed from the long-term interim storage containers and repacked in stainless steel or carbon steel containers for disposal as HAW to a GDF. However, this option introduces an additional radiation dose to a future workforce from repackaging the waste and decontaminating the redundant storage packages. It also means that the redundant storage packages must be re-used, disposed or treated for recycling. Hence an alternative is considered, over-packing the long-term interim storage packages and disposing them to a near surface disposal facility as suggested in NDA 2013b. This option removes the double handling of DNLEU and managing the redundant DV70 boxes. Also, since a near surface disposal facility would require less excavation the disposal impact and costs should be significantly reduced.

9.2 Disposal Scenarios

This section discusses the repackaging of UK civil depleted, natural and low enriched uranium for disposal as higher activity waste (HAW) to a future geological disposal facility (GDF), or as overpacked HAW to a GDF or low level waste (LLW) to a potential near surface disposal facility.

9.2.1 Repackaged DNLEU as HAW to a Future Geological Facility

Five LCA scenarios were initially considered for the disposal of DNLEU as HAW to a GDF. Hence, the functional unit for these LCAs, and the others LCAs in the chapter, was – the disposal of the UK legacy civil depleted, natural and low enriched

uranium following an indeterminate period of long-term interim storage at sites. The sources and data for these options are summarised in Table 9-4.

Table 9-4 Initial Deep Disposal Scenarios for UK Civil DNLEU (at 2015) investigated in the research

	Deep Disposal Scenario 1	**Deep Disposal Scenario 2**	**Deep Disposal Scenario 3**	**Deep Disposal Scenario 4**	**Deep Disposal Scenario 5**
Scenario Source	Hickford 13H[1]	Hickford 16H[3]	NDA 2009b[2]	NDA 2009b[2]	NDA 2009b[2]
Waste	Cemented	Cemented	Cemented	Unconditioned	Cemented
Container	500l SS drum	CS HHISO with Titanium Liner	SS 4m box (1.3t/m^3)	200 litre SS drum super-compacted SS 4m boxes (5.5t/m^3)	SS 3m^3 drum
No. Containers	148,000	5,400	*7,780*	*382,320 (drums)*[2] *3480 (boxes)*[2]	*28,320*[2]
Cont. Mass (te)	19,240	17,000	*32,680*	*7,110 (drums)*[2] *14,620 (boxes)*[2]	*17,840*[2]
No. Stillage	37,000	0	0	0	*0*
Still. Mass (te)	*37,000*	0	0	0	*0*
Waste Volume (m^3)	*134,090*	*105,300*	*167,907*	*75,209*	*74,174*[2]
Titanium (te)	0	5,000	0	0	0
Cement (te)	26,000	120,000	*172,890*[3]	0	*128,010*[5]

1 – Reference case for NDA 2007 but extrapolated for the current DNLEU inventory.
2 – Based on the Urenco UK estimates from NDA 2009b but scaled to the full DNLEU inventory..
3 – Based on the Hickford et al 2012, scenario 16H option for higher strength rock.
4 – Based on the Hickford et al 2012, scenario 4H option for higher strength rock
5 - Based on the Hickford et al 2012, scenario 13H option for higher strength rock.
Normal text – data direct from option source. Italic text, data extrapolated by mass (te) or volume (m^3) where appropriate from option source.
SS – Stainless Steel, CS – Carbon Steel

In each case it was assumed that all the DNLEU was converted to U_3O_8 for long-term interim storage in Areva DV70 3m^3 mild steel boxes at each site and then transferred to an acceptable HAW disposal packaged. The disposal of the DV70 boxes is discussed in Section 9.2.2. It was also assumed that any grouting of the U_3O_8 in the final HAW disposal packages was done at each site and the packages transported by rail to a GDF. It was further assumed that since DNLEU is low activity waste it was handled and transported consistent with LLW requirements, i.e. without the need for extensive external shielding packages.

The disposal scenarios assumed for depleted uranium are summarised in Box 9-1.

Box 9-1 UK civil cepleted, natural and low enriched uranium disposal options

Repackaged Disposal
Deep Disposal Scenario 1 – 500l Drum (Baseline)
The Deep Disposal Scenario 1 (DDS1) baseline assumed all U_3O_8 was transferred and grouted into 500 litre stainless steel drums for disposal (see

Box1-2 for container summary data)). The drums were then placed in a stainless steel stillage frames (i.e. four drums to a stillage) for transport to a GDF. The basic data were taken from Hickford et al. 2012 (Scenario 13H) and are summarised in Table 9-4.

Deep Disposal Scenario 2 – HHISO/Liner

The DDS2 assumed all the U_3O_8 was transferred and grouted into carbon steel half height ISO (HHISO) freight containers with titanium liners for disposal (see Box1-2). The data were taken from Hickford et al. 2012 (Scenario 16H) and are summarised in Table 9-4.

Deep Disposal Scenario 3 - Grouted 4m Box

The DDS3 assumed all the U_3O_8 was transferred, mixed externally with cement grout and poured in 4m stainless boxes for disposal (see Box1-2). The data were taken from NDA 2009b and are summarised in Table 9-4. This was Urenco UK Ltd's first disposal option in their Conceptual Letter of Compliance (CLoC) application assessed in NDA 2009b.

Deep Disposal Scenario 4 – 200l Drums in 4m Box

The DDS4 assumed all the U_3O_8 was transferred to 200litre stainless steel drums (see Box 1-2). The drums were then assumed to be super-compacted and put in 4m stainless steel boxes for disposal (see Box1-2). Hence, the waste-form was ungrouted 200litre drum compacted pucks packed in 4m boxes. This was Urenco UK Ltd's second disposal option in the CLoC application. The data were again taken from NDA 2009b and are summarised in Table 9-4.

Deep Disposal Scenario 5 - 3m^3 Drum

The final scenario (DDS5) assumed all the U_3O_8 was transferred to 3m^3 stainless steel drums for disposal (see Box1-2). It was assumed that the U_3O_8 mixed with cement grout using an in-drum sacrificial mixing paddle. This was the NDA's suggestion to Urenco UK Ltd CLoC as an alternative to the grouted 4m boxes and the ungrouted super-compacted 200litre drums in 4m boxes. The data were again taken from NDA 2009b and are summarised in Table 9-4.

Over-packed Disposal

Three scenarios were considered. In each case the original long-term interim storage packages presented in Table 9-3 were assumed to be over-packed for transport and disposal. This removes the double handing of the DNLUE for re-packaging the waste for disposal as discussed for Deep Disposal Scenarios 1 to 5.

Deep Disposal Scenario 6 – Overpacked DNLEU as HAW

The DDS6 assumed that the 14,050 DV70 boxes were over packed in 7,025 stainless steel 4m boxes. The 64,000 of 200l drum were packed in 1780 stainless steel (SS) 4m boxes (i.e. 36 to 4m box as described in NDA 2007). The 25,000 of 50l kegs were packed in ~390 special full height ISO (FHISO) freight containers (i.e. 64 to a FHISO from Flynn 2007 and Southern 2011)(see Box 1-2). The 4m boxes and FHISOs containers were then disposed as HAW.

Shallow Disposal Scenario 1 – Overpacked DNLEU as LLW (SS)

The Shallow Disposal Scenario 1 (SDS1) assumed that the DV70 boxes, 200l drums and 50l kegs were over-packed as described in DDS6 but disposed as LLW at a proposed near surface facility.

Shallow Disposal Scenario 2 – Overpacked DNLEU as LLW (CS)

The SDS2 assumed that the DV70 boxes and 200l drums were over-packed carbon steel (CS) 4m boxes and the 50l kegs packaged in FHISO containers as above. These packages were also disposed as LLW at the proposed near surface disposal facility.

No data exist in SimaPro for disposal at a near surface disposal facility hence the scenarios are intended to represent the upper and lower bounds of disposal to such a facility. DDS6 represents the worst case, with the disposal equivalent to HAW disposal at a GDF. SDS1 represents a compromised position where the disposal is equivalent to LLW using the same over-packing arrangement as DDS6. The final scenario assumes that carbon steel overpacks can be used instead of stainless steel overpacks and the disposal is still LLW. This is a rather crude approach and only intended to give a first order approximation in the absence of data for near surface disposal facility impacts.

Case study data and results (indexed in Table D-2), supporting the scenarios are summarised in the MicroSoft Excel spreadsheets on the compact included in Appendix D.

9.2.2 Modelling Assumptions

The general modelling assumptions for the various Swiss disposal impact databases and data assumptions about materials are covered in Chapter 6. However, there are a few specific modelling assumptions discuss in Box 9-2.

Box 9-2 UK civil depleted, natural and low enriched uranium storage and disposal assumptions

Waste Material Assumptions

There is currently a preference for U_3O_8 as the preferred material for long-term storage of the majority of the DNLEU. However, it was assumed that the current business cases and safety cases for Magnox depleted uranium, THORP product uranium and miscellaneous UO_3 allowed storage in their current form and containers. It is also assumed the storage facilities for these

latter materials would continue to meet regulator and industry standards. A further assumption was that UO_3 can be readily converted to U_3O_8 should it become a condition of acceptance for a GDF or near surface disposal facility. Hence these latter materials can be modelled in their current containers as well as U_3O_8 in DV70 boxes.

Container Assumptions

The location in France of the Areva facility producing the DV70 boxes is unknown. Also, the production of UK ILW/HAW approved disposal containers appears to have gone through several iterations in the last two decades. The source of these containers for the potentially disposal of DNLEU several decades into the future is also unknown. Hence, a centralised UK production facility was assumed for all the storage and disposal containers. An average distance, weighted by the mass of radioactive metals at each site, was used for the transport distance between nuclear sites and the LLW Repository in Cumbria and a GDF adjacent to Sellafield for disposal of metals in Chapters 7 and 8. This was a conservative assumption as the majority of LLW and ILW arise at Sellafield. This approach was also used for DNLEU disposal containers. Also, all the DNLEU container models included the Ecoinvent average metal working impact for metal product manufacturing as well the container metal impact, hence was consistent with the previous chapters.

Radioactive metals treatment modelling showed that the road transport impact of containers was low compared to container materials, grouting and disposal impacts. Hence, the results in the previous chapters were relatively insensitive to LLW disposal transport. Given that DNLEU is a low activity waste that can be handled like LLW it was presumed that its transport impacts would not be significant. If DNLEU can be transported by rail rather than road the impact would be further reduced.

A specialised FHISO container has already been designed and used for THORP product uranium transport (Southern 2011). It was assumed that they would be used to transport the 50 litre kegs from Sellafield to a disposal facility. It was further assumed that the FHISO containers could be handled at the disposal facilities and would be grouted prior to emplacement. It is appreciated that this may be a contentious issue due to the size and final weight of the FHISO container but it is only intended as a first order approximation for the disposal of THORP product uranium in this streamlined analysis.

It was further assumed that the redundant DV70 boxes would be thoroughly cleaned and used to dispose of VLLW at a licensed facility. The actual VLLW disposed is not of interest as only the redundant DV70 disposal volume is modelled to complete their life cycle.

9.3 Depleted, Natural and Low Enriched Uranium Storage and Disposal Potential Environmental Impact Results

This section presents and discusses the results of repackaged disposal of DNLEU as HAW to a future GDF or as LLW to a potential near surface disposal facility.

9.3.1 Repackaged DNLEU Disposal to a GDF

Deep Disposal Scenario 1 – 500l Drum (Baseline) Results

The results for the baseline scenario assuming stainless steel 500l drums and stillage frames are shown in Figure 9-1.

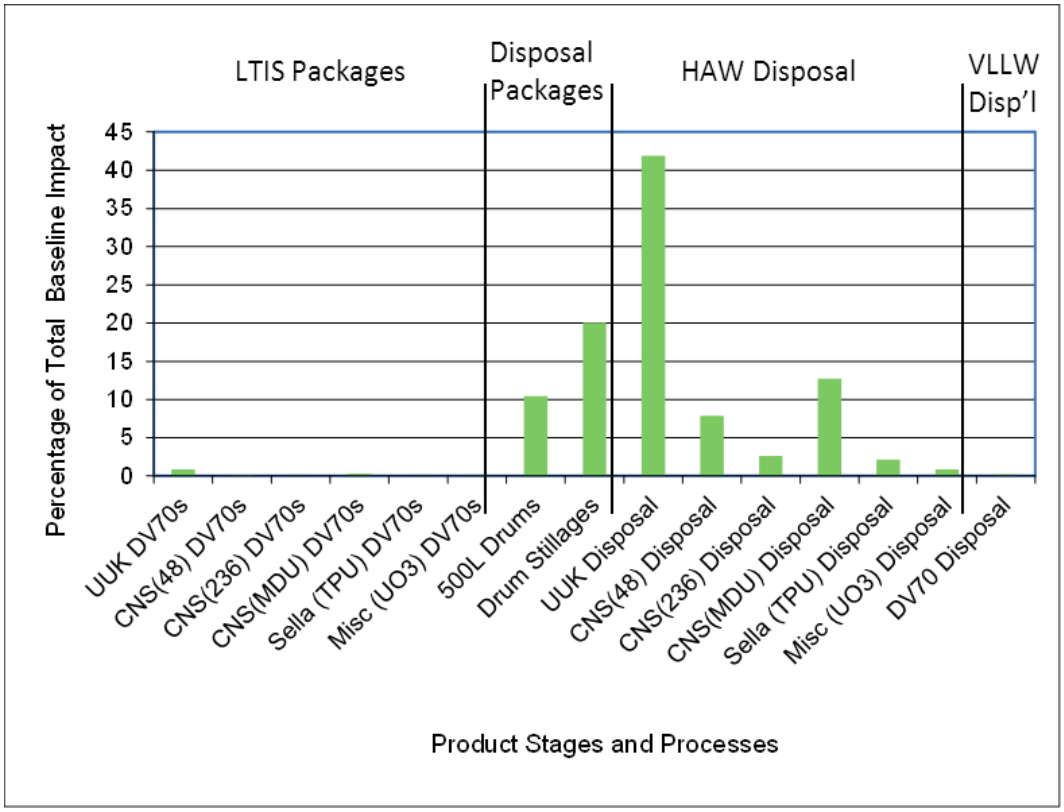

Figure 9-1 500l Drum (Baseline) scenario environmental impacts (from Eco-Indicator 99 (Hierarchist/Average)) for all DNLEU as U_3O_8 in DV70 boxes repackaged in 500 litre drums for HAW disposal to a future GDF. The redundant DV70s are disposed as VLLW. Where UUK is Urenco UK Ltd, CNS is Capenhurst Nuclear Services, (48) and (236) represent Type 48 cylinder and Type 0236 cylinder UF_6 deconverted to U_3O_8 respectively, MDU is Magnox depleted uranium, TPU is THORP product uranium from Sellafield (Sella) and Misc is the miscellaneous UO_3 from various sites. LTIS is long-term interim storage.

Figure 9-1 shows the potential environmental impacts as a percentage of Total Baseline Impact (TBI), which was estimated as 3.2E8Pt. The figure presents the results for the long-term interim storage packages, disposal packages, HAW disposal and the disposal of the redundant DV70 boxes as VLLW. The total HAW disposal impact is about 68.1% of the TBI. The Urenco UK Ltd disposal impact dominates at

~42% TBI followed by the Magnox depleted uranium at ~12.8% TBI and the total Capenhurst Nuclear Services legacy U_3O_8 at ~10.5% TBI.

The THORP product uranium and miscellaneous UO_3 disposal impacts are negligible. The disposal packages impact is about 30.5% TBI, with the stillage frame impact about double the drum impact. Both these impacts are significant and comparable with Magnox depleted uranium and Capenhurst Nuclear Services disposal impacts. The combined DV70 package impacts and their VLLW disposal impacts are negligible at ~1.4% TBI. This suggests that the choice of DV70 boxes is not important in the overall baseline life cycle.

The Eco-Indicator 99 (Hierarchist/Average) normalised environmental impact category results are shown in Figure 9-2.

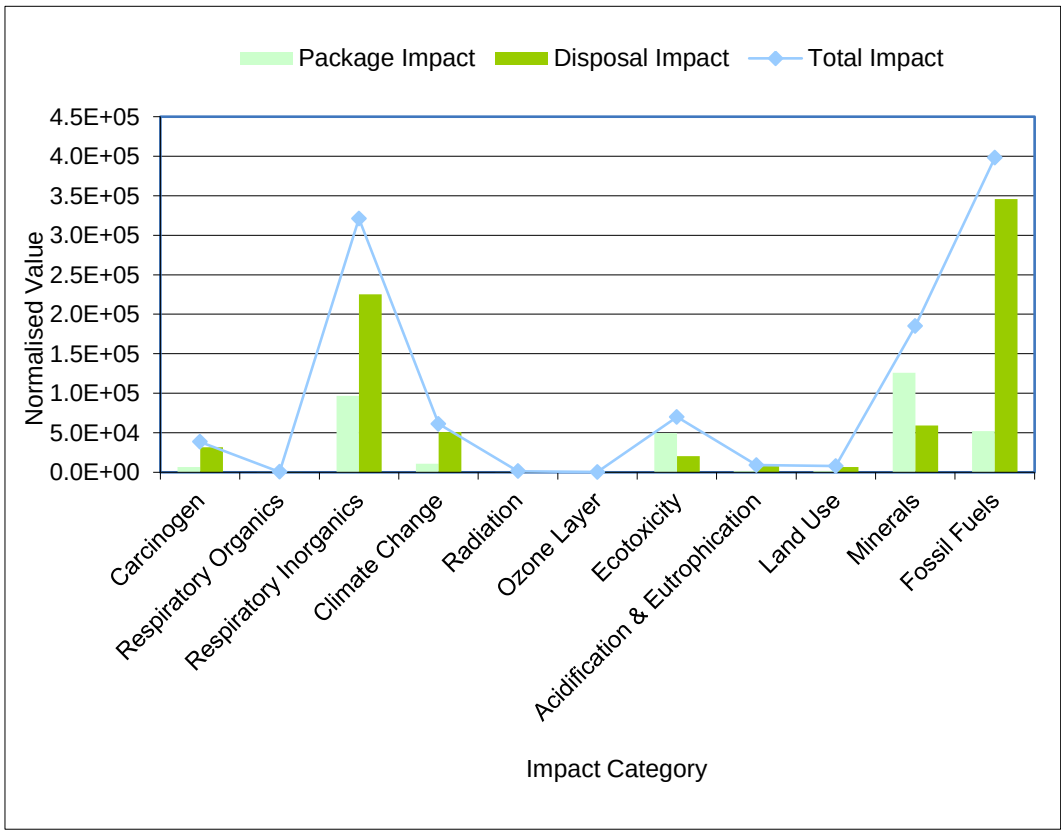

Figure 9-2 500l Drum (Baseline) scenario normalised package and disposal environmental impact category results (from Eco-Indicator 99 (Hierarchist/Average)) for DNLEU disposal as HAW to a future GDF. The package impact includes the long-term interim storage packages (DV70 box) and disposal packages (500l Drum). The disposal impact includes the HAW disposal to a GDF and VLLW disposal of the DV70s.

The overall impact in Figure 9-2 is dominated by the potential Fossil Fuels, Respiratory Inorganics, Minerals, Climate Change and Eco-toxicity then the Carcinogen impacts. The remaining impact categories are negligible.

Environmental Impact Category Results for HAW

The disposal impact dominates in all categories expect Eco-toxicity and Minerals. This was presumed to be due to material and production impacts for the large number of carbon steel DV70 boxes but especially the stainless steel 500 litre drums and stillage frames. Fossil Fuels and Respiratory Inorganics disposal impacts are significantly higher than the package impacts due to the energy and atmospheric discharges from the excavation and backfill of HAW disposal facility vaults.

The Eco-indicator 99 (Hierarchist/Average) weighted impact results are presented in Figure 9-3.

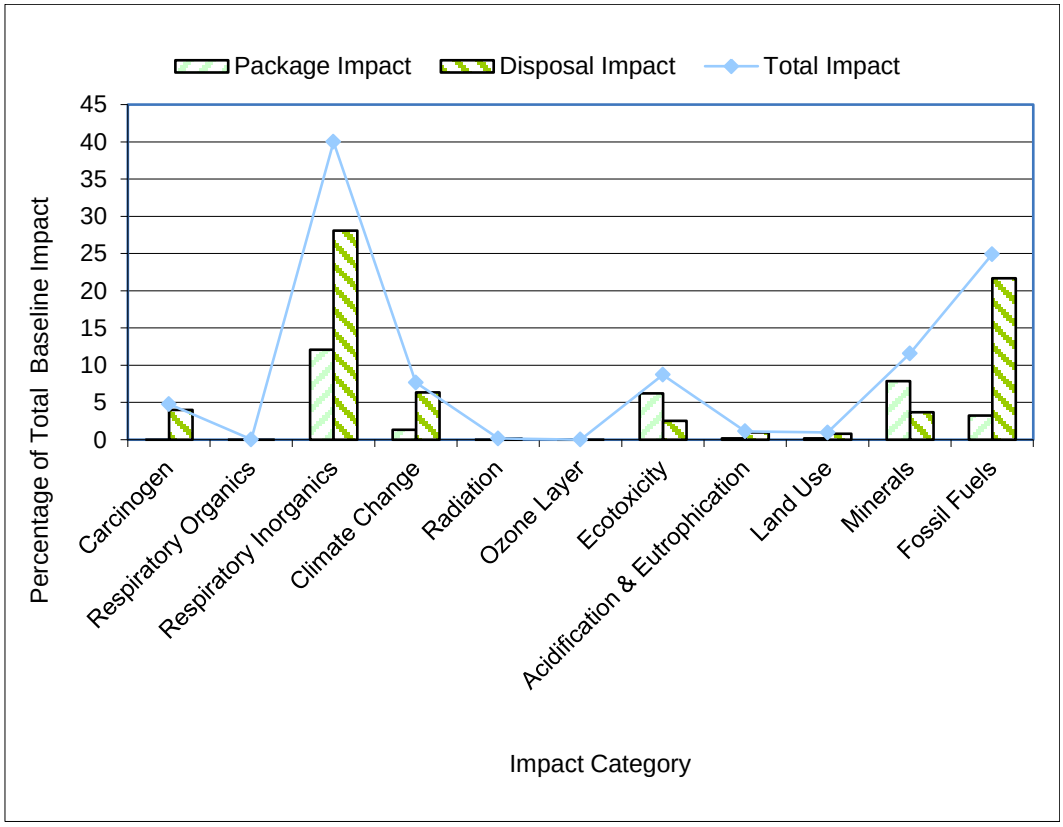

Figure 9-3 500l Drum (Baseline) scenario weighted package and disposal environmental impact category results (from Eco-Indicator 99 (Hierarchist/Average)) for DNLEU disposed as HAW to a future GDF. The results are presented as a percentage of Total Baseline Impact (TBI).

The underlying pattern in the figure is similar to the normalised value presented in Figure 9-2, but the Mineral and Fossil Fuel values are now reduced. This is because these impact categories have half the weighting to the other impact categories in the Hierarchist cultural perspective as discussed in Chapters 7 and 8. Hence, the Respiratory Inorganic impact dominates over the Fossil Fuel impact in the weighted results. The order of the remaining impact categories is the same as Figure 9-2 but there is a smaller margin between the Minerals impact and those for Eco-toxicity and Climate Change. The weighted results show the disposal impacts are about two thirds of the Baseline total impact and the package impacts about one third of the Baseline total impact.

Comparison of GDF Disposal Product Stage/Process Results

Having presented the baseline scenario results the combined results of all the GDF scenarios are presented in Figure 9-4. The figure shows the distribution of long-term interim storage package impacts, the various disposal packages impacts plus the disposal impacts for HAW in a future GDF.

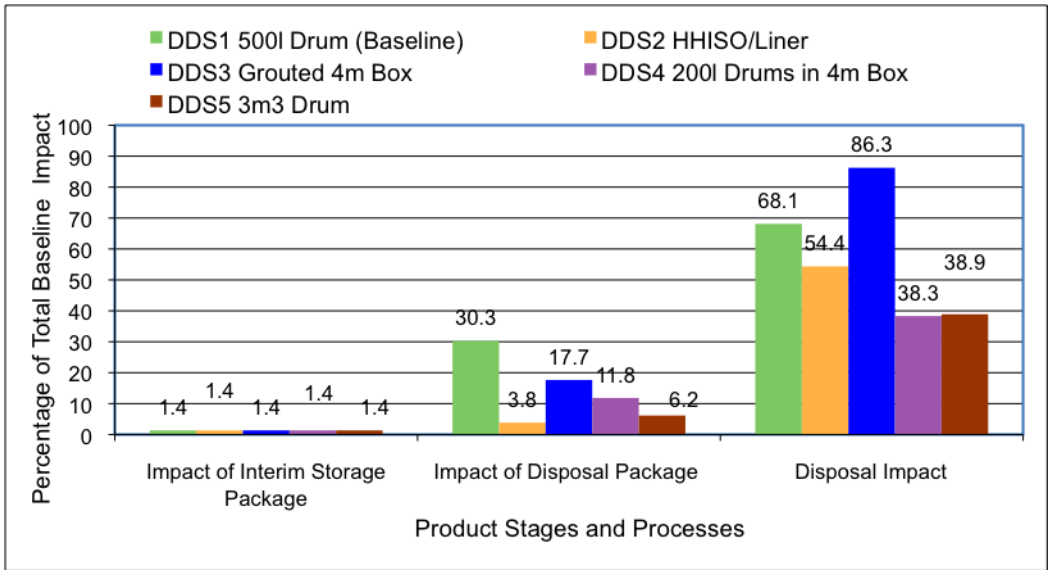

Figure 9-4 Comparison of the interim storage package impacts, the disposal package impacts and disposal impacts (from Eco-Indicator 99 (Hierarchist/Average) for the five deep disposal scenarios for DNLEU as HAW to a GDF. The interim storage packages were DV70 boxes, the disposal packages were as identified by each scenario.

The impacts are dominated by disposal. The DDS 3 (Grouted 4m Box) disposal impact is ~18% higher than the DDS1 (500l Drum (Baseline)) disposal, but their package impact is ~13% lower than the 500 litre drum scenario. Hence the total life

cycle impact for the Grouted 4m Box scenario is about 5% higher than the baseline at 3.37E8Pt.

The DDS2 (HHISO/Liner) scenario offers about a 14% reduction in disposal impact compared to the baseline. Carbon steel HHISOs with titanium liners, rather than 500 litre stainless steel drums and stillage frames, reduces the package impacts by a factor of about 8. Hence the total life cycle impact for DDS2 (HHISO/Liner) is about 40% lower than the DDS1 500l Drum (Baseline) at 1.9E8Pt.

The DDS4 (200l Drums in 4m box) and DDS5 ($3m^3$ Drum) scenarios both offer about a 40% reduction in disposal impact compared to the DDS1 baseline. They also give a factor of ~2.6 to 4.9 reduction in package impacts respectively. This is due to the better utilisation of the waste containers and hence lower disposal volume. Their total life cycle impacts are about 48% and 54% lower than DDS1 500l Drum (Baseline) at 1.64E8Pt and 1.48E8Pt respectively.

The disposal package impacts are a significant proportion of the net life cycle impact but considerably lower than the disposal impacts. This shows the variation in impact associated with stainless steel HAW containers. The DDS2 (HHISO/Liner) impact is significantly smaller than the other disposal packages impacts since they are made of carbon steel rather than stainless steel. The DV70 long-term interim storage package impacts are trivial compared to the other impacts in all cases. This confirms that the choice of long-term interim storage package is not important in the disposal of DNLEU as HAW disposal to a GDF.

Comparison of GDF Disposal Impact Category Results

Figure 9-5 shows that the Fossil Fuel impact dominates the normalised environmental impact category results in all five scenarios followed by the Respiratory Inorganics, Minerals, Climate Change and Eco-toxicity the Carcinogens. All the other impact categories results are negligible. This is consistent with the pattern of results for the baseline case in Figure 9-2.

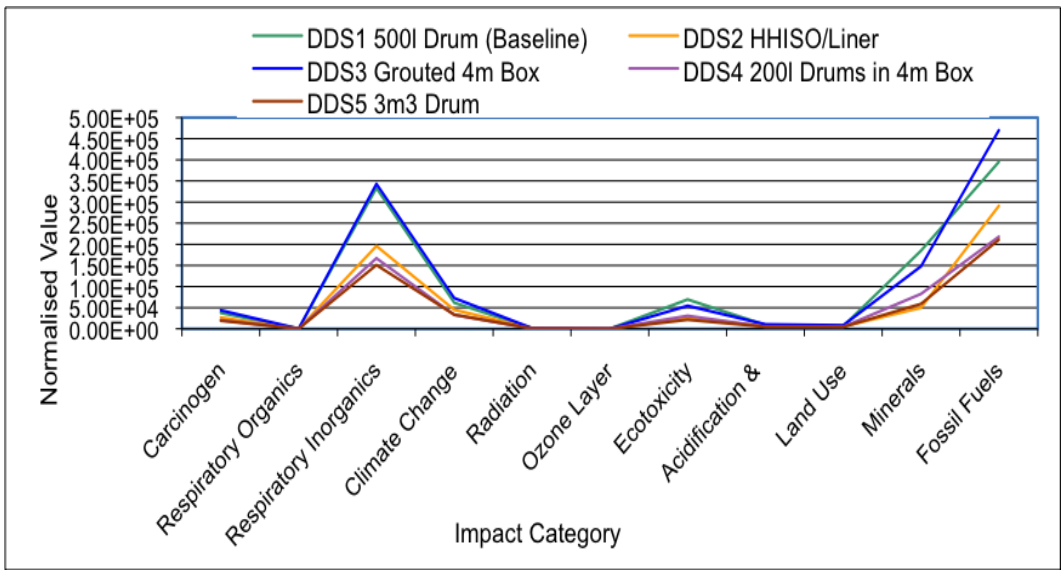

Figure 9-5 Comparison of Eco-indicator 99 (Hierarchist/Average) normalised impacts for the five HAW deep disposal scenarios for DNLEU.

The Eco-indicator 99 Hierarchist/Average weighted impacts, as a percentage of Total Baseline Impact, for the five deep disposal scenarios are shown in Figure 9-6. The figure again shows that applying the Eco-Indicator 99 weighting factors changes the ordering of the environmental impact categories and Respiratory Inorganic impacts dominate rather than Fossil Fuel impacts. The Minerals, Climate Change and Eco-toxicity are about equal and the Carcinogen impacts are only slightly smaller. All the other impact categories are negligible.

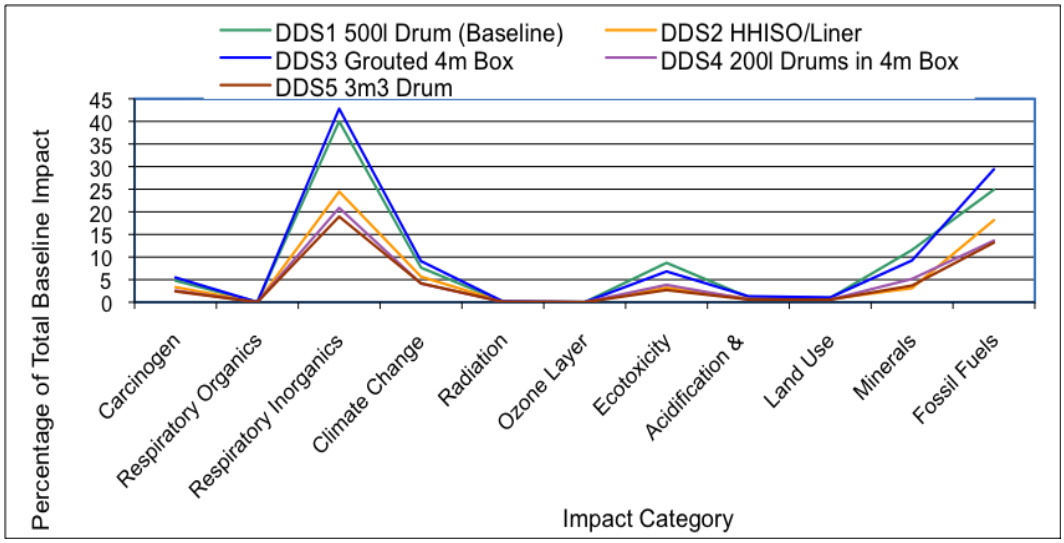

Figure 9-6 Comparison of Eco-indicator 99 (Hierarchist/Average) weighted impacts for the five HAW deep disposal scenarios for DNLEU. Results are in percentage of total baseline impact (TBI).

Overall, there is no major difference between the DDS1 500l Drum (Baseline) scenario and the DDS3 (Grouted 4m Box) scenario impact category results. Their potential environmental impacts are clearly grouped together and significantly higher than those for the other disposal scenarios.

The DDS2 (HHISO/Liner) 40% reduction in total impact compared to the DDS1 500l Drum (Baseline) scenario, discussed previously, is clearly reflected in the dominant environmental impact categories results shown in Figure 9-6. This constitutes an interesting alternative posed by Hickford et al 2012 since HHISOs are not approved HAW disposal packages for a future GDF. HAW disposal packages are generally made of stainless steel as it affords better corrosion resistance than carbon steel. However, the titanium liner compensates for this and the carbon steel package option represents a plausible alternative for DNLEU disposal based on these results. It also shows the potential benefits of challenging the norm and opens the way to considering other disposal packages for DNLEU.

There is only a marginal difference between the DDS4 (200l Drums in 4m Box) and the DDS5 ($3m^3$ Drum) disposal options. Both scenarios give about a 50% reduction in impact compared to the DDS1baseline scenario. However, the best overall environmental option of the five deep disposal scenarios modelled is the DDS5 in-drum grouting and disposal in $3m^3$ drums.

The above results were for the long-term interim storage of all DNLEU as U_3O_8 powder, the repackaging and disposal of the DNLEU in a range of packages suitable for a future GDF. Attention now turns to the possible disposal of the various forms of DNLEU, in current and planned long-term interim storage packages, but over-packed in suitable containers rather than repackaged in new containers.

Given the very low activity of DNLEU, and the extremely long half-life of the main nuclides (up to 4.5 billion years for U238), the difference in release from carbon steel or stainless steel containers over their corrosion periods is moot. This suggests that

container and disposal options could be usefully explored further for a future GDF or a potential near surface disposal facility.

9.3.2 Over-packed Disposal to a GDF or Near Surface Disposal Facility

This section presents the results of the three scenarios (DDS6, SDS1 and SDS2) for the disposal of DNLEU in overpacks to a future GDF and a potential near surface disposal facility some tens of metres below the surface (NDA 2013b).

DDS6 - Overpacked DNLEU as HAW

This scenario assumed that the stored DNLEU was disposed as HAW, hence the DDS6 life cycle impacts are superimposed on the results of the previous five deep disposal scenarios and are presented in Figure 9-7.

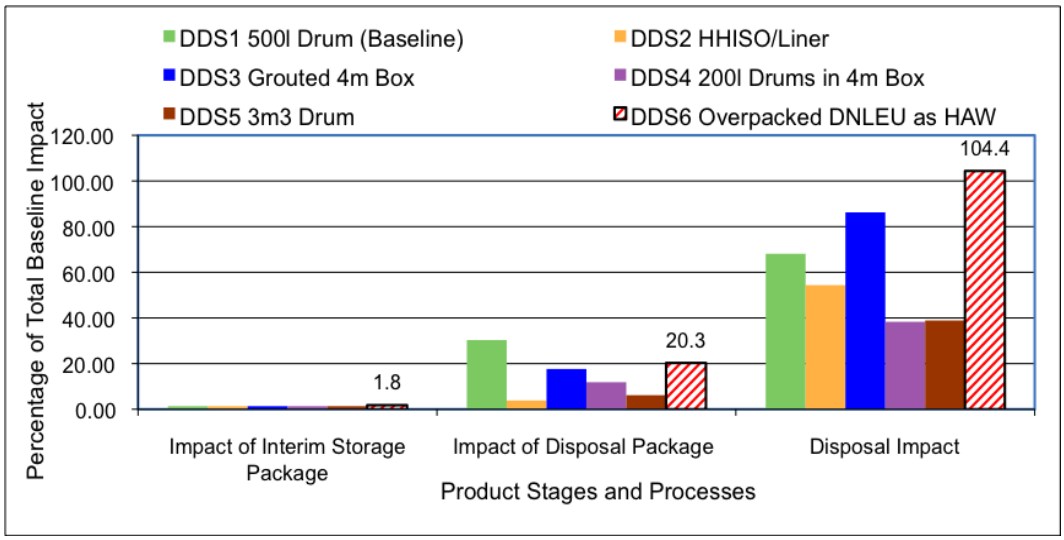

Figure 9-7 Comparison of deep disposal of over-packed DNLEU consigned to a GDF with repackaged DNLEU in the previous five deep disposal scenarios as a percentage of total baseline impact (TBI) (from Eco-Indicator 99 (Hierarchist/Average)).

The results in Figure 9-7 show that the over-packed disposal impact as HAW to a GDF is higher than the previous five deep disposal scenarios. This is due to the poor utilisation of the 4m boxes and the special FHISOs.

The over-packed disposal impact is about 105% TBI (i.e. ~26% higher than the DDS1 500l Drum (Baseline). The over-packed impact is subdivided as follows:

~62% for Urenco UK Ltd U_3O_8 in DV70 boxes,

~16 % for Capenhurst Nuclear Service U_3O_8in DV70 boxes,

~18% Magnox depleted uranium in un-compacted 200 litre drums,

~8% THORP product uranium in un-compacted 50 litre kegs, and

~1% miscellaneous UO_3 in un-compacted 200 litre drums.

The DDS6 packaging impact is about 22% TBI (i.e. ~10% lower than the DDS1 500l Drum (Baseline) disposal package impact) and dominated by the 4m boxes as overpacks (~20%). The long-term interim storage package impact is small at ~2% TBI. The increased impact is also evident in the weighted environmental results presented in Figure 9-8. The environmental impact category trend is the same as the five deep disposal scenarios discussed in Section 9.3.1 and the order of significance of the impact categories remains unchanged.

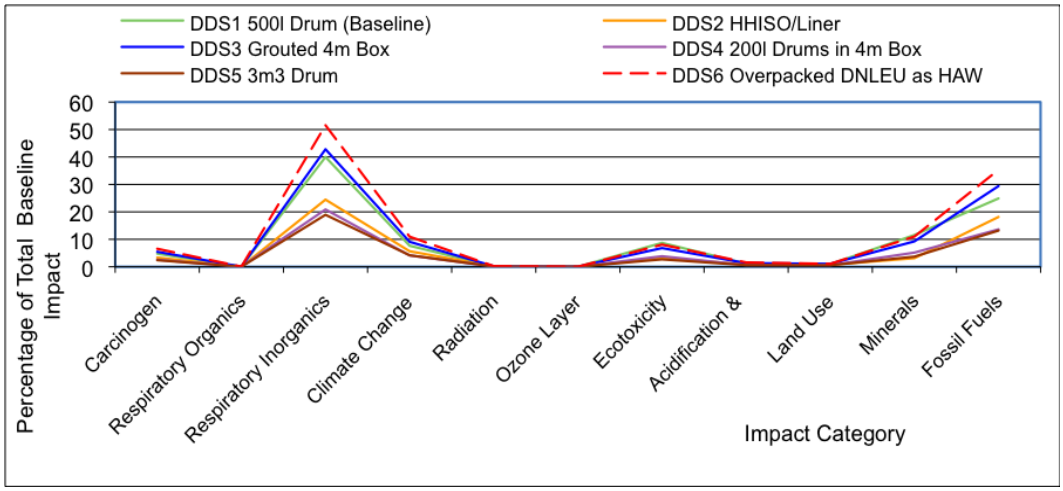

Figure 9-8 Comparison of the weighted impact results for over-packed DNLEU consigned to a GDF with repackaged DNLEU in the previous five deep disposal scenarios as a percentage of total baseline impact (TBI) (from Eco-Indicator 99 (Hierarchist/Average)).

SDS1 – Overpacked DNLEU as LLW (SS)

This scenario assumed the same DNLEU stock, long-term interim storage packages and overpacked in stainless steel (SS) 4m boxes and special FHISOs as the DDS6 scenario. The only change was the packages were consigned to a potential near surface disposal facility with disposal impacts broadly equivalent to LLW disposal rather than HAW disposal.

SDS2 – Overpacked DNLEU as LLW (CS)

This scenario assumed the DNLEU stock and long-term interim storage packages were as DDS6 and SDS1, but carbon steel (CS) rather than stainless steel (SS) 4m

box over-packs were used for disposal. The disposal volume remained unchanged from SDS1 and the waste was consigned as LLW to a potential near surface disposal facility.

Comparison of General Over-pack Results

A comparison of the results of the three over-packed disposal scenarios is presented in Figure 9-9. The figure shows that the total cycle impact of the over-packed DNLEU, disposed as HAW to a GDF, was about ~126% TBI. Disposal of the same over-packs as LLW to a near surface disposal facility reduces the total impact to ~32% TBI. The total impact is further reduced to ~16% TBI if carbon steel 4m box over-packs can be used instead of stainless steel boxes.

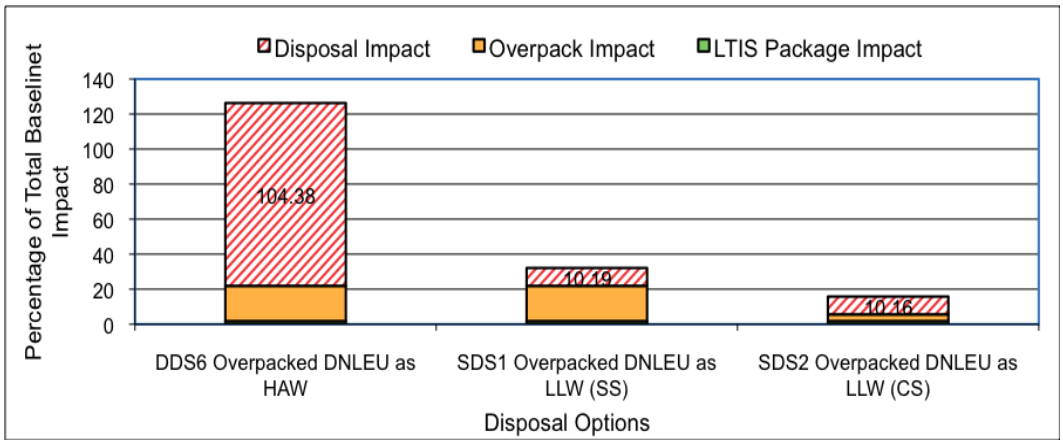

Figure 9-9 Comparison of over-packed DNLEU disposed as HAW to a future GDF or as LLW in stainless steel (SS) or carbon steel (CS) 4m box over-packs to a potential near surface disposal facility (from Eco-Indicator 99 (Hierarchist/Average)). The results are in percentage total baseline impact (TBI).

Since the package volume is the same for the HAW disposal and the initial LLW disposal (DDS6 and SDS1) their package impacts are constant but the disposal impacts are significantly different. When carbon steel 4m boxes are used for LLW disposal in SDS2 the disposal impact remains constant but the packaging impact is significantly reduced because of the change of steel. Hence, the overall life cycle impact for LLW disposal is essentially halved. This is an over simplification and the carbon steel 4m boxes may have to be thickened to be able to take the DV70 boxes. However, it was assumed that the external dimensions and hence volume remained constant. The graph attempts to show the potential reduction in environmental impact

if the DNLEU could be disposed as LLW to a potential near surface disposal facility rather than HAW for deep disposal in a future GDF.

Container Implications

DNLEU shows ingrowth of radioactivity with time rather than the usual decay and the uranium isotopes and their daughter products have very long half-lives, in the span of about 250,000 years for U-234 to 4.5E9 years for U-238. The difference in stainless steel and carbon steel corrosion rates is unlikely to make a significant difference to uranium radionuclide release from disposal packages over such a long timescale. Hence, Figure 9-9 shows the benefit of changing the disposal container material to carbon steel if a suitable safety case can be made for uranium release rates to the environment from a near surface disposal facility.

Comparison of Near Surface Disposal Facility Product Stage/Process Results

Figure 9-10 presents a comparison of the overall life cycle impacts in more detail, based in the different waste producers/owners.

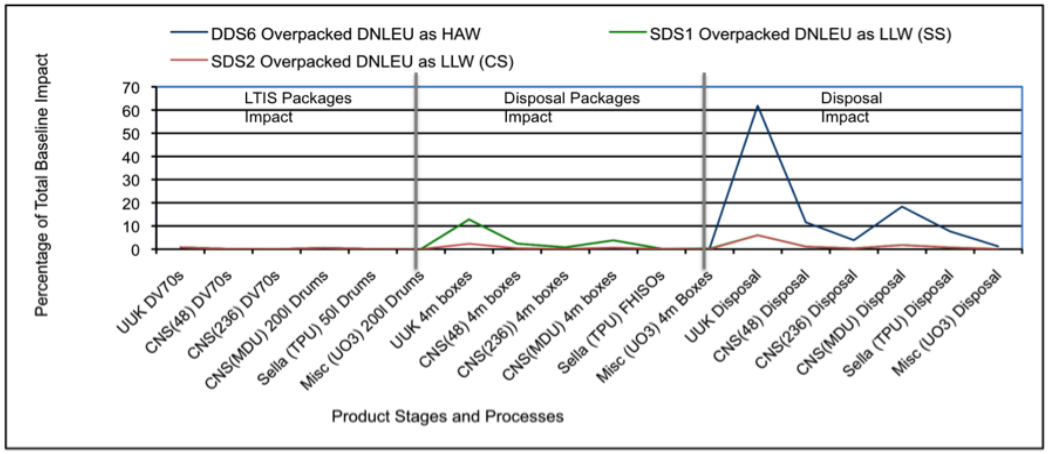

Figure 9-10 Total life cycle impacts (from Eco-Indicator 99 (Hierarchist/Average)) for the three over-packed DNLEU disposal scenarios. Where UUK is Urenco UK Ltd, CNS is Capenhurst Nuclear Services, (48) and (236) represent Type 48 cylinder and Type 0236 cylinder UF_6 deconverted to U_3O_8 respectively, MDU is Magnox depleted uranium, TPU is THORP product uranium from Sellafield (Sella) and Misc is the miscellaneous UO_3 from various sites. LTIS is long-term interim storage. The results are in percentage of total baseline impact (TBI).

Figure 9-10 shows that the disposal impacts dominate the life cycle for stainless steel over-packs disposed as HAW. The disposal impact dominates the DDS6 scenario, the long term interim storage package and disposal package impacts are minor. This is due to the high disposal impact for HAW for a future GDF. In the SDS1 scenario

the disposal package impacts are slightly higher than the disposal impacts. This is because in this is scenario the waste is disposed as LLW to a potential near surface facility with about an order of magnitude lower disposal impact than HAW. In this case the stainless steel drum and boxes represent a significant potential environmental impact in their own right. In the SDS2 scenario, the disposal package impact is significantly reduced since carbon steel is assumed for the 4m boxes rather than stainless steel. Hence, a large reduction in potential environmental impact can be achieved if the DNLEU is disposed to a potential near surface disposal facility. Further environmental savings can be achieved in the 4m box over-packs can be made of carbon steel.

9.4 DNLEU Disposal Cost Estimates

Having generated the potential environmental impacts for the DNLEU disposal scenarios attention now turns to estimating the financial costs of each option. The estimates are based on the cost of the long-term interim storage packages, the disposal packages and the transport and disposal of the waste packages to a potential GDF or near surface disposal facility. The section discusses the scarcity of HAW costs data for containers, transport and disposal. It also provides estimates of disposal costs for the shallow and deep disposal scenarios.

9.4.1 Scarcity of HAW Cost Data

As stated previously VLLW and LLW disposal container, transport and disposal costs are readily available. Equivalent data for HAW packages and disposal to a GDF are commercially sensitive and not readily available hence estimates are made from costs data in the literature.

Storage Container Costs

The cost of a carbon steel DV70 box was unknown, but two estimates were made. The first estimate was based on their mass (0.75te from Jones 2014) compared to a carbon steel HHISO from the LLW Repository Ltd disposal service price list (LLWR Ltd 2014). This gave a cost estimate of £1,950 per DV70. The second estimate assumed carbon steel to be about ¼ the cost of stainless steel, hence extrapolating from a stainless steel 3m^3 box gave a DV70 cost of £2,270.

The cost of a 200litre stainless steel drum was also unknown. The manufacturer of drums for Sellafield was approached but could not supply the costs for contractual reasons. A minimum cost of ~£120/drum was taken from a commercial internet site and an upper value of ~£250/drum estimated by scaling the cost of the stainless steel $3m^3$ box by the 200 litre drum mass. The cost of super-compacting the 200 litre drums for the fourth GDF scenario was taken from the LLW Repository service price list (LLWR Ltd 2014). The cost of a 50 litre stainless steel kegs was also unknown. Far fewer of these will be made than the 200 litre drums hence their production costs may be higher. They were assumed to be half the cost of a 200 litre drum.

The cost of the 500litre stainless steel drums for the baseline GDF scenario was estimated as £1500 (Bryan 2005). Scaling the 200litre drum by mass gave a 500litre drum cost of ~£1,570, and scaling the $3m^3$ box cost by 500litre drum mass gave ~£725. The cost of the stainless steel stillage frame for four 500litre drums was also unknown. However, the mass was stated to be 1tonne (NDA 2008) and a stillage cost of £1,490 to £1,850 was estimated from current stainless steel price. The costs include a 25% increase to represent the production cost of a stillage. This is a crude estimate but gives an indicative stillage cost range.

The current cost of a HHISO is £8,000 (LLWR Ltd 2014). The cost of the HHISO/titanium liners, for the second GDF scenario, was based on the number of HHISOs and the mass of titanium for the liners from Hickford et al 2012 (Scenario 16H). A titanium price of £3,390 to £3,890/tonne for 2013/2014 was taken from the internet to estimate the liner costs.

The cost of a stainless steel box was estimated as between £16,000 to £32,000 in Chapter 8. The cost of a carbon steel 4m box, for use at a potential near surface disposal facility, was estimated as ~£11,000 based on the mass ratio of HHISO and a 4m box.

The cost of a stainless steel $3m^3$ drum was assumed to be the same as a $3m^3$ box. The early development $3m^3$ box cost from Ove Arup 1997 was inflated to current prices giving a cost of ~£9,000 per box. However, CoRWM stated that the cost for each $3m^3$ box could be as high as £25,000 (CoRWM 2008b). It was also noted from NDA 2012a that the current production cost of a $3m^3$drum was £20,000. Hence the cost range estimated for a $3m^3$ drum for the fifth GDF scenario addressed the current high unit production cost but allowed for cost reductions from improved production methods and economies of scale from increased demand.

Transport Costs

A HAW transport cost of £1,250/m^3 was assumed for ILW metal disposal in Chapter 8. This was based on estimates for core graphite disposal to a GDF in NDA 2012a which showed that transport represented about 10% of the total HAW disposal cost. However, DNLEU is a low activity material and would require minimum shielding for contact handling and transport. The current joint waste management plans (e.g. Shipton and Falconer 2013) indicate that transport represents about 2% of the total LLW disposal cost. Hence DNLEU transport costs were assumed to be £250/m^3, i.e. 1/5 of the ILW transport cost.

Disposal Costs

As stated previously the LLW disposal volume cost norm is currently £2,990/m^3 (LLWR Ltd 2014). There is no cost data for a potential near surface disposal facility for DNLEU hence the LLW cost norm was assumed. VLLW disposal cost estimates range from £300 to £700m^3 with a planning norm of £500/m^3 (LLWR Ltd 2014) and this was assumed for the disposal of redundant DV70 boxes in the GDF scenarios.

The shielded packaged disposal cost of £3840/m^3 was assumed for the 4m boxes and HHISO/liner HAW disposal scenarios and the unshielded package cost of £10,260/m^3 was assumed for the 500 litre drum and $3m^3$ drum scenarios based on NDA 2012a. The baseline cost of £8,380/m^3 for mixed shielded and unshielded packages was also calculated for each HAW scenario. These estimates allow for the different handling regimes for disposal a GDF.

9.4.2 DNLEU Disposal Cost Estimates

General Cost Comparison

The cost estimates for the deep disposal scenarios to a future GDF and shallow disposal a potential near surface disposal facility scenarios are summarised in Table 9-5 and Table 9-6 respectively. The DDS4 (200l Drums in 4m Box) disposal package costs are relatively high in Table 9-5 because of the large number of 200 litre drums and the additional super-compaction costs at Sellafield. This suggests some consideration could be given to building a dedicated super-compaction facility at Capenhurst as the site contains the bulk of the DNLEU. The DDS5 ($3m^3$ Drum) disposal package costs are relatively high in Table 9-5 because of the number of $3m^3$ drums and their high unit costs from the literature.

Table 9-5 Comparison of financial costs of deep disposal of DNLEU as HAW to a Future GDF

Scenario	**LTIS Package Cost (£million)**	**Disposal Package Cost (£million)**	**Trans-port Cost (£million)**	**DV70 VLLW Disposal Cost (£million)**	**GDF Disposal Volume Cost (£million)**	**Total GDF Disposal Volume Cost (£million)**
DDS1 500l drum (Baseline)	45 -61	162 - 300	34	5 – 16	1124 - 1376	1370 – 1787
DDS2 (HHISO/ Liner)	45 - 61	60 - 63	26	5 – 16	404 - 882	540 – 1048
DDS3 (Grouted 4m Boxes)	45 - 61	125 - 250	42	5 – 16	645 - 1407	862 – 1776
DDS4 (200l Drums in 4m Boxes)	45 - 61	237- 343	19	5 – 16	289 - 630	595 – 1069
DDS5 ($3m^3$ Drums)	45 - 61	258 - 711	19	5 - 16	621 - 761	948 – 1568

Red – highest cost, Yellow – medium costs, Blue- low cost, Green – lowest cost

The tables show that there is a large variation in the total disposal costs due to the variation in waste package number and disposal volume. The GDF disposal costs dominate the total costs in each case but package costs are also significant. The DV70 long-term interim storage package costs are constant at about 3-6% of the total costs. The disposal package transport costs vary between 1-6% of the total cost. This is due to the variation in number of packages and their volume. The VLLW disposal

of the redundant DV70s is small at ~1% of the total costs for GDF and zero for the near surface disposal facility as they are over-packed in 4m boxes for disposal. The long-term interim storage packaged cost in Table 9-5 is marginally higher than in Table 9-6. This is because the Magnox depleted uranium, THORP product uranium and miscellaneous UO_3 were converted to U_3O_8 and packaged in additional DV70 boxes for GDF disposal. The transport costs are higher in Table 9-6 because of the poor utility of the 4m box over-packs. The SDS1 and SDS2 LLW disposal costs are relatively high at ~£600m because of the poor utility of the 4m box over-packs. However, using carbon steel over-packs for the SDS2 LLW disposal in Table 9-6 saves £44 - £185m in container costs. Further, the total cost for shallow disposal of DNLEU as LLW is between 20% and 57% cheaper than the total cost for deep disposal as HAW.

Table 9-6 Comparison of financial costs of the DNLEU in over-packs for deep disposal to a future GDF or shallow disposal to a potential future near surface disposal facility

Scenario	**LTIS Package Cost (£million)**	**Disposal Package Cost (£million)**	**Trans-port Cost (£million)**	**Disposal Volume at GDF Costs (£million)**	**Total Disposal at GDF Costs (£million)**	**LLW Disposal Cost (£million)**	**Total LLW Disposal Cost (£million)**
DDS6 (Over-packed DNLEU as HAW)	37 - 51	146 - 287	51	785 - 1712	1019 - 2101	N/A	N/A
SDS1 (Over-packed DNLEU as LLW (SS))	37 - 51	146 - 287	51	N/A	Not applicable as LLW disposal costs assumed	611	845 – 1000
SDS2 (Over-packed DNLEU as LLW (CS))	37 - 51	102	51	N/A	Not applicable as LLW disposal costs assumed	611	801 – 815

Red – highest cost, Yellow – medium costs, N/A – Not Applicable

A comparison of the average costs from Table 9-5 and Table 9-6 are shown in Figure 9-11. Figure 9-11 shows three broad groups of total average disposal costs. The highest disposal costs are for the DDS1 (500l Drum (Baseline)) and DDS6 (Overpacked DNLEU as HAW) scenarios for deep disposal to a future GDF. The middle cost group includes the DDS3 (Grouted 4m Box) and DDS5 ($3m^3$ Drum) as

HAW to a future GDF. These give a 16-20% reduction compared to the DDS1 (500l Drum (Baseline)) total cost. The lowest cost group includes the DDS2 (HHISO/Liner) and DDS4 (200l Drums in 4m Box) disposal as HAW to a future GDF. It also includes disposal of the SDS1 (Overpacked DNLEU as LLW (SS)) and SDS2 (Overpacked DNLEU as LLW (CS)) shallow disposal to a potential near surface disposal facility. The disposal of the over-packed DNLEU in stainless steel 4m boxes as LLW is marginally higher than the other three costs in the group due to the poor utility and cost of the stainless steel 4m boxes. This group shows a 40 – 50% reduction compared to the 500l Drum (Baseline).

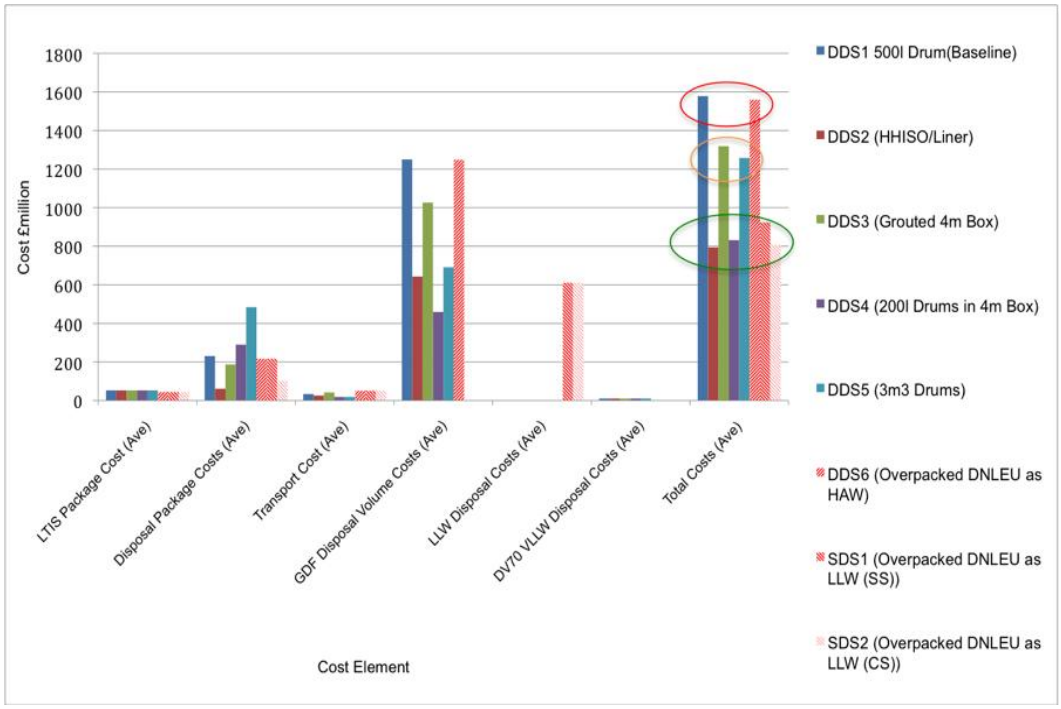

Figure 9-11 Average storage costs and disposal costs for DNLEU. Disposal costs are based on the NDA 2012a £/m^3 costs for HAW and LLW Repository Ltd £/m^3 for LLW and VLLW.

Overall the DDS2 (HHISO/Liner) scenario offers the lowest total GDF disposal cost followed closely by the DDS4 (200l Drums in 4m Box). Although both scenario options were taken from the literature, further research is needed to confirm that these waste-forms and packages would be acceptable for disposal at a future GDF or a potential near surface disposal facility.

The choice of container, their utilisation and costs will be significant issues for DNLEU disposal. Although it is low activity material, and should not be subject to

the high costs associated with ILW transport to a future GDF, the transport costs for DNLEU could still be substantially higher than the 2% for LLW. This is also an area that needs further research.

Near Surface Disposal Facility Limiting Costs

The HAW disposal costs in Table 9-6, estimated using GDF costs, give a potential upper limit to the variable cost for disposal to a near surface disposal facility assuming over-packed disposal. The LLW disposal costs in Table 9-6 give a potential lower limit to the variable disposal costs for over-packed disposal. The fixed cost of designing, planning, authorising, constructing, operating and backfilling a near surface disposal facility have not yet been estimated. These could be significant even if the vaults are only a few tens of metres below the surface (NDA 2013b). The costs should be considered as indicative only as no comparable near surface disposal facility for DNLEU yet exists. However, the disposal of 175,000m^3 of LLW and VLLW at the new LAW disposal facility being built at Dounreay can give an insight to the potential costs of a near surface disposal facility for DNLEU. The Dounreay facility is six concrete vaults at about 11m below the surface and has an estimated lifetime costs of £110m (McVay 2014). However, no breakdown of this cost has been found. Hence it could be expected that any near surface disposal facility several tens of metres below the surface may be considerably more expensive than the Dounreay facility.

Cost Comparison with NDA Macro-economic study

The costs for a range of scenarios for the disposal of uranium and plutonium are presented in NDA 2007. This macro-economic study showed disposal costs for uranium ranging from £20m to £26m for packaging in bespoke containers or 2m boxes - £86m using 200lite drums in dedicated stillages - between £110m to £140m using 500litre drums or 3m^3 boxes and over-packing existing DNLEU packages in 2m and 4m boxes. The highest costs were between £270m for direct disposal and over-packing of uranium in standard 500litre drums and stillages and £310m for over-packing existing DNLEU packages in 3m^3 boxes. Hence macro-economic study also shows a large variation in disposal costs for scenarios it considered. The

costs for the macro-economic study were based on a DNLEU inventory of 50,000 (tU) (NDA 2007) and historical disposal costs, compared to 162,000 (tU) and current published costs used in this research. The increased inventory and current container, transport and disposal costs result in the higher total costs presented in this research. The macro-economic study does, however, show the benefits of developing bespoke disposal packages and stillage or repackaging DNLEU in 2m boxes giving a high packing density. These options were not explored in the research. Optimising the disposal option costs is therefore key to in the macro-economic study and this research.

9.4.3 Costs excluded from the estimates

The costs of deconversion of UF_6 or conversion of UO_3 to U_3O_8 and the long-term interim store construction, operation and demolition are excluded from the estimates. The cost of DNLEU production and storage relate to the material as a potentially commercial product. Once DNLEU is designated a waste these costs cease to be relevant. Urenco UK Ltd estimate €400m - €500m for the construction of the Tails Management Facility but the cost associated with the uranium oxide store is unknown. It is assumed to be small if similar to the equivalent steel and concrete COVRA VOG store in the Netherlands, therefore currently excluded. The transport of empty storage and disposal containers to sites are expected to be small compared to the other costs, hence they are also excluded. The labour costs for repackaging of reactor core graphite in NDA 2012a were highly uncertain and excluded from that analysis. The same was assumed for DNLEU disposal hence repackaging labour costs also excluded.

9.5 Analysis Review

The results show that:

- The estimated disposal volumes for DDS4 (200l Drums in 4m Box) and DDS5 ($3m^3$ Drum) are similar to the estimated DNLEU long-term interim storage volume, suggesting good package utilisation for these disposal options.

- The estimated disposal volumes for DDS1 (500l Drum (Baseline)), DDS2 (HHISO/Liner) and DDS3 (Grouted 4m Box) are about a factor of 1.5 to 2.5 higher than the DNLEU storage volume, suggesting poor package utilisation.
- The estimated over-packed disposal volume was about a factor of ~2.8 higher than the DNLEU long-term interim storage package volume. This suggests the initial proposed over-pack utilisation was not optimised, hence some potential improvement are discussed in Chapter 10.

9.6 Chapter Summary

The results demonstrate the storage and disposal life cycle potential environmental impacts and financial costs are highly variable and are dominated by the disposal volume impacts for HAW to a future GDF or potential near surface disposal facility. The environmental impact and financial costs estimate results are summarised in Table 9-7.

The cost estimates are crude and have a high degree of uncertainty. They do not include the cost of the long-term storage facilities at Capenhurst and Sellafield or the repackaging costs for disposal. Nor do they include the conversion costs for UF_6 to U_3O_8, or UO_3 to U_3O_8 or the potential ongoing DNLEU production from the Urenco UK Ltd enrichment business. This latter point could be significant given the company throughput and extended order book. It is anticipated that improved package, transport and disposal costs will result from the Integrated Project Team on Uranium and discussions with packagers (NDA 2013b and 2013a respectively). The estimates presented here should therefore be treated with caution and regarded as broad outline estimates for the current inventory only.

The results suggest that significant potential environmental improvements and financial cost savings can therefore be achieved by choosing alternative disposal packages to the current 500l Drum (Baseline). Disposal of DNLEU as LLW to a near surface disposal facility reduced the disposal costs to an average of ~£865m. This cost is high due to the poor utilisation of the 4m box over-packs and may reduce if mixed disposal could be justified.

Table 9-7 Summary of the potential environmental impacts and costs of the long-term storage and disposal of depleted, natural and low enriched uranium

Scenario	Environmental Impacts	Estimated Total Average Cost
Deep Disposal Scenario 1 (Baseline)	◉	●
Deep Disposal Scenario 2 (HHISO/Liner)	○	⊙
Deep Disposal Scenario 3 (Grouted 4m boxes)	◉	○
Deep Disposal Scenario 4 (200litre drums in 4 m boxes)	◉	⊙
Deep Disposal Scenario 5 ($3m^3$ drums)	◉	○
Deep Disposal Scenario 6 (Overpacked DNLEU as HAW)	●	●
Shallow Disposal Scenario 1 (Overpacked DNLEU as LLW)	⊙	◉
Shallow Disposal Scenario 1 (Overpacked DNLEU as LLW in carbon steel 4m boxes)	⊙	⊙

Key: Lowest ⊙, Low ◉, Medium ○, High ◉, Highest ●, costs based on those presented in Figure 9-11. The lowest potential environmental impacts for repackaged DNLEU disposal are the DDS5 ($3m^3$ Drum), DDS4 (200l Drums in 4m Box) and DDS2 (HHISO/Liner) scenarios. The highest potential environmental impacts arise from the DDS6 (Overpacked DNLEU as HAW) and DDS3 (Grouted 4m Box) (DDS3) and DDS1 (500l Drum (Baseline)) to a future GDF. The lowest environmental impacts are the two shallow disposal scenarios SDS1 and SDS2 (Overpacked DNLEU as LLW (SS) and LLW (CS) respectively) to a potential near surface disposal facility. Long-term interim storage package impact and their disposal impact constitute only 1.4% to 1.8 % of the baseline to potential environmental impact. Hence the choice of DV70 boxes for long-term storage packages is not a significant environmental issue in the DNLEU storage and disposal life cycle.
The DDS1 (500l Drum (Baseline)) and DDS6 (Overpacked DNLEU as HAW) give the highest total average costs. The DDS2 (HHISO/Liner) and DDS4 (200l Drums in 4m Box) scenarios gave the lowest total average cost at ~50% of the baseline costs.

10 Conclusions, Discussion and Recommendations

The research aim of the research was how best to maximise the environmental benefits, or minimise the environmental detriments, of radioactive waste management decision-making. This chapter presents the conclusions of the research, discusses some of the key findings and limitations then makes recommendations for industry and for further research.

10.1 Conclusions

The research aim and objectives of Section 1.5 were met. Simplified partial life cycle assessment (LCA) models and financial cost estimates were developed for end-of-life disposition options for significant UK radioactive wastes. The LCA models were to investigate the potential environmental impacts of the disposal and treatment for recycling or re-use of two radioactive metal inventories; the Windscale Advanced Gas-cooled Reactor (WAGR) boilers and the entire UK radioactive metals inventory. Having developed the methodology the research was extended to investigate hypothetical scenarios for a potentially significant future radioactive waste management issue: namely, the long-term interim storage and disposal of the UK stock of civil depleted, natural and low enriched uranium (DNLEU) at 2015. Cost estimates were generated for each scenario using data from the literature to identify the cost benefits or detriments of each option. The conclusions of the radioactive metals research are summarised in Box 10-1.

Box 10-1 Conclusions for UK radioactive metals

- Both the radioactive metals studies showed there are potential significant environmental advantages in radioactive metals recycling promoted by the current solid LLW policy, strategy 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
- Overall the potential environmental impacts are dominated by ILW metals disposal but costs are dominated by LLW metals disposal costs.
- ILW, LLW and VLLW metals potential 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 improvements

The conclusions of the DNLEU research are summarised in Box 10-2.

Box 10-2 Conclusions for the 2015 UK civil depleted, natural and low enriched uranium inventory

- Storage and disposal potential environmental impacts and costs are highly variable and are dominated by the disposal volume impacts
- Disposal package potential environmental impacts are significant, but the long-term interim storage package impacts are negligible
- Disposal packages with a high utilisation factor minimises the disposal volume and potential environmental impacts
- The $3m^3$ drums and super-compacted 200litre drums in 4m boxes give the lowest potential environmental impact, closely followed by disposal in half height ISO containers with titanium liners
- The 500litre drum baseline and grouted 4m boxes give the highest potential environmental impact
- HHISOs with titanium liners give the lowest disposal cost, followed by super-compacted 200litre drums in 4m boxes
- The 500litre drum baseline and 4m box overpacks gives the highest disposal costs
- The potential environmental impacts and cost for disposal to a proposed near surface disposal facility are highly speculative. Poor utilisation factors mean a large number of stainless steel overpacks are needed, giving high transport and disposal costs, but avoids repackaging requirements.

10.2 Discussion of main issues

10.2.1 Eco-Indicator 99 as the life cycle impact assessment method

Eco-indicator 99 focuses on European environmental impacts to model potential damages to Human Health, Ecosystem Quality and Resources and is compatible with international standard for LCA (ISO 14040 2006). It is a rather old life cycle impact assessment (LCIA) method, but well respected as shown in Chapter 6. The limitations of Eco-indicator 99 are discussed in the method manual (Goedkoop and Spriensma 2001a), the International Reference Life Cycle Data System document (EC/JRC 2010a and 2011, Wolf et al. 2012) and the EC clean materials technology assessment (Phylipsen et al 2002), which used Eco-Indicator 99 as the LCIA method. A more modern LCIA method, ReCiPe 2008, was developed based on Eco-indicator 99 and CML 2001 as discussed in Chapter 6. However, the total life cycle impacts for the WAGR boilers scenarios calculated by both Eco-indicator 99 and ReCiPe in Chapter 7 were not greatly different, giving confidence in Eco-indicator 99. It has to be agreed that companies producing high environmental impacts materials such as iron and steel or concrete have been making considerable effort to reduce their impacts and this may not be fully addressed in the Ecoinvent 2.2 database and Eco-indicator 99 method. However, it was not the intent of this research to comparing

the potential environmental impacts of multiple LCIA methods. Hence, this is left as an area for further research.

10.2.2 Comparison with the Nuclear Fuel Cycle LCAs

The above limitation of Eco-indicator 99 means that the results do not allow direct comparison with the nuclear fuel cycle LCAs in Chapter 4 and general benefits for metal recycling from Chapter 5. Rather, they show the difference in potential environmental impacts for the scenarios in each phase of the research.

The overall results do, however, appear to support the claim of significant benefit from metals recycling by Wallbridge et al. 2012a, i.e. 15 to 55% reduction in impact for recycling 70% steel with 90% recycling rate. This compares to a 53% reduction in total life cycle impacts for international treatment of the UK radioactive metals inventory without claiming the benefit of avoided metal in recycling as discussed in Section 8.3.1. The outcome is a little different for this research as it assumed a 95% recycling rate and almost the same percentage of metals available for treatment, i.e. 5% of VLLW and 5% of LLW metals were not suitable for decontamination or melting, so were directly disposed, and ~14% of LLW metals had surface oxide films containing fission products or activation products, hence could not be recycled. However, this research included melting these later metals and ILW metals decayed or decontaminated to LLW for volume reduction only, which was not considered by Wallbridge and his colleagues.

This research also supports the claim by Wallbridge et al. 2012a and others that steels and concrete are important environmental issues during the waste and decommissioning phase of the nuclear fuel cycle as discussed in Chapter 4. This research did not consider the wider aspects of decommissioning a power station in general, nor the impacts associated with the on-site interim storage of wastes that Wallbridge et al. 2012a included in their analysis. However, the potential environmental impacts of the long-term storage and disposal of the depleted uranium in this research represents is an aspect of waste management and decommissioning of a non-power station that is not addressed in Wallbridge et al. 2012a and others.

10.2.3 Radioactive metals issues

Radioactive Metal Melting Process Applications

The radioactive metals melting process is ideal for complex metal structures where general decontamination processes, surface radiation monitoring technique (or limits) are problematic or time consuming. Activated metals and ILW metals decayed or decontaminated to LLW levels can also be melted for waste volume reduction only. However, these metals may have higher radioactive content and present issues for operator radiation doses during size reduction, decontamination and melting. Also, care must be taken not to concentrate activity in melt ingots or secondary wastes that could result in it being categorised as ILW, which is then an issue for the return transport to the UK.

To Melt or Not to Melt?

Allwood and his colleagues rightly point out "Our pre-occupation with re-melting metals in order to reduce emissions is misplaced" but they recognise that "...recycling offers vast and important savings compared to primary production but we can do more – reuse, where possible is better" (Allwood et al 2010b). There are limited opportunities to re-use radioactively contaminated metals, but it can be done. For example, steel used fuel skips were decontaminated at Chapelcross for re-use by Sellafield (NDA 2014b). This saved an estimated £1m pounds in new fuel skip manufacture (i.e. ~£14,000 each) plus the virgin materials and production process environmental impacts. Another example is contaminated lead shielding blocks, these can be surface decontaminated for direct re-use as shielding.

Radioactive iron and steel treated for recycling to open markets has to be re-melted in commercial foundries. Similarly, radioactive iron and steel used for ductile casts iron containers has to be re-melted with conventional scrap to produce the containers. However, shielding blocks and cylinders, and steel shot for high-density cement shielding, can be produced from the induction melt with the minimum amount to reheating rather than re-melting. Therefore there is a range of options for these treated metals.

Quantities of radioactive aluminium, copper, nickel and zinc are much lower. These are not commonly re-used in the nuclear industry as they are not commercially viable (e.g. Europeam Commission (EC) 1998), hence, they are general sold directly to respective metals markets after treatment.

Benefits and Detriments to Melting

Melting radioactive metals may not be as environmentally advantageous as re-use but it does offer:

- Significant waste volume reduction for LLW surface contaminated metals, typically about 95% depending on the metals as shown in Chapter 8 and a potentially significant volume reduction for activated LLW metals (but requires further research)
- Further decontamination opportunities by partitioning radionuclides like Uranium (U), Plutonium (Pu), Americium (Am) and Europium (Eu) isotopes in the melt slag, volatile nuclides like Caesium (Cs 134 and Cs137) can be retained in the slag or captured in dust in filters and retaining nuclides such as Cobalt (Co60), Nickel (Ni63) and Iron (Fe55) in the melt ingot as shown in
- Figure 5-6
- Homogenisation of the residual radionuclides in the melt ingot allowing good radiological characterisation
- Reduction in cost and increased radiological efficiencies for treating large and complex nuclear plant items like boilers, steam generators and pressurisers (Walberg et al. 2008, Nitsche and Fasten 2010, World Nuclear News (WNN) 2008 and 2010).
- Minimisation of “single use” waste containers for LLW and HAW reducing the environmental impacts caused by material production, processing and transport
- Reduction in the cement grout needed to produce a stable waste-form or as back fill at disposal facilities
- Preservation of declining resources by saving on virgin material consumption and the energy used in metal production.

There are of course drawbacks to melting such as the need for additional equipment, implementation of additional processes to minimise dose to plant operators, and the requirement to transport radioactive metals and metallic components from the UK to international facilities in France, Germany, Sweden or the USA. The transport issues require compliance with international transport regulation, robust safety cases and special road transporters and ships. Transport can be contentious. An example is the cessation of a contract to transport 32 steam generators from a Bruce Power facility in Canada to the Studsvik Nykoping facility in Sweden. The Canadian and international transportation requirements were met (WNN 2009, Canadian Nuclear Safety Commission (CNSC) 2010, 2011 and 2012, plus Nuclear Free Local Authorities (NFLA) 2011) and preparations made by Bruce Power and Studsvik but the process is in abeyance through objections raised in the public consultation.

Operating the treatment and melting plants requires considerable investment. The companies rely on sufficient annual throughput of radioactive metals to keep the facility viable. Any accidents, such as those at the ECOMET-S and the SOCODEI facilities can have significant impact to public opinion and acceptance of the process (Cable News Network International (CNN) 2005, Autorite De Surete Nucleaire (ASN) 2011 and IAEA 2011). These events can have direct impact on a facility's viability and can have knock-on effects on national programmed waste treatment plans. Hence availability, reliability and operability are issues that need to be considered as well as environmental impacts. These have implications on the bottom line costs and hence prices to customers.

Markets Within and Outside the UK Nuclear Industry

The market value and demand for metals play a role in the business case for recycling radioactive metals as discussed above. Exempt metals from the secondary side of nuclear facilities, or primary side radioactive metals decontaminated to exempt levels, can be recycled directly in to open metal markets. Treated radioactive metals can also be beneficially re-used in the nuclear industry.

A comprehensive study of recycling and re-use options for materials from nuclear facilities that cannot be released to material or metal markets is presented in EC

1998a. It presents twelve costed scenarios, eight of which related to LLW metals use and four that were considered viable. These were cast carbon steel boxes, stainless steel waste drums, carbon steel reinforcing bars and carbon steel granulate or fibres.

Siempelkamp already manufacture various ductile cast iron containers for ILW disposal, some of which can be made from 15-25% treated LLW metals that cannot meet exempt levels (Quade and Muller 2005, Quade and Kluth 2009). They also granulate the iron to produce shot for high density concrete shielding and waste boxes. Making reinforcing bars was considered in early French facilities but proved to be unviable.

Manufacturing 500litre stainless steel drums at Sellafield was a potential option given the high demand for these ILW drums and the large amount of LLW iron and steel available at the site. This option was given careful consideration in the early 2000's. It was finally rejected because ILW container manufacture was not considered a core business for Sellafield. Plans were already developing to use international treatment facilities for UK radioactive metals and aligned with the NDA desire to maximise the use of existing techniques and processes.

Making ILW containers from LLW metals still warrants consideration for the future. It closes the cycle and reduces the disposal of 'single-use' new containers. Ductile cast iron containers made partially from recycled LLW steel has been approved by German nuclear and environmental regulators. These containers are thick walled, hence self-shielding. There is a growing interest in them by Magnox decommissioning sites for some ILW streams (NDA 2013a). These containers offer an alternative to the shielded store/unshielded boxes or unshielded store/shielded boxes discussed in Chapter 2. Although the ductile cast iron containers are expensive they require lower engineered interim storage facilities that are commercially and operationally attractive. The Office of Nuclear Regulation (ONR) has stated that Magnox cannot use ductile cast iron containers made from partial recycled LLW in their storage and disposal safety cases (ONR 2011a, 2011b) even though this is allowed by the equivalent German regulator. However, as discussed in Section 6.1,

Magnox are aware of the potential benefits of using their treated cast iron and carbon steel as part of the material for their ductile cast iron containers and they may wish to explore this in the future.

In discussion with nuclear industry representatives it became clear that a case for using recycled LLW iron and carbon steel in future ductile cast iron containers may be possible as the knowledge and acceptance of the underlying metallurgy of the processes improves. However, it was unlikely to be attractive for thin walled stainless steel containers. This is partly due to the high specification for approved ILW containers via the Radioactive Waste Management Ltd 'Letter of Compliance' process discussed in Chapter 2. Another part of the reason for rejecting the option was the anticipated extent of stakeholder management that would be needed to make the option viable. Given the quantities and arisings of LLW metals and the need for ILW containers for long-term storage and disposal it may be possible to investigate this again in the future. It is hoped that this research can help inform any decision on this topic.

National Employment, Skills and Potential Future International Services

The demand for metals in the next 50 years is expected to rise significantly according of Allwood et al 2011. Metals recycling could therefore gain even more prominence and treating radioactive metals for recycling may become a viable business for the UK. It could perhaps also offer the potential for a UK melting and recycling facility. Discussions with Siempelkamp, Studsvik and SOCODEI representatives suggest that reliable incidence of radioactive metals to ensure a reasonable through put would be a deciding factor for investing in a UK facility. The timing of a decision on such a facility would be critical. There appears to be no requirement in the immediate future. For example, the Studsvik Nykoping facility has authorisation to melt up to 5000tonne/y. In 2010 the facility had an annual through put of 3500tonne (Stenmark and Wirendal 2010). Similarly, the Siempelkamp CARLA and GERTA facilities have a 4000tonne/y capacity and an annual through put of 1250-2000tonne/y (Quade and Muller 2005, Quade and Kluth 2009). The current UK joint waste management plans (e.g. Shipton and Falconer 2013) programme about 13,000tonnes of legacy

radioactive metals for treatment between 2013 and 2018, with an annual rate of 2320 to 3390tonne/y. Hence, existing facilities appear to have sufficient spare capacity to meet the current requirements and are actively seeking UK contracts to maximise the utility of their facilities.

However, in the longer term (e.g. 30 years) there may be an opportunity for a UK melting facility. The peaks in incidence of LLW and ILW in Section 2.6.1, plus an NDA graph of projected container demand (NDA 2009a), suggest two possible decision points for a potential UK radioactive metals melting facility. The suggested decision dates for are shown in Figure 10-1 and Figure 10-2.

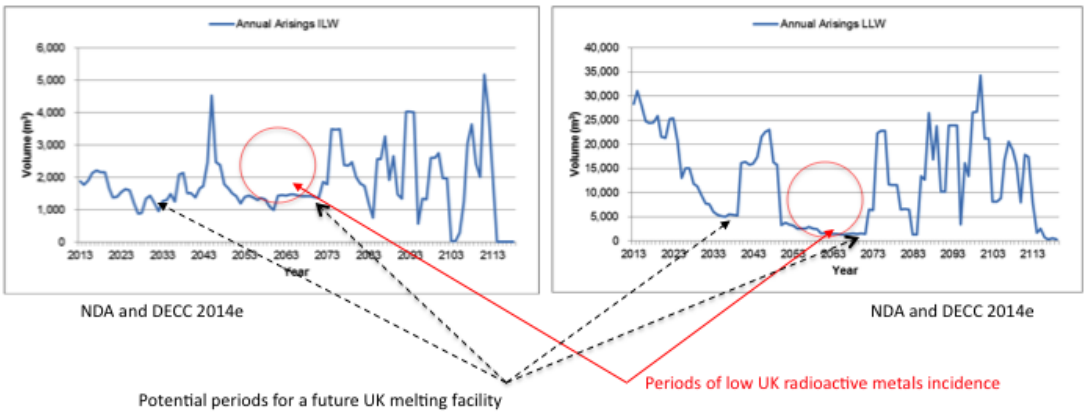

Figure 10-1 Potential periods for constructing a UK melting facility and periods of low incidence of radioactive waste (based on NDA and DECC 2014e Figures 2 and 3)

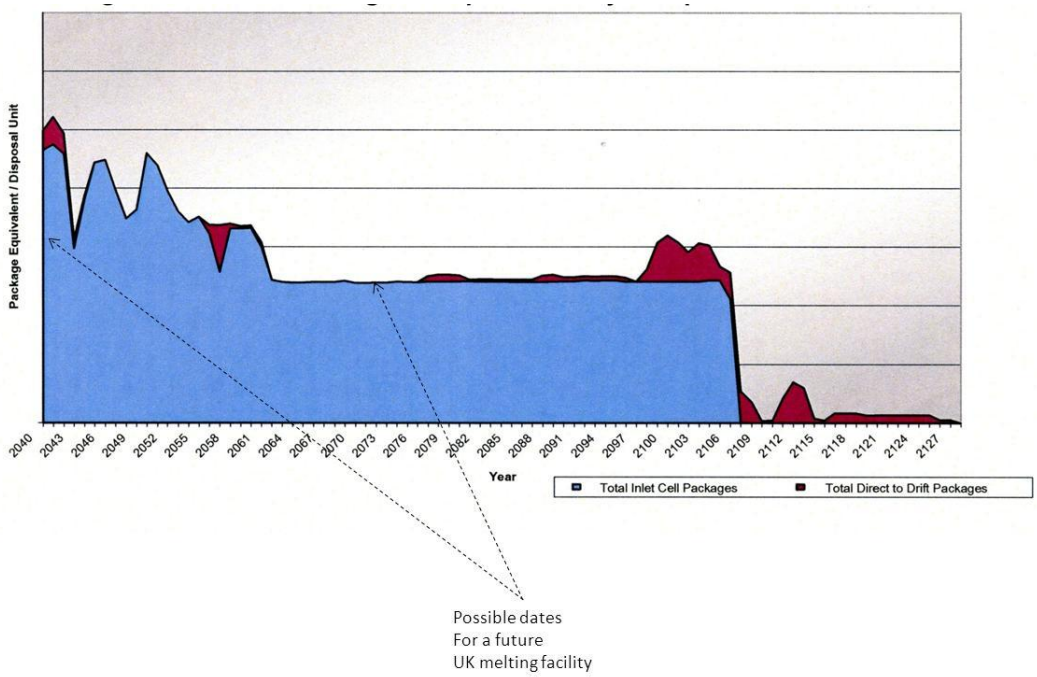

Figure 10-2 Predicted HAW emplacement at a future Geological Disposal Facility showing when a potential future UK radioactive metal melting facility could supply iron and steel for waste container production (based on NDA 2009a, Figure 9)

The first option is just prior to the Sellafield and Springfields decommissioning circa 2035 to allow time for construction and commissioning. The second is just before final decommissioning of legacy waste, circa 2070. If the first option was chosen it could be used to treat and recycle foreign radioactive metals in the quiescent period after the Springfields and Sellafield decommissioning. Alternatively this period could be used to refurbish and update the facility in preparation for final decommissioning. It is hoped that this research can help inform the decision for a potential future UK radioactive melting facility and HAW container production.

10.2.4 Depleted uranium issues

There is an extensive on going NDA supported research project on the potential disposal of DNLEU (NDA 2013a) due to finish in March 2016. There are also existing option studies and economic studies as discussed in Chapter 9. Representatives of the future licensee for a GDF, Radioactive Waste Management (RWM) Ltd, have recently visited Capenhurst to see the Urenco UK Ltd stores and enrichment facilities and to discuss the company's plans for the future disposal of DNLEU (NDA 2014b).

This research supports the findings from previous studies (NDA 2007 and 2013a, Wilson et al. 2012, Hickford et al. 2012) that the baseline disposal option of 500 litre drums may not be the optimal technical solution. It also supports the previous macro-economic study (NDA 2007) which showed that the baseline disposal option was not the lowest financial cost option. That study showed that disposal in 2m boxes and bespoke packaging gave substantial economic benefits. Hence the LCA model results and cost estimates appear to be fairly representative as they both look at the difference in potential environmental and costs impacts for a range of packaging and disposal options.

10.3 Recommendations

This section presents and discusses recommendations susggested to industry and areas for further research.

10.3.1 Recommendations to industry

The research suggest that a number of recommendations could be made to the UK nuclear industry that may help conserve scarce current and future disposal facility capacity and associated packaging demands. These recommendations would also help conserve energy and materials. The recommendations are presented in Box 10-3.

Box 10-3 Recommendations to the UK nuclear industry

1) The financial savings from the ongoing treatment of low level waste metals for recycling or re-use could be used to fund a future UK radioactive metals melting facility.

2) UK depleted uranium owners, regulators and waste disposal facility operators should consider developing bespoke packages for the disposal of depleted, natural and low enriched uranium pending eventual classification as waste.

3) A co-ordinated approach to the disposal of depleted, natural and low enriched uranium could be adopted using a centralised surface facility at a future geological disposal facility or near surface disposal facility.

4) Given the high number of DV70 boxes made redundant if depleted uranium has to be repackaged for disposal it is recommended that these boxes are considered for re-use for the disposal of low level or very low level waste, or contaminated soil.

Recommendation 1 would help make the UK self-sufficient in radioactive waste management, enhancing the national economy and improving industrial knowledge and capability. It would also mitigate the risk of the loss of availability of current international radioactive metals facilities through accidents or changes in national policies affecting the acceptance to UK radioactive metals in America, France, Germany and Sweden.

Recommendation 2 arises from the large variation in potential environmental impacts and costs and the potential interim storage period of depleted, natural and low enriched uranium for up to 100 years. The potentially long interim storage period for the different forms of depleted uranium suggest merit is developing and approving bespoke disposal packages should depleted uranium finally be classified as waste. Bespoke packages was suggested in the NDA assessment of the Urenco UK Ltd Conceptual letter of Compliance and the macro-economic study (NDA 2009b and 2007 respectively), but no actual packages were identified. For example, stainless steel overpacks with sufficient space to allow cement grouting of DV70 boxes to produce an 'approved' waste package would improve the utilisation factor and shielding for transport and disposal. The option would save the addition radiation dose to operators and financial costs of repacking depleted uranium for disposal.

Recommendation 3 offers benefits from a central intermediate level waste packaging facility being considered for spent fuel disposal (NDA 2010b) if it could be extended to include the repackaging of depleted uranium should this remains the preferred disposal requirement. The long-term interim storage DV70 boxes are approved transport packages and the 200 litre drums and 50 litre kegs could be transported to a future disposal facility in 4m boxes and special FHISO as discussed in Chapter 9. The optimum package load could be a mix of super-compacted 200 litre drums and 50 litre kegs in 4m boxes depending on their assay levels. DV70 boxes could be disposed in bespoke packages as discussed in Recommendation 2. This would reduce the repackaging and disposal volume requirements for a future higher activity waste disposal facility.

The re-use of a potentially large number of redundant DV70 boxes in Recommendation 4 is preferable to recycling in the Waste Hierarchy. No melting or international transport would be required and this could be the best environmental and cost option for these packages as the sites storing the depleted uranium.

10.3.2 Recommendations for future research

The research also suggest that a number of recommendations could be made future research in life cycle assessment of waste and decommissioning of radioactive metals and depleted uranium. These recommendations would also help conserve energy and materials. The recommendations are presented in Box 10-4.

Box 10-4 Recommendations for future research

A) Collaborate with the Berkeley boiler project team to undertake a detailed inventory study of process from initial boiler dismantling to induction furnace melting to help validate a generic radioactive metals melting LCA model.

B) Extend the vision of the radioactive metals treatment and melting philosophy to investigate longer term and more problematic metals (i.e. activated metals and decayed or decontaminated metals that may not be able to meet exempt waste level).

C) The LLW Repository Ltd and Nuclear Decommissioning Authority could approach Ecoinvent to produce a radioactive waste disposal impact database for the Low Level Waste Repository (near Drigg) to improve future UK waste and decommissioning life cycle assessments.

D) Since Urenco UK Ltd and Capenhurst Nuclear Services depleted, natural and low enriched uranium dominate the UK inventory consideration could be given a near surface disposal facility close to Capenhurst.

Since undertaking this research fifteen 310tonne Magnox boilers have been transported from Berkeley nuclear power station in Gloucestershire to Nykoping in Sweden for treatment and melting (NDA 2014a). The project has resulted in ~3% residual waste and over 4,000tonne of steel sold in the open metal market. The residual waste will be returned to the UK for disposal but the treatment has saved an estimated 5,500m^3 of disposal volume at the LLW Repository (Drigg). Given that the Berkeley boiler project engineering, radiological, best available technique and costs

data are well documented the project offers an ideal opportunity to improve future LCA modelling of nuclear decommissioning. Hence, Recommendation A could help validate such modelling by:

i. confirming the detailed processes and equipment of the treatment facility and recording the associated energy, materials and waste flows
ii. comparing and contrasting the environmental impacts and costs of the equivalent data for the dismantling and decontamination of the first Berkeley boiler with the treatment project for the remaining boilers
iii. using the environmental and cost data from the Berkeley boiler project to compare with planned disposal or treatment of the UK radioactive waste metals inventory.

The current UK solid low level waste policy and strategy focus on conserving scarce low level waste disposal capacity. A longer term vision of potential uses of treated metals for intermediate level waste containers and other products for the nuclear industry in Recommendation B may be advantageous for treated metals that do not meet exemption level. This would build on historical studies in this area (e.g. EC 1998a) and the experience in Germany of using 15-25% of treated metals in manufacturing ductile cast iron containers as discussed in Chapters 2 and 8.

Recommendation C could be achieved by using the post closure environmental safety case data (e.g. LLWR 2011c) as the basis for UK low level waste disposal in a surface engineered vault facility. This work could be expanded to investigate:

i. the impacts of the production of the various ‘single use’ steel storage and disposal packages used in the UK
ii. the impacts of the various higher activity waste long-term interim storage facilities already built and proposed for the UK

If these proposed inventory databases was produced consistent with Ecoinvent approved procedures the quality of the entries would be guaranteed. Hence these data could be made available to life cycle assessment practitioners and researchers internationally who are interested in nuclear waste management and decommissioning.

The facility in Recommendation D could either be totally new excavated caverns or selected caverns of the redundant Winsford salt mine. The mine is already used for the storage of hazardous waste, albeit currently excluding radioactive waste (British Geological Society (BGS) 2008, Veolia 2011). The chemical hazard of depleted UF_6, which outweighs the radiological hazard, would already have been addressed in the deconvertion to U_3O_8. DV70 boxes are already stored in the Bessines mine in France (Capus and Durante 2007), hence there is precedence. Therefore, it may be possible to extend the specification for Winsford mine storage to meet the current requirements for near surface disposal of solid radioactive waste (Environment Agency (EA) et al. 2009). Significant environmental and financial cost savings may be possible if existing mine caverns could be used or new caverns excavated from mine access tunnels. This is an area of further research and would require consultation with the regulators.

10.4 What this research adds to the current Knowledge

The research confirms that significant reductions in potential environmental impacts and direct financial costs can be achieved by treating UK radioactive metals for recycling and re-use. These benefits are in addition to conserving the scarce radioactive waste disposal volume for current and future repositories that is the focus of current policies, strategies and plans. The research also shows that the range of potential environmental impacts that need to be considered is wider than GHG emissions and energy consumption and that the non-radiological impacts are more significant than the radiological impacts for the wastes considered. Further, it shows that the potential environmental impacts associated with waste disposal containers are a significant proportion of the overall environmental impacts, hence warrant further research to optimise the environmental benefits.

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A Appendix of UK radioactive waste container physical data plus treatment and disposal cost data

This appendix summarises the physical data for the main UK radioactive waste containers for each UK waste category plus container transport, disposal and treatment cost data. It also includes data on UK and International radioactive metals treatment facilities (LLWR Ltd and NDA 2009a), costs for a proposed American dedicated steel mill for radioactive metals (USDOE 2001) and cost for decommissioning the Capenhurst Gas Diffusion Plant (GDP) (CDDUEF 1996). Further, it includes the container data for the storage and disposal of depleted, natural and low enriched uranium.

A.1 Waste Container Physical Data

Box A-1 presents the physical data for the waste containers considered in the research (as summarised in Box 1-2 of the thesis). Also included is physical data for other waste containers that were not considered in the research but may be of interest to other researchers interested in this area. Further details of the containers can be found on the LLW Repository website (llwsites.com) and the Nuclear Decommissioning Authority (NDA) website (www.nda.gov.uk)

Box A-1 UK Radioactive waste containers

Exempt waste: This waste is not included in the research. The data for the waste bags and wrapping is not discussed here.

LAW Containers

Low Volume VLLW: This waste is not included in the research. The packaging is not discussed here but the 200-210 litre drums data could be used as a first approximation.

High Volume VLLW: 200 – 210 litre drum, carbon steel, internal volume ~0.2m3, external volume ~0.25m^3, empty mass 300kg. The data for the pallets and any wrapping needed for transport from the site to the disposal facility is not discussed here.

LLW[1]

Full height ISO freight (FHISO1) container: carbon steel, internal volume 32.1m^3, external volume 38.5 m^3, empty mass 2700kg.

FHISO2 container: carbon steel, internal volume 31.4m^3, external volume 38.3 m^3, empty mass 4800kg.

ISO Skip: carbon steel, internal volume 8.5m^3, external volume 11.5m^3, empty mass 2360kg. – not included in the research

Three Quarter Height ISO (TQHISO) container: carbon steel, internal volume unknown, external volume 29.7m^3, empty mass 3500kg – not included in the research

Two Thirds Height ISO (TTHISO) container: carbon steel, internal volume 22.3m^3, external volume 26.8m^3, empty mass 4000kg – not included in the research.

Half Height ISO (HHISO) container: carbon steel, internal volume 17.9 m^3 (15.5 m^3 useable), external volume 19.5m^3, empty 3080kg.

One Third Height ISO (OTHISO) container: carbon steel, internal volume 11.3m^3, external volume 13m^3, empty mass 3000kg – not included in the research.

HAW Containers

Depleted UF_6 storage and transport containers

Type 48 UF_6 cylinder physical data given in detail in Table A-9

Type 0236 UF_6 cylinder physical data given in detail in Table A-9

ILW – Unshielded Containers

50litre keg: stainless steel, internal volume 0.05m^3, external volume 0.07m^3, empty mass 8kg.

200 litre drum: stainless steel drum, internal volume ~0.2m^3, external volume ~0.25m^3, empty mass 300kg.

500 litre drum: stainless steel drum, internal volume ~0.5m^3, external volume ~0.6m^3, empty mass 120kg

3m^3 drum: stainless steel, internal volume 2.61m^3, external volume 2.85m^3, empty mass 634kg

3m^3 box: stainless steel, internal volume 2.9m^3, external volume 3.3m^3, empty mass 734kg (There are two variants of the box but the basic physical data is the same)

ILW – Shielded containers

4m box (no internal shielding): stainless steel, internal volume 16.49m^3, external volume 21.52m^3, empty metal mass 4200kg.

4m box (100mm thick shielding)[2]: stainless steel, internal volume 13.51m^3, external volume 21.52m^3, empty metal mass 4200kg, not specifically included in the research

4m box (200mm thick shielding)[2]: stainless steel, internal volume 10.90m^3, external volume 21.52m^3, empty metal mass 4200kg, not specifically included in the research

4m box (300mm thick shielding)[2]: stainless steel, internal volume 8.08m^3, external volume 21.52m^3, empty metal mass 4200kg, not specifically included in the research

2m box (no internal shielding)[2]: stainless steel, internal volume 8.17m^3, external volume 10.56m^3, empty metal mass 3000kg, not specifically included in the research

2m box (100mm thick shielding)[2]: stainless steel, internal volume 6.99m^3, external volume 10.56m^3, empty metal mass 3000kg, not specifically

included in the research

6m^3 concrete box[3]: for Sellafield beta-gamma wastes, purely concrete hence not included in the research

WAGR Box[3]: especially designed for specific WAGR ILW wastes, not included in the research

Ductile Cast Iron Containers[4]

Konrad Type VI box: ductile cast iron container, internal volume 2.83m^3, external volume 5.39m^3, empty mass 18300kg, not included in the research

Mosaik Type II flask: ductile cast iron flask, minimum shielding, internal volume 0.49m^3, external volume 1.33m^3, empty mass 6000kg, not included in the research

Mosaik Type II flask: ductile cast iron flask, maximum shielding, internal volume 0.165m^3, external volume 1.33m^3, empty mass 6000kg, not included in the research

HLW Containers

HLW is not included in the research hence the containers are not discussed.

SNF Containers

SNF is not included in the research hence the containers are not discussed

Notes

1 – The specialised 1/3 and ½ height fissile disposal containers and WAMAC disposal containers are not included in the research hence are no discussed here.

2 - The shielded variants of the 4m box and the unshielded and shielded variants of the 2m box were not included in the research. They are presented here for completeness and for the potential use by future researchers.

3- The 6m^3 concrete box and the WAGR box were not included in the research as they were deemed inappropriate for the radioactive metals of interest and are only mentioned for completeness.

4 - The DCIC were not included in the research. They are presented here for completeness and for the potential use by future researchers.

A.2 Waste Container Cost Data

The container cost data used for the cost estimates presented in Section 7.4.3, Table 8-2, Tables 9-5 and 9-6 is summarized in Table A-1.

Table A-1 Published cost sources for key waste containers

Waste	Containers	Costs	Sources Cost Data
VLLW	200- 210 litre carbon steel drum	£25 - £38 £35	Internet search LLWR 2014
LLW	FHISO	£8570 – 2011/12 £12,470 – 2013/14	Own estimate extrapolating for HHISO mass & cost
	HHISO	£5500 – 2011/12 £8000 – 2013/14	LLWR Ltd llwrsite.com 2011, 2012 and 2014
	HHISO titanium liner	£3,390 - £3,890/te	Based on titanium prices 2014 and mass of titanium in Hickford et al 2012
	THISO	£4800 – 2011/12 £10,000 – 2013/14	LLWR Ltd llwrsite.com 2011, 2012 and 2014 for basic price and variants

Waste	Containers	Costs	Sources Cost Data
HAW	50 litre keg	£60 - £125	Own estimate, based on 200l drum and stainless steel prices 2014
	200- 210 litre stainless steel drum	£120 - £250	From websites, e.g. James G. Carrick & Co Ltd and own estimate 2014
	500litre drum	£1500 £750 - £1,570	Bryan 2005 Own estimate based on $3m^3$ box and HHISO costs and stainless steel prices in 2014
	500litre drum stillage	£1,490 0 £1,850	Own estimate based on NDA 2008 1te stillage mass and stainless steel prices in 2014
	$3m^3$ drum	Same as $3m^3$ box	Same as $3m^3$ box
	$3m^3$ box	£4,500 - £5,500 £25,000 £20,000	Ove Arup 1997 development costs. CoRWM 2008. NDA 2012a
	2m box	Not estimated as not used in the research	Not Applicable
	4mbox	£60,000 £16,000 - £32,000 £11,000 for possible carbon steel variant	NDA 2012a Own estimate, assuming mass production, HHSO prices and stainless steel prices in 2014
Long-term Interim Storage Packages	DV70 boxes for U_3O_8 powder	£1,950 - £2,270	Commercially confidential, hence own estimate based on HHISO, 3m3 box and carbon steel and stainless steel prices in 2014
	Type 48 cylinders for UF_6	Unknown	Commercially confidential, hence not available
	Type 0236 cylinders for UF_6	Unknown	Commercially confidential, hence not available

The associated costs data for the treatment and disposal of metals, presented in Section 7.4.3, Table 8-2 and Tables 9-5 and 9-6, are summarised as:

- Decontamination costs were estimated as £563/te (USDOE 2001 in Table A-6) or £570/te (CDDUEF 1996 in Table A-7), both were converted from US dollars to £ Sterling for the year reported and inflated to 2013 prices using a Bank of England average annual inflation rate on-line calculator.

- Size reduction costs were estimated as £375/te (USDOE 2001 in Table A-6) or £385/te (CDDUEF 1996 in Table Table A-7), both converted from US dollars to £ Sterling for the year reported and inflated to 2013 prices using a Bank of England average annual inflation rate on-line calculator.
- VLLW disposal cost was £300/m^3 - £700/m^3 with an average of £500m^3 from LLWR 2014, and £500/te to £2000/te from LLWR and Entec 2009.
- LLW disposal cost was £1,735/m^3 (2012/13), £2911/m^3 (2013/14) for WAGR boiler disposal costs and £2990/m^3 (2014/15) for UK metals inventory (LLWR 2014).
- The activity cost per nuclide for LLW wastes is given on the llwrsite.com website documents, e.g. Service Price – Overview or Service Price List (i.e. £0.01/MBq for tritium, £0.2/MBq for cobalt and other nuclides, ££9.0/MBq for uranium and other alpha emitters, £55.0/MBq for radium-226/thorium-232, carbon-14 and iodine-129).
- Disposal cost for super-compacted waste was £526/m^3 (2012/13), £882/m^3 (2013/14) and £906/m^3 (2014/15) from llwrsite.com website documents, e.g. Service Price – Overview or Service Price List.
- LLW and VLLW UK transport cost was £600/trip based on a single HHISO or equivalent pallets for VLLW. The LLW overseas transport cost was £5,500 per trip (based on the average for Europe), from llwrsite.com documents for 2011 and 2014, (e.g. LLWR 2014)).
- HAW disposal costs were £4,000/m^3 (NDA 2007), £15,000/m^3 - £20,400/m^3 (DECC 2009), ILW shielded package disposal £3,840/package, unshielded package disposal £10,260/package and baseline disposal £8,380/package (NDA 2012a).
- HAW transport cost at £1,250/m^3 from NDA2012a
- The disposal cost for ~780te of Magnox metallic waste at the planning norm of 10te/HHISO for 2013/14 gave £5.89m, based on a disposal cost of £66,765/HHISO (i.e. £3,424/m^3 for 19.5m^3 per HHISO) (Shipton and Falconer 2013). The LLW Repository Ltd disposal volume cost was £2911/m^3, the additional cost was for containers, transport and activity charges.

- The treatment cost for the ~780te of Magnox metallic waste was £3.56m. i.e. £4571/te (Shipton and Falconer 2013). The planning norm metallic waste treatment cost was £3500/te (Shipton and Falconer 2013), the additional costs is from containers, transport (UK and Europe) and disposal.

A.3 Radioactive Metals Treatment Facility Summary Data and Costs

Data for a recent UK decontamination study, a proposed dedicated steel mill in the USA and the decommissioning of the Capenhurst gas diffusion plant (GDP) are given to show the large variation in cost and for comparison with the values presented in the thesis.

A.3.1 UK and International decontamination facility data

LLWR Ltd and NDA 2009a summarises a study of UK and international facilities used by UK nuclear sites for decontaminating radioactive metals. There are fixed and mobile plant/equipment currently available in the UK that can decontaminate intermediate level waste metals to low level waste (e.g. Hinkley Point A) or from low level waste to exempt levels (e.g. Sellafield MRF, Winfrith WACM and Studsvik MRF) as presented in Table A-2.

Table A-2 Summary of UK fixed decontamination facilities data at 2009

Key Issues and Costs	**Sellafield (MRF)[1]**	**Winfrith (WACM)[2]**	**Hinkley Point A (UHPWJ)[3]**	**Studsvik UK Ltd MRF**
Year Commissioned	2005	2000, relocated 2009	2005/06	2009[5], estimated cost £6million[6]
Capacity (t/y)[4]	500	100 to 150	Not specified	3000[5]
Processed (t)	339	143	76	20t/week, 1000t/y[6]
Predicted Process Amount (t) 2009/10	400	100	Unspecified but to plan	1000 estimated
Recycled (t)	339	15	Pre-treatment only	95.9%[7]
Secondary Waste (t)	30 (8.8%)	1.6 (1.1%)	0.7 to 0.9 (1.1%)	4.1%[7] based on 95.9% recycling
Staff	36	5	10	20[8]
Operating hours per annum	5824	1400 or 175 days/year	On Demand	Unknown
Cost (Annual)	£3042.4k/year	£1736.09 per day, £303.8k/year	Not provided	Unknown
Cost (Specific)	£7.61/kg	£3.04/kg	Unknown	Unknown
Price (Specific)	£0.32 to £5.6/kg	£1736.09/day	£50/kg	Unknown
Service offered to other sites	Yes	Yes	No	Yes
NDA Approved Business Case	No	No	No	Not Applicable

Key Issues and Costs	Sellafield (MRF)[1]	Winfrith (WACM)[2]	Hinkley Point A (UHPWJ)[3]	Studsvik UK Ltd MRF
Energy Consumption	Not Specified	Not Specified	Not Specified	Not Specified
Water Consumption	Not Specified	Not Specified	Not Specified	Not Specified
Liquid Emissions	Not Specified	Not Specified	Not Specified	Not Specified
Gaseous Emissions	Not Specified	Not Specified	Not Specified	Not Specified
Metal decontaminated	Steel (carbon and stainless), Lead (separate plant), galvanised materials	Steel (carbon and stainless, lead, copper and aluminium cables, titanium	Carbon steel, others possible as required	Steel (carbon and stainless, cast iron, aluminium & copper (plus cables), brass and lead

Key - 1 MRF - Metal Recycling Facility. Note, this is separate to the Studsvik MRF at Lillyhall, 2 WACM - Winfrith Abrasive Cleaning Machine, 3 UHPWJ - Ultra High Pressure Water Jetting for removing paint from pond skips, 4 t is tonne (1000kg), 5 From Robinson 2009, 6 Based on NDA 2010c, 7 From Wilkinson 2011 and 8 NWBLT 2011

Information on the international facilities used by UK nuclear sites is summarised in Table A-3 and Table A-4. LLWR Ltd and NDA 2009 do not include summary data for the SOCODEI CENTRACO facility in France or the Ecomet-S Facility in the Russian Federation as these facilities are not currently used by UK nuclear operators. Data for these latter facilities are summarised in Table A-5.

Table A-3 Summary of International treatment facility capability data (LLWR Ltd and NDA 2009a)

Plant Capability	Studsvik MRF	Siempelkamp Germany[1,2]	Bear Creek USA (melting)[3]	Studsvik Sweden[4,5]
Capacity (tonnes/year)	3000	4000	4000	5000
Weight limitations	5 tonnes	Heavy weight lifting can be arranged	Approximately 9.1tonnes (20,000lb)	300+ tonnes
Size limitations	Min. thickness 3mm			4m diameter for cutting
Typical % of secondary waste	Assumed to be similar to Studsvik Sweden	5%		4%[6]
Activity or dose rate limitation	Surface dose rate <0.2mSv/h averaged per item. No hot spots, then > or = 0.5mSv/h Average specific activity per container < 500Bq/g	Specific total activity , < 1000Bq/g For the nuclides H3, C14, Fe55 and Ni63 <10,000 Bq/g in total	0.2mSv/h, but not clear if this is an average surface dose rate.	Surface dose rate <0.2mSv/h "hot spots", up to 0.5mSv/h Allowed after acceptance

Plant Capability	Studsvik MRF	Siempelkamp Germany[1,2]	Bear Creek USA (melting)[3]	Studsvik Sweden[4,5]
Recycling of metals	Yes, according to substances of low activity levels	Yes, as shielding blocks, waste containers and heavy concrete	Yes, e.g. shielding blocks within the nuclear industry	Yes, according to European Commission (EC) 1998b

Notes

1 This may be the combined annual limit for the CARLA (Centrale Anlage zum Recyclieren Leichtakiver Abfalle - central Plant for recycling slightly radioactive waste) and GERTA (Grosstechnische Einrichtung zum Recyclieren Toxischer Abfalle –industrial-scale melting of the recycling of toxic waste) plants at Krefeld in Germany.

2 CARLA uses a 3.2tonne medium frequency induction furnace with a melt performance of 2ton/h for steel for 1 tonne ingots or granules (Quade and Kluth 2009, LLWR Ltd and NDA 2009, Buckley et al 2004). GERTA uses a 8tonne net frequency induction furnace to melt metals contaminated with NORM, mercury, asbestos PCB (Polychlorinated biphenyl) and dioxins/furans (www.siempelkamp.com/GERTA-Melting-Plant-for-Metals-with-Toxic- Contamination, Buckley et al 2004).

3 The specification for the Bear Creek furnace is not given in LLWR Ltd and NDA 2009 but it is presumed to be a 3 to 4te medium frequency induction furnace. LLWR Ltd and NDA 2009 notes "importing metallic waste to the U.S. is currently under review by US Nuclear Regulatory Commission which might limit this route in the future."

4 Lorenzen and Lindberg 2000 state that "…two induction furnaces in carrousel geometry... Each furnace has a capacity for steel of approximately 3.5 tons - resulting in 10-15 ingots per 8hr working day…"

5 LLWR Ltd and NDA 2009 notes the Studsvik Nykoping facility has a licensed capacity of 5000tonne/y but a maximum capacity of 8000tonne/y.

6 Ashton 2013 notes that the typical recycling rate for containerised metals to Nykoping in 95% with 5% secondary waste. Ashton also notes that for large metallic items requiring shielded treatment the recycling rate is in the order of 80% to 85% with secondary waste of 15% to 20%.

Table A-4 Summary of Metals treated at UK and International treatment facilities data (LLWR Ltd and NDA 2009a)

Metals	Studsvik MRF	Siempelkamp Germany	Bear Creek USA (melting)	Studsvik Sweden	Sellafield	Winfrith
Metals – surface contaminated possible to treat						
Carbon Steel	Yes	Yes	Yes	Yes	Yes	Yes
Stainless Steel	Yes	Yes	Yes	Yes	Yes	Yes
Copper	Yes	Yes	After approval	Yes	No	No
Aluminium	Yes	Yes	After approval	Yes	No	No
Brass	Yes	Yes	After approval	Yes	No	No
Lead	Yes	Yes	After approval	Yes	No	Yes
Titanium	No	No	No	No	No	No
Cables – Cu or Al conductor	Yes	Not Known	After approval	Yes	No	Yes
Other cables	No	Not Known	Not Known	No	No	No
Galvanised material	No	Yes	After approval	No	Yes	No
Metals – Other						
Tritiated metals	Yes	Yes	Not Known	After approval	No	Yes

Metals	Studsvik MRF	Siempelkamp Germany	Bear Creek USA (melting)	Studsvik Sweden	Sellafield	Winfrith
Activated Metals	Yes	Not Known	Not Known	After approval	No	No
Uranium contaminated Metals[1]	Yes	NORM[2]	Not Known	Yes	No	No
Alpha dominated contaminated metals such as Plutonium contaminated Material	After approval	Not Known	Not Known	After approval	No	No

Notes

1 – See Lorenzen and Lindberg 2000.

2 – Metals contaminated NORM (naturally occurring radioactive material) are processed at the GERTA plant at Krefeld.

Cu – copper, Al - aluminium

Table A-5 Summary of capability of other International treatment facilities circa 2009

Plant Capability	SOCODEI CENTRACO Marcoule, France[1,2,3]	ECOMET – S Russian Federation[4,5]
Capacity (tonnes/year)	4500	5000
Weight limitations	Standard ISO freight container limits	50ton limit, items 26-33tons treated
Size limitations	Standard 2m, 1tonne Custom 12m, 15 tonne	Tank diameter to 2.9m and 2.5m high treated
Typical % of secondary waste	Not specified	Design (9%), average 5-7%, max ~16%
Activity or dose rate limitation	Beta and gamma 20,000 Bq/g max Alpha 370 Bq/g max	100 GBq total, 18.9 GBq actual,
Recycling of metals	Yes, waste containers and shielding cylinders	Yes, unrestricted re-use in industry, or restricted re-use in nuclear industry

Notes

1 - Data taken from Institut De Radioprotection Et De Surete Nucleaire (IRSN) 20004. Carbon steel, stainless steel and non-ferrous metals melted.

2 - Additional data and confirmation of data from Buckley et al 2004.

3 - The CENTRACO facility has a storage capacity of 3000 tonne and a daily production of up to 24tonne/day. Buckley et al state the high frequency induction furnace has a capacity of 4 tonne. IRSN 2004 note the melt charge can be cast into ingot or poured into a centrifuge to produce shielding cylinders to tubes for waste drums. The volume reduction claimed for the process in 1/10 for ingots and 1/20 for re-useable products (IRSN 2004) but Buckley at al 2004 note this may be optimising and a volume reduction factor of 4-6 may be more realistic.

4 - Data taken from Gelbutovsky et al. 2006, carbon steel, stainless steel, non-ferrous metals including aluminium and copper-nickel allows melted. Metal sections are cut in maximum lengths of 500mm or 800m dependent on the type of component. A 2.5tonne induction furnace is used for steels, an electric heating over is used to melt aluminium and lead.

5 - Data taken from Gelbutovski et al. 2009, metal section from 150 – 300m thickness and weight of 200 – 250 kg melted. The facility can also accept NORM contaminated metals from the oil and gas industry.

A.3.2 American dedicated steel mill proposal data

A conceptual study was undertaken in America to evaluate the potential for a dedicated steel mill for radioactive metals (USDOE 2001). A summary of the costs is presented in Table A-6.

Table A-6 Estimated cost for a proposed American dedicated steel mill for radioactive metals at 2001

Cost Assumption	Model Costs ($, 2000)	Model Costs (£, 2000)	Model Costs (£, 2012)	Units	Alternative Source	Comments
Purchase Costs of Carbon Steel (Hot-rolled)	380	251.2	356.6	£/te	575	DOE Range of original values, $360 to $400/ton Range from Allwood 2010 £400 to £750/te)
Purchase Cost of Stainless Steel	2200	1454.1	2064.8	£/te		Range of original values, 304SS($1800 to $2000/ton), 316SS($2400 to $2600/ton)
Selling Price of Scrap Carbon Steel	67.5	44.6	63.4	£/te	50	DOE Original value $90/ton with 25% discount, Allwood 2010 range £50 to £300/e)
Selling Price of Scrap Stainless Steel	450	297.4	422.3	£/te	300	Original value $0.3/lb with 25% discount, Allwood 2010 range £50 to 300/te)
Decontaminate Scrap Metal	600	396.6	563.1	£/te	570	Capenhurst decontamination for ~27500 to ~31,900 te of scrap metal for £10m, CDDUEF 1996, inflated by 1.68 (Bank of England)
Quality Assurance for Free Release of Scrap Metal	190	125.6	178.3	£/te		
Nickel Processing to Remove Tc-99	2000	1321.9	1877.1	£/te		Original value $1/lb
Cut into 5-ft pieces	400	264.4	375.4	£/te	385	Capenhurst disassembly ~130,000te concrete @£125/te, ~30,000te metals at £679/te for £20m, inflated by 1.68 (Bank of England)
Storage Costs	2	1.3	1.9	£/te-y		
Operation Cost of MSC Induction Furnace	7000	4626.6	6569.7	£/te		Original value $3.5/lb, this is presumed to be the same as the

Cost Assumption	Model Costs ($, 2000)	Model Costs (£, 2000)	Model Costs (£, 2012)	Units	Alternative Source	Comments
						Energy Solutions Bear Creek plant
Disposal Cost @ Hanford ERDF	62	41.0	58.2	£/m^3		Specialist on-site disposal
Disposal Cost @ Fernald OSDF	135	89.2	126.7	£/m^3		Specialist on-site disposal
Disposal Cost @ ORR EMWMF	180	119.0	168.9	£/m^3		Specialist on-site disposal
LLW Disposal Cost @ Hanford/NTS/Envirocare	605	399.9	567.8	£/m^3		Range of original values, $516 to $709 per cubic metre
MLLW Disposal Cost @ Hanford/NTS/Envirocare	1002	662.3	940.4	£/m^3		Range of original values, $889 to $1228 per cubic metre
Retrofitting Existing EAF	22,440,000	14,831,461	21,060,674	£		
Move EAF to Nuclear Site	82,704,000	54,662,260	77,620,410	£		
Purchase New EAF	94,627,000	62,542,631	88,810,535	£		
Permitting and Licensing	5,000,000	3,304,693	4,692,664	£		
Instrumentation to Measure Radioactivity	1,000,000	660,939	938,533	£		Original value $200,000 per unit
Operation of EAF	12,575,000	8,311,302	11,802,049	£/y		This assumes a 6 year operational life, which is short
Maintenance of Permits/Licensing	500,000	330,469	469,266	£/y		
Decommissioning and Dismantling of EAF	100	66.1	93.9	£/ft^2		Original value for decontamination & decommissioning cost at Fernald

EAF – Electric Arc Furnace, Hanford ERDF – Environmental Restoration Disposal Facility, MSC – Manufacturing Sciences Corporation, NTS – Nevada Test Site, ORR EMWMF – Oak Ridge Reservation Environmental Management Waste Management Facility, OSDF – On Site Disposal Facility.

A.3.3 Capenhurst Gas Diffusion Plant decommissioning cost data

The estimated cost of decommissioning the Capenhurst gas diffusion plant (GDP) and recycling metals, concrete, plant and equipment in the mid 1990s was £86m ($160m) at 1994 prices (CDDUEF 1996). The cost breakdown is summarised in Table A-7 and quantities of material is summarised in Table A-8. The melting facility details are discussed in Bradbury et al. 1995, Christopher et al. 1998, Clements 1998, plus CDDUEF 1996.

Table A-7 Cost breakdown for the decontamination and decommissioning of the Capenhurst GDP at 1994 prices presented in 1996

Cost Element[a]	£million	Percentage	$million
Pre-treatment	2	2.3	3.72
Planning & Management	10	11.6	18.6
Technology Development	17	19.8	31.63
Characterisation	2	2.3	3.72
Disassembly	20	23.3	37.21
Removal and treatment of hazardous waste	2	2.3	3.72
Decontamination	10	11.6	18.60
Metal Melting[bc]	2	2.3	3.72
Health and Safety	2	2.3	3.72
Monitoring (including analytical)	7	8.1	13.02
Radioactive waste treatment and disposal	3	3.5	5.58
Overhead	8	9.3	14.88
Total	86	100	160.0

Notes a)Data came from CDDUEF 1996, b) Using a "… 2Mg capacity oil fired, sloping hearth reverberatory furnace and 2 Mg coreless induction furnace for aluminium melting. A 1.5 Mg capacity induction furnace is available for steel melting" (Christopher et al. 1998), c) "The facility had a throughput of up to 150 tonnes per week…" (Clements 1998)

Table A-8 Capenhurst GDP Material Quantities

Material	GDF Quantity (tons)[a]	GDF Quantity (tons)[b]	Quantity Recycled (tons)[b]
Aluminium	8,300	8,000	5,000
Ferrous metals/steel	14,200	23,000[c]	23,000[c]
Nickel	400	320	Unspecified
Copper & Brass	Included in Misc Metals	70 Cooper, 300 Al Bronze	Unspecified
Monel	Included in Misc Metals	200 cupro-nickel	Unspecified
Miscellaneous Metals	4,500	unknown	Unspecified
Total	27,400	31,890	28,000

Notes a) CDDUEF 1996, b) Bradbury et al 1995, c) This may include some of the 18000t to 35,500t of structural steel Bradbury et al 1995 and CDDUEF 1996

The data suggests a 'real' metal recycling cost between £2700/t and £3100/t at 1994 prices. Capenhurst sold the aluminium, steel and nickel in the open market to off-set the project costs, but the sale values are not known. Bradbury et al 1995 states that about 67,300m^3 of LLW disposal capacity was saved by the free release/recycling project saving Capenhurst about £97m ($180m). Hence the cost/tonne depends on what is included in the calculation.

A.3.4 Depleted, natural and low enriched uranium data

The following data are provided as the basis for the tables and scenarios presented in Chapter 9 of the thesis:

- Table A-9 Physical data for UF_6 cylinders stores at Urenco UK Ltd and Capenhurst Nuclear Services at 2015 (Table A-9),

- Table A-10 Urenco UK Ltd Type 48 UF_6 storage cylinder data for 100,000 tU currently stored and 5000 tU/y production at 2015
- Table A-11 Capenhurst Nuclear Service Type 48 and Type 0236 UF_6 storage cylinder data at 2015
- Table A-12 Estimated DV70 boxes storage volume data for Urenco UK Ltd Uranium Oxide Store (100,000 tU as 118,000 tU_3O_8) at 2015
- Table A-13 Estimated DV70 boxes storage volume data for Urenco UK Ltd Uranium annual enrichment contract work (5,000 tU/y as 5,900 tU_3O_8 /y) at 2015
- Table A-14 Estimates of 200l drums and 50l kegs for Capenhurst Nuclear Services UO_3 storage (for 30,000tUO_3 of Magnox Depleted Uranium (MDU) and 5,000tUO_3 of THORP Product Uranium (TPU)) at 2015
- Table A-15 Estimates DV70 boxes for Capenhurst Nuclear Services UO_3 converted to 29,500tU_3O_8 at 2015
- Table A-16 Estimates of 500l drums for UK depleted uranium (for all waste converted to U_3O_8 powder only, hence 1.87te waste per load) at 2015
- Table A-17 Estimates of 500l drums for UK depleted uranium (for all waste converted to U_3O_8 powder mixed with cement, hence 0.97t waste per load) i.e. basis of Deep Disposal Scenario 1 (DDS1)
- Table A-18 Estimates of grouted 4m boxes for UK depleted uranium (for all waste converted to U_3O_8 powder) basis for DDS3
- Table A-19 Estimates of super-compacted 200l drums in ungrouted 4m boxes for UK depleted uranium (for all waste converted to U_3O_8 powder) basis for DDS4
- Table A-20 Estimates of $3m^3$ drums for UK depleted uranium (for all waste converted to U_3O_8 powder) basis for DDS5
- DDS2 HHISO/Liner assumed a waste mass of 191,160te and 5,400 HHISOs with titanium liners.
- Table A-21 Estimate of overpacked depleted uranium in U_3O_8 and UO_3 waste containers, the bases of DDS6 and Shallow Disposal Scenario 1 and 2 (SDS1 and SDS2).

Table A-9 Physical data for UF_6 cylinders stored at Urenco UK Ltd and Capenhurst Nuclear Services at 2015

Cylinder	Length (m)	Outside Diameter (m)	Wall Thickness (mm)	Disposal Volume (m^3)	Internal Volume (m^3)	Tare Weight (tonne)	Load Weight (tonne)	Gross Weight (tonne)	Material
Type 48[1]	3.81	1.219	12.7	4.04	N/C	2.359	12.501	14.86	Carbon Steel
Type 0236[2]	1.525	0.775	6.4	~0.72	N/C	~0.3	U/K	U/K	Carbon Steel

Notes 1) From IAEA 1995, 2) From Drawing OCNF7380 supplied by Mr. D. Nelligan of Capenhurst Nuclear Services N/C not calculated, U/K unknown

Table A-10 Urenco UK Ltd Type 48 UF_6 storage cylinder data for 100,000 tU currently stored and 5000 tU/y production at 2015

Container	No. Of Cylinders	Tare Mass of Cylinders	External Volume ($4.04m^3$/cylinder)	Store footprint area	Material
Type 48 Cylinders	8000	~18,870te	~32,320m^3	Not estimated	Carbon Steel
Type 48 Cylinders	400	940te	~1620m^3	Not estimated	Carbon Steel

Table A-11 Capenhurst Nuclear Service Type 48 and Type 0236 UF_6 storage cylinder data at 2015

Container	No. Of Cylinders	Tare Mass of Cylinders	External Volume	Store footprint area	Material	Estimated Split of 25,000tUF_6
Type 48 Cylinders	1,500	~3540te	~6060m^3	Unknown	Carbon Steel	18,750t (@12.5t per cylinder)
0236 Cylinders	10,000	~3000te	~7200m^3	Unknown	Carbon Steel	6,250t

Table A-12 Estimated DV70 boxes storage volume data for Urenco UK Ltd Uranium Oxide Store (100,000 tU as 118,000 tU_3O_8) at 2015

DV70 Loading	No. Of Boxes	Mass of Boxes at $0.75t^2$ each	External Volume for $3.6m^3$/box	External Volume for $3.68m^3$/box	Store footprint area at $3.55m^2$/DV70 stack[1]
10te	~11,800	~8850te	~42,480m^3	~43,420m^3	13,960m^2
10.5te[3]	**~11,240**	**~8430te**	**~40,460m^3**	**~41,360m^3**	**13,300m^2**
11te	~10,730	~8050te	~38,630m^3	~39,490m^3	12,700m^2

Note 1) The footprint area assumes boxes are stacked 3 high and 2200 boxes need $2600m^2$ of floor space (OECD/IAEA 2001,i.e. $3.55m^2$ per three high stack of DV70 boxes). Density ranges from 2.7t/m3 to 3.1t/m3 for 10 to 11t load. 2) Personal Communication Jones 2014 (16/5/14). 3) Waste loading of 10.5te used.

Table A-13 Estimated DV70 boxes storage volume data for Urenco UK Ltd Uranium annual enrichment contract work (5,000 tU/y as 5,900 tU_3O_8/y) at 2015

DV70 Loading	No. Of Boxes	Mass of Boxes at $0.75te^2$ each	External Volume for $3.6m^3$/box	External Volume for $3.68m^3$/box	Store footprint area at $3.55m^2$/DV70 stack[1]
10te	~590	~440te	~2120m^3	~2170m^3	700m^2
10.5te[3]	**~560**	**~420te**	**~2020m^3**	**~2060m^3**	**660m^2**
11te	~540	~410te	~1940m^3	~1990m^3	640m^2

Note 1) The footprint area assumes boxes are stacked 3 high and 2200 boxes need $2600m^2$ of floor space (OECD/IAEA, 2001,i.e. $3.55m^2$ per three high stack of DV70 boxes). Density ranges from 2.7t/m3 to 3.1t/m3 for 10 to 11t load. 2) Personal communication Jones 2014 (16/5/14). 3) Waste loading of 10.5te recommended if annual production to be included.

Table A-14 Estimates of 200l drums and 50l kegs for Capenhurst Nuclear Services UO_3 storage (for 30,000tUO_3 of Magnox Depleted Uranium (MDU) and 5,000tUO_3 of THORP Product Uranium (TPU)) at 2015

Container	No. Of Drums	Tare Mass of Drums	External Volume	Store footprint area	Material
MDU 200 litre Drums	60,000	~1116te	~17,760m^3	~7440m^2	Stainless Steel
TPU (LEU) 50l kegs (200kg per drum)	25,000	~200te	~1655m^3 drums only	Not Estimated	Stainless Steel
TPU (LEU) 50l kegs with overpacks	12,500	~1000te	~3470m^3 for 50% overpacks	Not Estimated	Stainless Steel

Note – data from NDA, 2007, "… 500kg of product is stored in each [200 litre] drum…" Foot print (60,000/3)*0.31*1.2 = 7740m^2 and pers. comm. Jones 2014 (20/5/14).

Table A-15 Estimates DV70 boxes for Capenhurst Nuclear Services UO_3 converted to 29,500tU_3O_8 at 2015

DV70 Loading	No. Of Boxes	Mass of Boxes at 0.75t[1] each	External Volume for 3.6m^3/box	External Volume for 3.68m^3/box	Store footprint area at 3.55m^2/DV70 stack[2]
10te	~2950	~2210te	~10,620m^3	~10,860m^3	3490m^2
10.5te	**~2810**	**~2110te**	**~10,120m^3**	**~10,340m^3**	**3330m^2**
11te	~2680	~2010te	~9650m^3	~9860m^3	3170m^2

Note 1) Personal communication from Jones 2014 (16/5/14). 2) The footprint area assumes boxes are stacked 3 high and 2200 boxes need 2600m^2 of floor space (OECD/IAEA 2001,i.e. 3.55m^2 per three high stack of DV70 boxes). 3) Waste loading of 10.5te used

Table A-16 Estimates of 500l drums for UK depleted uranium (for all waste converted to U_3O_8 powder only, hence 1.87te waste per load) at 2015

Waste U_3O_8 and UO_3	No. of drums[1]	Mass of SS drums	Volume of drums	Volume of stillages	Store footprint area 6 high stillage footprint x 1.2	Stillage Mass
118,000te	~63,100	~8200te	~39,120m^3	~57,100m^3	~37,360m^2	~15,780t
191,160te[2]	**~102,230**	**~13,290te**	**~63,380m^3**	**~92,510m^3**	**~60,520m^2**	**~25,560t**
212,400te	~113,580	~14,770te	~70,420m^3	~102,790m^3	~67,240m^2	~28,400t

Note, 1) NDA 2007 estimates a total depleted uranium mass of 60,000t resulting in 50,000 of 500l drums and 12,500 stillages. 2) Waste mass of 191,160te recommended to be used if this scenario to be tested. SS – stainless steel.

Table A-17 Estimates of 500l drums for UK depleted uranium (for all waste converted to U_3O_8 powder mixed with cement, hence 0.97t waste per load) i.e. basis of Deep Disposal Scenario 1 (DDS1)

Waste U_3O_8 and UO_3	No. of drums	Mass of SS drums	Volume of drums	Volume of stillages	Store footprint area 6 high stillage footprint x 1.2[1]	Stillage Mass
118,000te	~122,030	~15860te	~75,660 m^3	~110,430m^3	~72,240m^2	~30,510t
191,160te[2]	**~197,680**	**~25,700te**	**~122,560m^3**	**~178,900m^3**	**~117, 030m^2**	**~49,420t**
212,400te	~219,650	~28,550te	~136,180m^3	~198,780m^3	~130,030m^2	~54,910t

[1] Estimate of the store footprint for four 500l drums in a stillage, with 2.96m^2 footprint stacked 6 high (NDA 2008). It was assumed that the store footprint for each stillage stack would be 1.2 x the actual stillage footprint. This is consistent with the 3.55m^2/DV70 stack and assuming the DV70 footprint is 2.96m^2. 2) Waste mass of 191,160te used. SS – stainless steel.

Table A-18 Estimates of grouted 4m boxes for UK depleted uranium (for all waste converted to U_3O_8 powder) basis for DDS3

Waste U_3O_8 and UO_3	Waste Volume	No. of SS 4m Boxes	Mass of SS 4m Boxes	4m Box Volume	4m Box footprint[1]
118,000te	90,700m^3	~4800	~20,160te	~103,200m^3	~11,270m^2
191,160te[2]	**147,050m^3**	**~7780**	**~32,680te**	**~167,270m^3**	**~18,260m^2**
212,400t	163,390m^3	~8640	~36,310te	~185,870m^3	~20,280m^2

Note 1) The stacking is taken as 5 boxes high from NDA, 2008 for the proposed GDF and the 1.2 factor used for surrounding void. Hence (4800/5)*9.78m^2*1.2 = 11,267m^2. 2) Waste mass of 191,160te used. SS – stainless steel

Table A-19 Estimates of super-compacted 200l drums in ungrouted 4m boxes for UK depleted uranium (for all waste converted to U_3O_8 powder) basis for DDS4

Waste U_3O_8 and UO_3	Waste volume @ 5.5te/m^3	No. of SS 4m boxes	Mass of SS 4m boxes	No. of SS 200l drums	Mass of SS 200l drums	External Volume of 4m boxes	Store footprint area 5 high stack 4m box 9.78m^2 footprint[1]
118,000te	~21,500m^3	~2150	~9030te	~236,000	~4,390te	~46,230m^3	5050m^2
191,160te[2]	**~34,760m^3**	**~3480**	**~14,616te**	**~382,320**	**~7,110te**	**~74,820m^3**	**8170m^2**
212,400te	~38,620m^3	~3860	~16,210te	~424,800	~7900te	~82,990m^3	9060m^2

Note 1) The stacking is taken as 5 boxes high from NDA for the proposed GDF and the 1.2 factor used for surrounding void. Hence (3480/5)*9.78m^2*1.2 = 8168.3m^2. The tare weight of the 4m box is 4.2t (Croft Associates data sheet) and the internal volume is 18.9m^3 (no internal concrete shielding) from Wilson et al 2012. Assuming 0.5t per 200l drum from NDA,2007, Macro-economic study. The 200l drum used in SimaPro assumed a tare weight of 0.0186t. Pscintergroup data shows 200l drums can vary from 16kg, 18kg or 21kg tare weight. Hence the SimaPro weight deemed acceptable. Drum System data show 21kg for 1.2mm thick stainless steel 200litre drum. The total 200l drum mass is about 89% of the total 4m box mass. 2) Waste mass of 191,160te used. SS – stainless steel.

Table A-20 Estimates of 3m^3 drums for UK depleted uranium (for all waste converted to U_3O_8 powder) basis for DDS5

Waste U_3O_8 and UO_3	Waste Volume	No.3m^3 Drums at 6.75t/drum	Mass of 3m^3 drums at 0.63t	Mass of grout at 4.52t/drum	3m^3 drums at 2.6m^3/drum	3m^3 drums footprint[1]
118,000te	90,700m^3	~17,480	~11,010te	~79,010te	~45,450m^3	~6,950m^2
191,160te[2]	**147,050m^3**	**~28,320**	**~17,840te**	**~128,010te**	**~73,630m^3**	**~11,260m^2**
212,400te	163,390m^3	~31,470	~19,820te	~142,240te	~81,820m^3	~12,520m^2

Note 1) The stacking is taken as 7 drums high from NDA for the proposed GDF and the 1.2 factor used for surrounding void. 2) Waste mass of 191,160te used. The drums are stainless steel.

The DDS2 HHISO/Liner disposal scenario assumed a waste volume of 191,160te consistent with the other deep disposal scenarios. It also assumed 5400 HHISOs with a carbon steel mass of ~17,000te and titanium mass of ~5,000te and cement grout of 120,000te. The waste volume was 5400x19.5m^3 = 105,300m^3.

Table A-21 Estimate of overpacked depleted uranium in U_3O_8 and UO_3 waste containers, the bases of DDS6 and Shallow Disposal Scenario 1 and 2 (SDS1 and SDS2)

Waste	Waste Mass (te)	Storage Containers	Disposal Containers	Disposal Container Volume (m^3)	Disposal Container Mass (te)	Grout Mass (te)
Urenco UK Ltd U_3O_8	118,000	~11,240 DV70 boxes[1]	~5620 4m stainless steel boxes	~120,800	~23,610	116,070
Capenhurst Nuclear Services U_3O_8	29,500	~2810 DV70 boxes	~1410 4m stainless steel boxes	~30,210	~5900	29,020
Capenhurst Nuclear Services UO_3	30,000	~60,000 of 200l drums[1]	~1670 4m stainless steel boxes	~35,840	~7000	~34,430
Miscellaneous UO_3	~2,000	~4,000 of 200l drums	~112 4m stainless steel boxes	~2,410	~470	~2,310
THORP Product Uranium UO_3	5,000	~25,000 of 50l kegs[1]	~390 Special FHISO	~14,980	~2070 including internal attachments	Not estimated[2]

Note 1) It was assumed 2 of DV70 boxes overpacked in each 4m stainless steel box, 36 of 200l drums overpacked in each 4m stainless steel box and 64 of 50l kegs overpacked in each specially design FHISO. 2) The cement grout for the FHISOs was not estimated, however it was expected to be small given only 390 FHISOs will 64 50l kegs and internal shelving and attachments.

DDS6 assumed that the overpacked depleted uranium was disposed as HAW to a future GDF in predominately 4m stainless steel boxes. SDS1 assumed the same packaging was used for disposal as LLW to a potential near surface disposal facility. SDS2 assumed the same disposal option as SDS1 but using carbon steel 4m boxes rather than stainless steel boxes, hence there may be a small disposal container mass difference. However, it presumed that the carbon steel boxes would be thicker than the stainless steel versions, but would retain the same external volume.

B Appendix of metal price data

This appendix presents the graphs that formed the basis of metals aluminium, copper, lead, nickel, steel and zinc prices shown in Table 5-1 of the thesis. The graphs show the volatile variation in metal prices with time. The values in Table 5-1 were the average price from each graph, converted from United States dollars (USD) to pounds Sterling using historical conversions where necessary and assumed to be for the year in which the graphs was produced. The average price for that year was then inflated to 2013 levels using a standard annual inflation rate from the Bank of England. Clearly, given the volatility of the prices within each year this is a rather crude approach and gives only an indicative price.

The graphs of metal prices presented here are:

- Figure B-1 Aluminium USD Weekly Spot Prices, London Metal Exchange (LME), from Bloomberg Finance at 13 July 2010.
- Figure B-2 Copper Grade A prices £/metric tonne (MT), LME, from Thomson Reuters obtained in 2010.
- Figure B-3 Copper USD Weekly Spot Prices, LME, from Bloomberg Finance at 13th July 2010
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- Figure B-4 Lead USD Weekly Spot Prices, LME, from Bloomberg Finance at 13th July 2010
- Figure B-5 Nickel USD Weekly Spot Prices, LME, from Bloomberg Finance at 13th July 2010
- Figure B-6 Steel, $/MT Monthly Prices, LME, from MEPS (International) Ltd obtained in 2010.
- Figure B-7 Zinc USD Weekly Spot Prices, LME, from Bloomberg Finance at 13th July 2010

Figure B-1 Aluminium USD Weekly Spot Prices, London Metal Exchange (LME), from Bloomberg Finance at 13 July 2010.

Figure B-2 Copper Grade A prices £/metric tonne (MT), LME, from Thomson Reuters obtained in 2010.

The prices in Figure B-2 compare reasonably well with the copper price trend presented in Figure B-3.

Figure B-3 Copper USD Weekly Spot Prices, LME, from Bloomberg Finance at 13th July 2010

Figure B-4 Lead USD Weekly Spot Prices, LME, from Bloomberg Finance at 13th July 2010

Figure B-5 Nickel USD Weekly Spot Prices, LME, from Bloomberg Finance at 13th July 2010

Figure B-6 Steel, $/MT Monthly Prices, LME, from MEPS (International) Ltd obtained in 2010.

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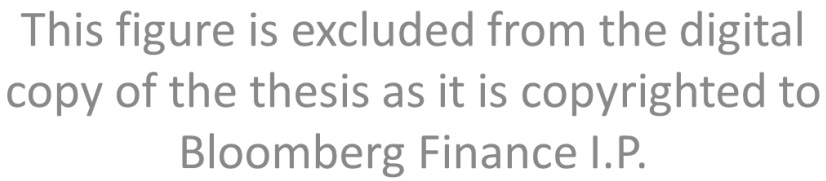

Figure B-7 Zinc USD Weekly Spot Prices, LME, from Bloomberg Finance at 13th July 2010

C Appendix of flowcharts and process images leading to the end-of-life conceptual model for radioactive metals treatment and disposal

This appendix presents the flowcharts and process images leading to the construction of the final end-of-life conceptual model for radioactive metals presented in Figure 6-2. The flowcharts and process images are: metals overview flowchart (Figure C-1), radioactive waste flowchart (Figure C-2), waste metals LCA flowchart (Figure C-3), Simplified waste metal generic flowchart (Figure C-4) generated in late 2010 and early 2011.

The initial metals overview flowchart (Figure C-1) was too general. It was therefore decided to narrow the modelling to radioactive waste metals only (Figure C-2) to decide what processes may be important However, this figure was also too general and a more detailed waste metals life cycle assessment (LCA) flowchart was produce (Figure C-3). This was based on the standard structure of waste generation, pre-treatment of the waste, main treatment, conditioning, storage, retrieval, further treatment to allow recycling or reuse of the radioactive metals and finally disposal. This was too detailed as it encompassed all the treatment for Sweden, Germany and the USA resulting in about 75 process steps and would have to be extended further to include the treatment in France and the Russian Federation for completeness. Hence it was decided to produce a simplified waste metal generic flowchart (Figure C-4).

Studsvik UK Ltd had kindly supplied a number of images of their treatment process at Nykoping (Rossiter 2007); containerised processing (Figure C-5), bulk processing (Figure C-6) and lead processing (Figure C-7). Hence it was decided to simply generic flowchart further to represent the Studsvik processes leading to the final end-of-life conceptual model in Figure 6-2.

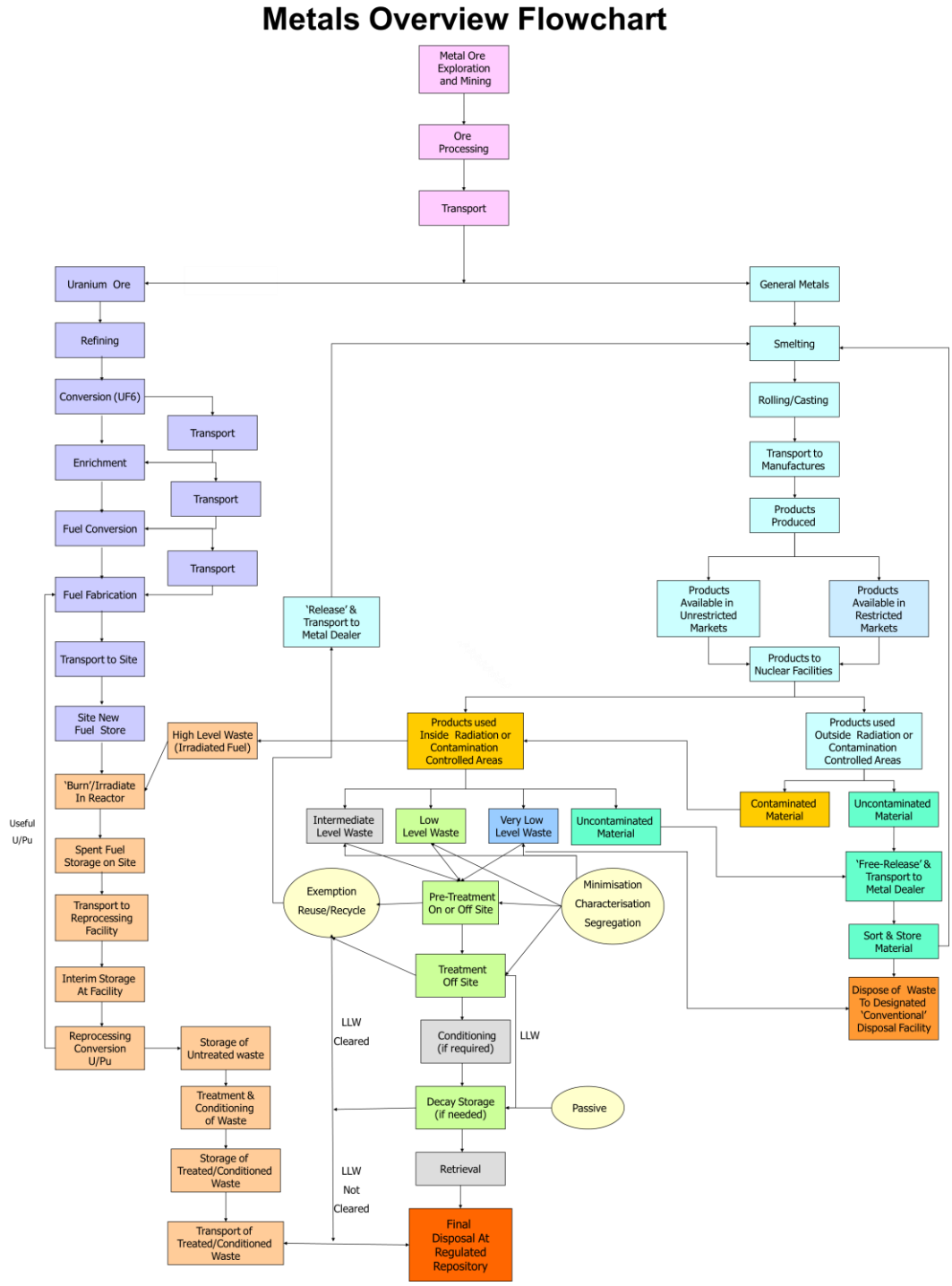

Figure C-1 UK metals overview flowchart generated 2010

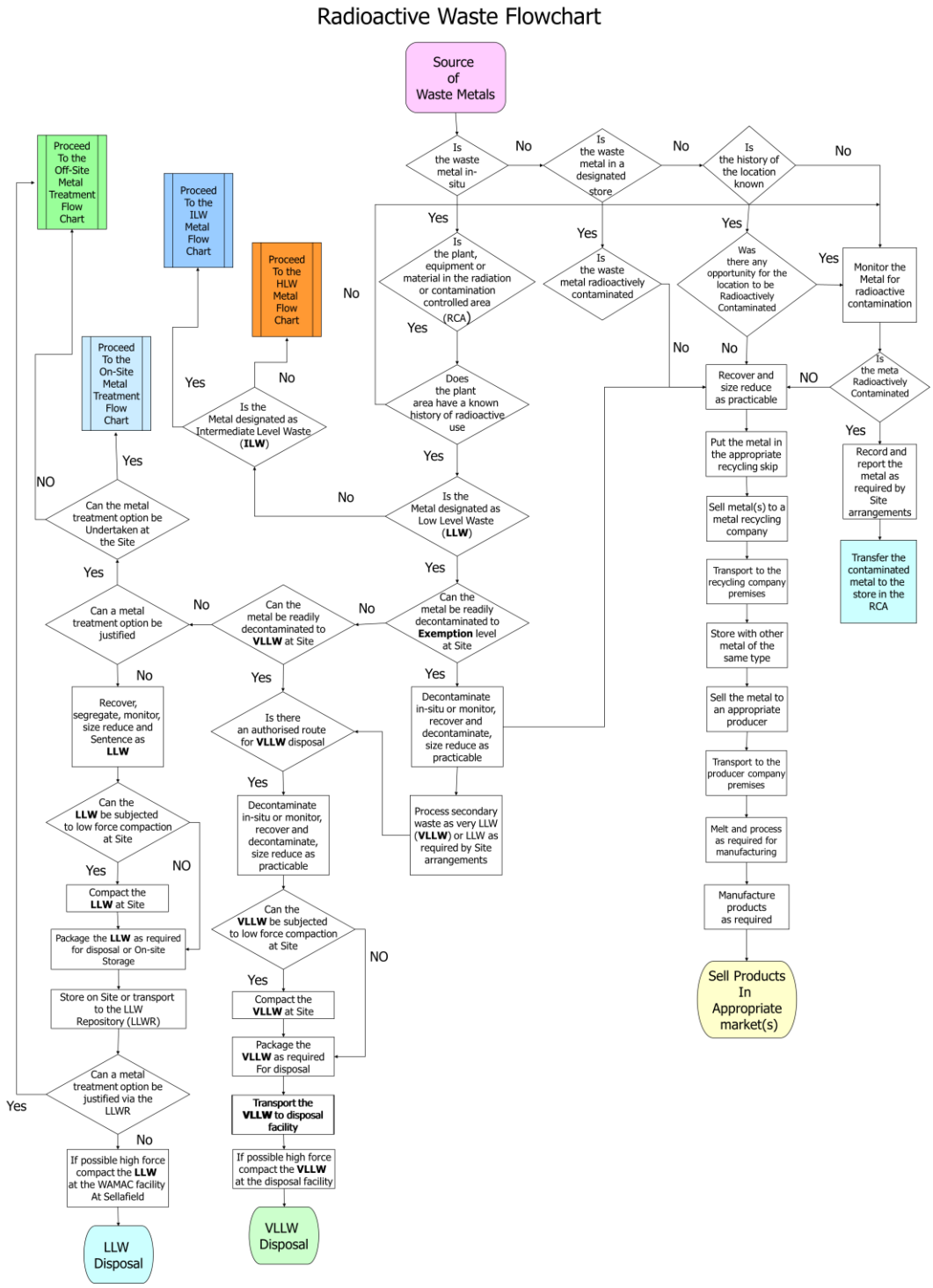

Figure C-2 UK Radioactive waste flowchart generated 2010

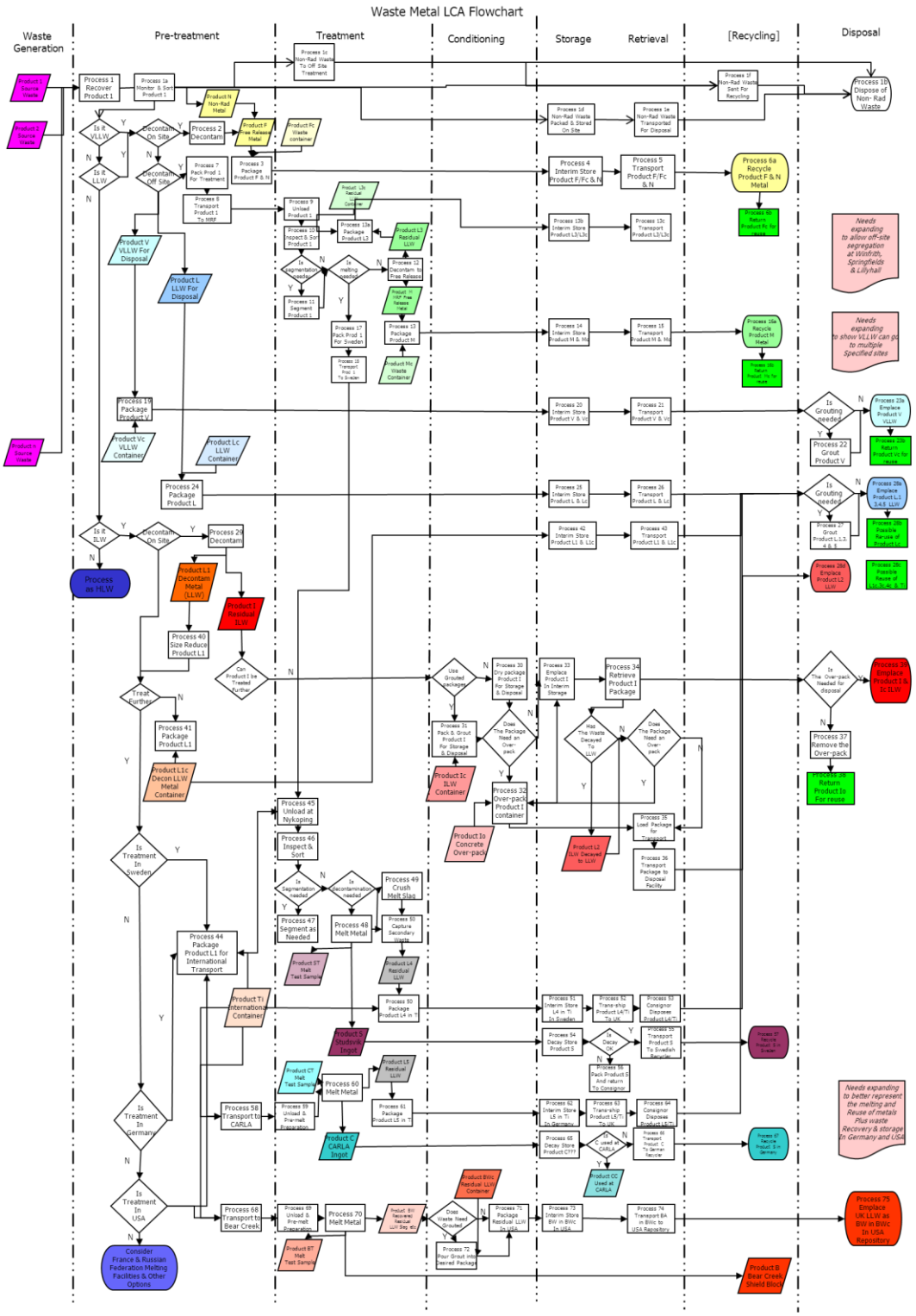

Figure C-3 Waste metal LCA flowchart generated 2010

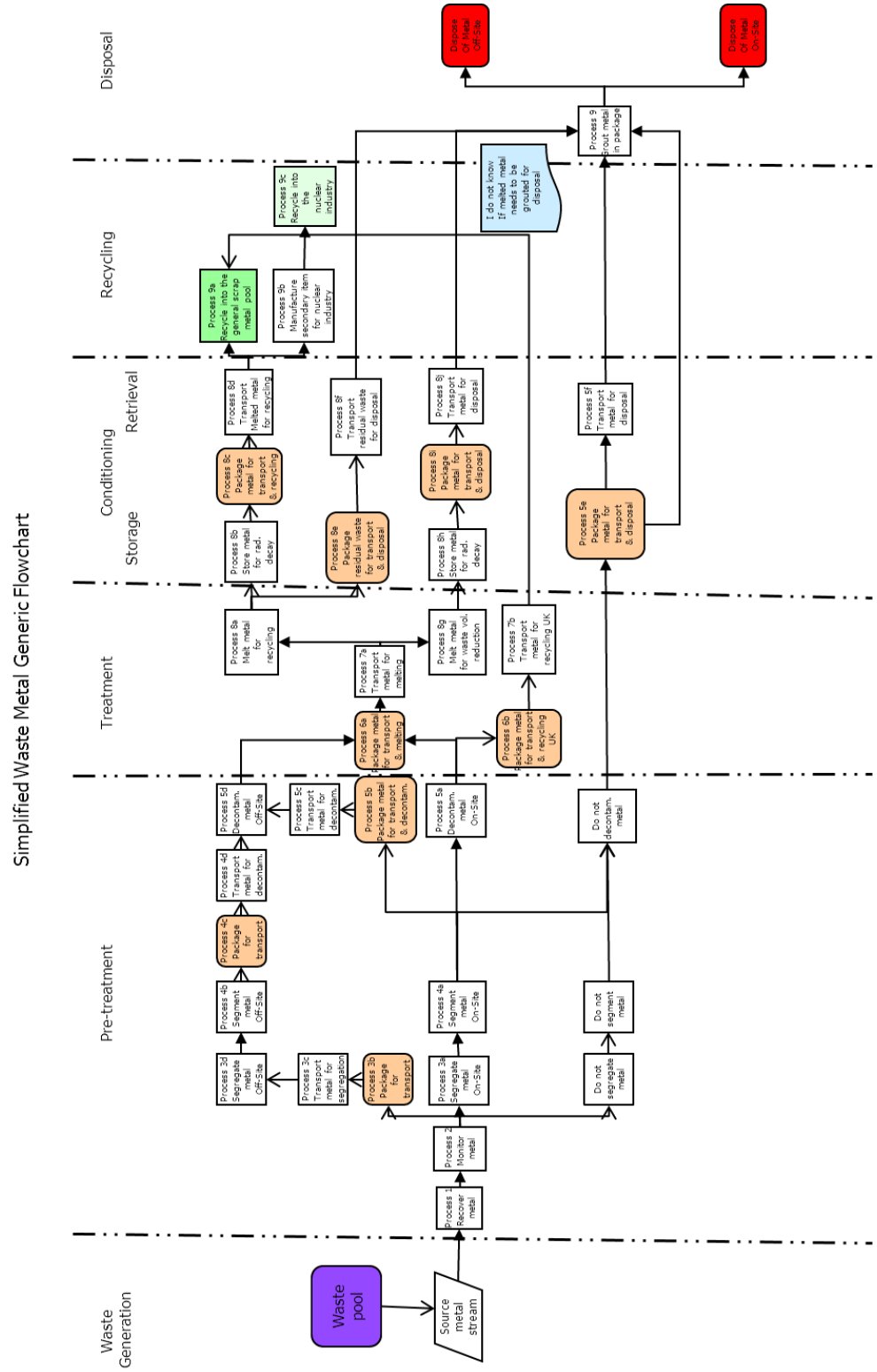

Figure C-4 Simplified waste metal generic flowchart generated 2011

Figure C-5 Studsvik containerised scrap processing at Nykoping (reproduced from pers. comm. Rossiter 2007)

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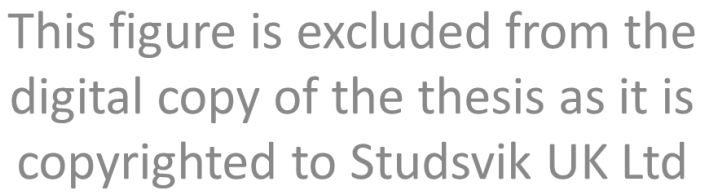

Figure C-6 Studsvik bulk scrap processing at Nykoping (reproduced from pers.comm. Rossiter 2007)

Figure C-7 Studsvik lead scrap processing at Nykoping (reproduced from pers. comm. Rossiter 2007)

D Appendix of Microsoft Excel spreadsheets supporting the thesis

This appendix presents index tables for the initial development stage data and results spreadsheet and main research data and results with tables and graphs supporting the thesis.

D.1 Initial development stage data and results

Table D-1 presents an index of the worksheet contents for the initial development stage data and results in the spreadsheet 'Basic Data and Development Results.xls' in the enclosed compact disc.

Table D-1 Index of worksheet contents for initial development stage data and results at 2015

Worksheet Title	Worksheet Contents	Comments
Eco indicator Points	Materials data from Eco-Indicator 99 Manual for Designers (VROM 2000)	Used for initial comparison with SimaPro results. Tables only.
EI99 Simple Calculations	Basic Data – WAGR Boilers, HHISO, Grout, Distances and Recycling Direct Disposal – 4 WAGR Boilers Packaged Disposal – 4 WAGR Boilers Recycling – 4 WAGR Boilers	Used for initial comparison with SimaPro results. Tables only.
Rad Metals for SimaPro	2010 UKRWI Data 2001 UKRWI Data for Specific Metals Springfields VLLW from 2010 UKRWI SimaPro Nuclear Waste Metals UK Scrap Metal Prices at 2015	Basic metals data to use in SimaPro models. Scrap prices for inventory metals. Tables only
Packages	ILW Package Data LLW Package Data 2010 UKRWI ILW Packages for Total Waste 2010 UKRWI LLW Packages for Total Waste 2010 UKRWI VLLW Packages for Total Waste Empty Container Eco Indicator 99 results August 2013 Container Variables	Basic data to allow some estimates of packaging impacts. Tables only.
Boiler Data	2010 UKRWI , boiler data per Magnox stations and WAGR Baseline Berkeley boiler disposal costs Amended Berkeley boiler disposal costs Baseline WAGR boiler disposal costs Amended WAGR boiler disposal costs	Used for initial comparison with SimaPro results. Tables only.
Boilers Disposal & Recycling	PRODUCT – Berkeley boilers USE - Berkeley boilers E.O.L. – Berkeley boilers DISPOSAL – Berkeley boilers RECYCLING – Berkeley boilers PRODUCT – WAGR boilers USE - WAGR boilers	Used for initial comparison with SimaPro results. Tables only.

Worksheet Title	Worksheet Contents	Comments
	E.O.L. – WAGR boilers DISPOSAL – WAGR boilers RECYCLING - WAGR boilers	
All Wastes	Production ILW Packages LLW Packages VLLW Packages USE DISPOSAL – metals Embodied Energy and CO2 Footprint for Recycling (for all metals in inventory) Embodied Energy and CO2 Footprint for Primary Production (for all metals in 2010 inventory)	Early spreadsheet estimates. Tables and graphs.
Metal Wastes	Production ILW Packages LLW Packages VLLW Packages USE DISPOSAL – (for all metals in 2010 inventory)	Early spreadsheet estimates. Tables and graphs.
Reactor Data	2010 UKRWI Power Station Reactor Data Magnox Power Stations EDF AGR and PWR Power Stations Packages Waste Data for Reactor Wastes Magnox Power Stations EDF AGR and PWR Power Stations	Early spreadsheet estimates. Tables only.
Radioactivity	WAGR Radioactive Fingerprint for LLW (2010 UKRWI) Berkeley Radioactive Fingerprint for LLW (2010 UKRWI) Berkeley Boiler Radioactive Decay Inventory Berkeley Gas Duct Radioactive Decay Inventory	Record of basic radioactivity data and graphs of decay patterns.
Early SimaPro Results	Tests of HHISO Transport for WAGR Boiler Packaged Disposal Comparison of Total Environmental Impact by Cultural Perspective Without Avoided Metal in Recycling. Comparison of Total Environmental Impact by Cultural Perspective With Avoided Metal in Recycling. Comparison of Total Characteristic Results for Direct Disposal for all Cultural Perspectives. Comparison of Total Damage Assessment Results for Direct Disposal for all Cultural Perspectives. Comparison of Total Normalised Results for Direct Disposal for all Cultural Perspectives. Comparison of Total Weighted Results for Direct Disposal for all Cultural Perspectives. Comparison of Total Weighted Results for Bulk Recycling (excluding Avoided Metal Benefits) for all Cultural Perspectives. Comparison of Total Weighted Results for Bulk Recycling (including Avoided Metal Benefits) for all Cultural Perspectives. Comparison of Net Life Cycle Impact for EI99 and ReCiPe (including Avoided Metal Benefits). Comparison of Net Life Cycle Impact for EI99 and ReCiPe (excluding Avoided Metal Benefits).	WAGR Boiler disposition options spreadsheet calculations using SimaPro data. Tables and graphs.

Worksheet Title	Worksheet Contents	Comments
	Comparison of ReCiPe Impact Category Results (including Avoided Metal Benefits). Comparison of ReCiPe Impact Category Results (excluding Avoided Metal Benefits). Comparison of Eco Indicator 99 Impact Category Results (excluding Avoided Metal Benefits). Comparison of Material and Process Impacts for 4 Reference Disposition Scenarios for EI99 and ReCiPe (including Avoided Metal Benefits). Reference Disposition Scenarios for EI99 and ReCiPe (excluding Avoided Metal Benefits).	
International Distances	For: HHISO (out), WAGR Boilers, HHISO (back), Melting (using country specific medium voltage electricity plus imports) and Waste (back). Countries: France, Germany, USA and Russian Federation	Germany used as proxy for Russian Federation. Tables only.

AGR – Advanced Gas-cooled Reactor, EDF – Electricite De France, EI99 – Eco Indicator 99 life cycle impacts assessment method, PWR – Pressurised Water Reactor, ReCiPe – life cycle impact assessment method, UKRWI – UK Radioactive Waste Inventory, WAGR – Windscale AGR

D.2 Main Research Data and Results

Table D-2 presents an index of the worksheet contents for the data and results in the spreadsheet 'Main Research Data and Results.xlsx' in the enclosed compact disc.

Table D-2 Index of worksheet contents for main research data and results at 2015

Worksheet Title	Worksheet Contents	Comments
General Metals	UK Steel Market Volume (Production Million tonne). Demand Million tonne and Import Share. Radioactive waste from all sources (HLW, ILW, LLW). Total Waste at 2007. Low Level Waste Volume. Total LLW and ILW Metal Masses. LLWR Consignments (1997-2015). All Waste Volumes (2010 UKRWI). LLW Arising Estimates. ILW Arising Estimates. Waste Volume (m^3). Nieves et al 1995/ Steel and Iron Cleanable Nieves et al 1995/ Steel and Iron Non-Active (Suspect Activity). Nieves et al 1995/ Steel and Iron Activated. All Waste Volumes (2013 UKRWI)	Data for general UK metals and waste metals. Tables and graphs.
UK Distances	Distance (in miles) between UK nuclear facilities and VLLW, LLW and potential geological GDF	Input data for SimaPro. Table only.
Scottish Distances	Distance (in miles) between each Scottish nuclear facility.	Potential input data for SimaPro. Table only.

Worksheet Title	Worksheet Contents	Comments
Diesel Costs and Dashboard Data	UK Diesel Costs (Based on AA monthly fuel price reports). LLWR Metallic Waste Treatment Costs (Dashboard August 2012). Dashboard monthly Results: 2011/12, 2012/13, 2013/14. Annual Recycling Rates: Sellafield, Research Reactor Sites, Magnox LLWR (2012 – 2017). Cost Comparison: Sellafield, Research Reactor Sites, Magnox LLWR (2012 – 2017). Savings: Sellafield, Research Reactor Sites, Magnox LLWR (2012 – 2017). Example of treatment of 779te of Magnox metals JWMP Benefit. Example of disposal of 779te of Magnox metals JWMP Benefit. Metal treatment cost 5 year increase (2012 – 2017). Metal disposal cost 5 year increase (2012 – 2017).	For transport costs and LLW Repository Ltd Monthly LLW metrics
Waste by Sites	UKWRI Site Waste and Metal Volume up to 2120. Used Fuel Skip Estimates. UKRWI Metals with less than 200Bq/g and adjustments. tkm for Total LLW metals to 2120 (mass weighted average transport distance). tkm of LA-LLW metals to 2026.	To allow for mas weighted average transport distance to be estimated and investigate used fuel skip treatment and disposal if needed.
Activated Metals	EDF nuclear power stations Magnox nuclear power stations	To estimate the amount of activated metals.
WAGR Steels Results	Comparison of steels for WAGR Boilers Direct Disposal (including boiler steel impacts). Comparison of steels for WAGR Boilers Packaged Disposal (including boiler steel impacts). Comparison of steels for WAGR Boilers Bulk Recycling (including boiler steel impacts). Comparison of steels for WAGR Boilers Containerised Recycling (including boiler steel impacts). Combined comparison of Disposal and Recycling Low Alloyed Steel (including boiler steel impacts). Combined comparison of Disposal and Recycling Unalloyed Steel (including boiler steel impacts). Combined comparison of all 4 disposition options for low and un- alloyed steel (including boiler steel impacts). SimaPro Life Cycle Normalised Results for low alloyed and unalloyed steel (including boiler steel impacts). SimaPro Life Cycle Weighted Results for low alloyed and unalloyed steel (including boiler steel impacts). Total Life Cycle Weighted Results for L.A. and U.A steel with uniform Weighting factors (including boiler steel impacts). Total Life Cycle Results for Unalloyed Steel with different weighting (including boiler steel).	Early development tests investigating the impacts of unalloyed and low alloyed boiler steels and including the boiler steel impacts as a proxy for the production and use of the boilers. This was later dropped to exclude the rough boiler steel impacts which masked the treatment process impacts. Tables and graphs.

Worksheet Title	Worksheet Contents	Comments
WAGR Container Results	Comparison of HHISOs from different source locations. Comparison of HHISO Normalised Impact Category Results. Comparison of HHISO Standard Weighted Impact Category Results. Comparison of Number of HHISOs (35, 48 and 74 HHISOs). Comparison of Normalised Impact Category Results for Different No. of HHISOs. Comparison of Standard Weighted Impact Category Results for Different No. of HHISOs. Comparison of HHISO Production Impacts. Comparison of Normalised Impact Category Results for Different HHISO Production Comparison of Standard Weighted Impact Category Results for Different Production.	Development calculations for WAGR HHISO options. Table and graphs.
WAGR Disposal Results	Comparison of LLW Disposal Volume Results. Comparison of LLW Disposal Volume Results Weighted Impact Category Results. Comparison of Storage and Disposal Facility Impacts. Comparison of Storage and Disposal Facility Normalised Impact Category Results. Comparison of Storage and Disposal Facility Weighted Impact Category Results. Comparison of Boiler Internal Volume Grouting. Comparison of Boiler Internal Volume Grouting Normalised Impact Category Results. Comparison of Boiler Internal Volume Grouting Weighted Impact Category Results. Comparison of Boiler Backfill Impact. Comparison of Boiler Backfill Normalised Impact Category Results. Comparison of Boiler Backfill Weighted Impact Category Results. Comparison of Disposal Facility Impacts. Comparison of Disposal Facility Normalised Impact Category Results. Comparison of Disposal Facility Weighted Impact Category Results. Comparison of Different Backfill Materials. Comparison of Different Backfill Materials Normalised Impact Category Results. Comparison of Different Backfill Materials Weighted Impact Category Results. Comparison of Different Cement Backfill. Comparison of Different Cement Backfill Normalised Impact Category Results. Comparison of Different Cement Backfill Weighted Impact Category Results.	Investigation of variation of disposal impact results for different materials. Tables and graphs.
WAGR Reference Results	Comparison of Material and Process Impacts of 4 Reference Disposition Scenarios. Comparison of Normalised Impact Category Results of 4 Reference Disposition Scenarios. Comparison of Weighted Impact Category Results of 4 Reference Disposition Scenarios. WAGR Basic Cost Data.	Reference results to include in thesis. Tables and graphs.

Worksheet Title	Worksheet Contents	Comments
	WAGR Boiler Disposal Costs.	
WAGR Other International Results	Comparison of Material and Process Impacts for all International Facilities With Avoided Metals. Comparison of Normalised Impact Category Results for all International Facilities With Avoided Metals. Comparison of Weighted Impact Category Results for all International Facilities With Avoided Metals. Comparison of Material and Process Impacts With and Without Recycling Avoided Metals Sweden. Comparison of Normalised Impact Category Results With and Without Recycling Avoided Metals Sweden. Comparison of Weighted Impact Category Results With and Without Recycling Avoided Metals Sweden. Comparison of Material and Process Impacts for International Facilities Without Avoided Metals. Comparison of Normalised Impact Category for International Facilities Without Avoided Metals. Comparison of Weighted Impact Category for International Facilities Without Avoided Metals. Comparison of Material and Process Impacts With Recycling Avoided Metals Sweden and UK. Comparison of Material and Process Impacts Without Recycling Avoided Metals Sweden and UK. Comparison of Normalised Impact Category Results With Recycling Avoided Metals Sweden and UK. Comparison of Normalised Impact Category Results With Recycling Avoided Metals Sweden and UK. Comparison of Normalised Impact Category Results Without Recycling Avoided Metals Sweden and UK. Comparison of Weighted Impact Category Results With Recycling Avoided Metals Sweden and UK. Comparison of Weighted Impact Category Results Without Recycling Avoided Metals Sweden and UK.	Results of variations of impacts for Swedish, International and UK facilities. Tables and graphs.
WAGR Recycling Variations	Comparison of Material and Process Impacts for Containerised Recycling with Different Avoided Metals. Comparison of Normalised Impact Category Results for Containerised Recycling with Different Avoided Metals. Comparison of Weighted Impact Category Results for Containerised Recycling with Different Avoided Metals. Comparison of Material and Process Impacts for Bulk Recycling with Different Recycling Percentages. Comparison of Normalised Impact Category Results for Bulk Recycling with Different Recycling Percentages. Comparison of Weighted Impact Category Results for Bulk Recycling with Different Recycling Percentages. Comparison of Material and Process Impacts for Bulk Recycling with Double Electrical Load for Melting. Comparison of Normalised Impact Category Results for Bulk Recycling with Double Electrical Melting Load. Comparison of Weighted Impact Category Results for Bulk Recycling with Double Electrical Melting Load. Comparison of Material and Process Impacts for Containerised Recycling with Double Electrical Load for Melting.	These are the basis of the sensitivity and uncertainty analysis. Table and graphs. Note, the German medium voltage with imports was used for the Russian Federation hence the results are not presented in the worksheet.

Worksheet Title	Worksheet Contents	Comments
	Comparison of Normalised Impact Category Results for Containerised Recycling with Double Electrical Melting Load. Comparison of Weighted Impact Category Results for Containerised Recycling with Double Electrical Melting Load. Comparison of Material and Process Impacts for Packaged Disposal with Quadrupled Size Reduction. Comparison of Normalised Impact Category Results for Packaged Disposal with Quadrupled Size Reduction. Comparison of Weighted Impact Category Results for Packaged Disposal with Quadrupled Size Reduction. Comparison of Material and Process Impacts for Packaged Disposal with Quadrupled Decontamination. Comparison of Normalised Impact Category Results for Packaged Disposal with Quadrupled Decontamination. Comparison of Weighted Impact Category Results for Packaged Disposal with Quadrupled Decontamination. Comparison of Material and Process Impacts for Containerised Recycling with Quadrupled Size Reduction and Decontamination. Comparison of Normalised Impact Category Results for Containerised Recycling with Quadrupled Size Reduction and Decontamination. Comparison of Weighted Impact Category Results for Containerised Recycling with Quadrupled Size Reduction and Decontamination. Comparison of Material and Process Impacts for Containerised Recycling with Quadrupled Slag and Slag Crushing. Comparison of Normalised Impact Category Results for Containerised Recycling with Quadrupled Slag and Slag Crushing. Comparison of Weighted Impact Category Results for Containerised Recycling with Quadrupled Slag and Slag Crushing. Comparison of Material and Process Impacts for Bulk Recycling with Melting Voltage Variation. Comparison of Normalised Impact Category Results for Bulk Recycling with Melting Voltage Variation. Comparison of Weighted Impact Category Results for Bulk Recycling with Melting Voltage Variation. Country Medium Voltage Impacts (for France, Germany, Sweden, the UK and the USA).	
WAGR Transport	Comparison of Material and Process Impact of Specialised Transport for Direct Disposal. Comparison of Normalised Impacts of Specialised Transport for Direct Disposal. Comparison of Weighted Impacts of Specialised Transport for Direct Disposal. Comparison of Material and Process Impact of Specialised Transport for Bulk Recycling Including Avoided Metals. Comparison of Normalised Impacts of Specialised Transport for Bulk Recycling Including Avoided Metals.	Investigation of specialised transport impacts. Tables and graphs.

Worksheet Title	Worksheet Contents	Comments
	Comparison of Weighted Impacts of Specialised Transport for Bulk Recycling Including Avoided Metals. Variation of Transport Impacts with Distance. Comparison of Material and Process Impact of Specialised Transport for Bulk Recycling Excluding Avoided Metals. Comparison of Normalised Impacts of Specialised Transport for Bulk Recycling Excluding Avoided Metals. Comparison of Weighted Impacts of Specialised Transport for Bulk Recycling Excluding Avoided Metals.	
Inventory Study Results	Comparison of Total Worst and Best Case Disposal Impact Category Results (i.e. Baseline and Improved Disposal). Comparison of Total Worst and Best Case Disposal Overall Results (i.e. Baseline and Improved Disposal). Comparison of Total Worst and Best Case VLLW Disposal Results (i.e. Baseline and Improved Disposal). Comparison of Total Worst and Best Case ILW Disposal Results (i.e. Baseline and Improved Disposal). Comparison of Total Worst and Best Case Net VLLW + LLW Disposal Results (i.e. Baseline and Improved Disposal). Comparison of Total Worst and Best Case LLW Disposal Results (i.e. Baseline and Improved Disposal). Planning Norm Total LC Impact (for all specified ILW, LLW and VLLW metals) Packaging and Disposal Impacts for Baseline Disposal. Comparison of Packaging and Disposal Impacts of Baseline and Improved Disposal (as % TBI). Comparison of Packaging and Disposal Impacts for Baseline Disposal and International Treatment With and Without Recycling (as %TBI). Comparison of Packaging and Disposal Impacts for Baseline Disposal and Improved Disposal and International Treatment With and Without Recycling (as %TBI). Comparison of Impact Category Results for Baseline Disposal and International Treatment With and Without Recycling (as %TBI). Comparison of Impact Category Results for Baseline and Improved Disposal and International Treatment With and Without Recycling (as %TBI).	
Inventory Study Costs	General Cost Assumptions from USDOE 2001, Exhibit 8-13, p82. Basic Cost Data. Worst and Best Case Disposal Costs (i.e. Baseline and Improved Disposal). Magnox and LLWR JWMP 2013 Treatment Costs for 779te of Metals. Magnox and LLWR JWMP 2013 Disposal Costs for 779te of Metals. RSRL Disposal and Treatment Costs 2013/14 to 2017/18.	Summary of various cost estimates. Tables and graphs.

Worksheet Title	Worksheet Contents	Comments
	Magnox Disposal and Treatment Costs 2013/14 to 2017/18. Sellafield Disposal and Treatment Costs 2013/14 to 2017/18.	
DNLEU Results	Long Term Interim Storage then Disposal at NSDF (DDS6, SDS1 and SDS2) EI99 H/A Weighted Results. EI99 H/A Normalised Results. Disposal, U_3O_8 in 500i Drums (DDS1) Normalised Results. Weighted Results. Disposal HHISO/Liner (DDS2) Normalised Results. Weighted Results. Disposal, Grouted 4m boxes (DDS3) Normalised Results. Weighted Results. Disposal, 200l drums in 4m boxes (DDS4) Normalised Results. Weighted Results. Disposal, grouted $3m^3$ drums (DDS5) Normalised Results. Weighted Results. Cost £million based on scaling NDA 2007 Macro-economic Study Scenario Costs. Disposal costs inflated from 2007. Cost £million based on NDA 2007 Macro-economic Study Scenario £/m^3 Costs. Disposal costs inflated from 2007.	

AA – Automobile Association, DDS – Deep Disposal Scenario, GDF – Geological Disposal Facility, HLW – High Level Waste, JWMP – Joint Waste Management Plan, LC – Life Cycle, LLW – Low Level Waste, LLWR LLW Repository, NDA – Nuclear Decommissioning Authority, NSDF – Near Surface Disposal Facility, RSRL – Research Sites Restoration Ltd, SDS- Shallow Disposal Scenario, TBI – Total Baseline Impacts, USDOE – United States Department of Energy, VLLW – Very Low Level Waste