



The environmental impacts of recycling portable lithium-ion batteries

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This thesis contains no material which has been accepted for the award of any other degree or diploma in any university. To the best of the author's knowledge, it contains no material previously published or written by another person, except where due reference is made in the text.

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Abstract

The aim of this project was to investigate the different processes that are currently used for recycling lithium-ion batteries, and to compare these processes focusing on the associated environmental impacts. In order to obtain data on the inputs and outputs of different processes, companies who recycle lithium-ion batteries from across the world were identified. Two rounds of surveys were sent to these companies requesting details on the processes used and materials recovered. The survey results were then compared and it was found that mechanical processes recover the largest number of materials. An environmental assessment of the processes was then performed using LCA and it was found that the largest contributors to the environmental impacts were electricity generation, incineration of plastics, and landfilling of residue. In terms of environmental effects, it is suggested that the most beneficial processes are those that utilise low temperatures, and are capable of recovering both plastic and lithium.

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Chapter 1 – Introduction

We as human beings, generate waste in every aspect of our daily lives. We utilise consumer products that produce waste in their manufacture, use and of course, disposal. In 2012 approximately 1.3 billion tonnes of municipal solid waste was produced by urban residents globally. This number is expected to almost double to 2.2 billion tonnes in 2025 (The World Bank, 2012). Waste from electronic and electrical equipment (WEEE) is one of the fastest growing waste streams contributing to this increase, with its volume expected to increase by a third by 2017 (StEP Initiative, 2013). An increasing amount of waste requires more land area for disposal, and adds to the amount of harmful chemicals that eventually re-enter the environment (Tammemagi, 2000). Lithium-ion batteries are the most common battery type used in portable electronic devices and their use is increasing (O'Farrell, 2014). Greatly contributing to this growth is the expected increase in electric vehicles, which will likely contain lithium-ion batteries. These batteries contain lower levels of toxic materials than other battery types (Gold Peak Industries, 2007), and in some countries, are considered suitable for disposal to landfill. In Australia, lithium-ion batteries are classified as hazardous waste under the Hazardous Waste Act 1989 (Australian Government, 2011) and there are laws regarding where and how they are transported.

It has been shown that recycling batteries is beneficial to the environment. Recycling lithium-ion batteries in particular reduces energy consumption (Gaines et al., 2010), reduces greenhouse gas emissions, and results in 51.3% natural resource savings when compared to landfill (Dewulf, 2010). The majority of benefits occur as a result of avoiding virgin materials production (Defra, 2006). However it is not possible to recycle WEEE without causing any environmental impacts (Hischier et al., 2005). Currently in Australia, no recycling of lithium-ion batteries is performed. However, there are multiple processes used globally to collect and recycle lithium-ion batteries.

In Australia, the Australian Battery Recycling Initiative (ABRI), a not-for-profit association, promotes responsible environmental management of batteries at their end-of-life phase. The ABRI is working to achieve increased recovery of all battery types and responsible environmental management in the battery recovery chain. MobileMuster is the Australian mobile phone industry's official product stewardship program to collect end-of-life mobile phones (including batteries) for recycling. These organisations must make decisions regarding where these batteries are recycled. Factors influencing these decisions may be costs, recycling efficiencies and environmental effects. The aim of this project is to investigate the different processes that are currently used for recycling lithium-ion batteries, and to compare these processes, focusing on the associated environmental impacts. The results will be presented in such a way that will aid these decisions, resulting in the best possible environmental outcomes. Overall, my thesis will:

- Outline the current understanding of lithium-ion batteries, including their components, chemical composition and applications
- Describe the current methods for recycling of lithium-ion batteries, and compare these methods in terms of their processes and recovered materials
- Compare the processes available for recycling lithium-ion batteries by evaluating costs, efficiency and environmental effects
- Identify the current trends in lithium-ion battery technology, and determine how these changes will affect current recycling infrastructure

1.1 Scope

This research considers lithium-ion batteries only. Other battery chemistries are excluded due to the wide use and growth of lithium-ion. For the environmental analysis, only portable batteries are considered. However, due to their large expected growth, lithium-ion batteries for electric vehicles are considered when examining current trends. Although there are environmental challenges associated with all stages of the product life cycle, the focus of this research is the end-of-life stage. Finally, the results are developed from the Australian context.

Chapter 2 – Literature review

The term e-waste describes waste from electronic goods such as computers, televisions and mobile phones (Robinson, 2009). In society today, there is a continuous availability of new technology and design that has led to an increasingly early obsolescence of these electronic devices. WEEE is one of the fastest growing waste streams around the world, growing at a rate of 3-5% per annum or approximately three times faster than normal municipal solid waste (Schwarzer et al., 2005). The increasing amount of e-waste presents waste management challenges, particularly relating to environmental impacts. Some forms of e-waste contain hazardous chemicals and require specialised disposal techniques. Furthermore, e-waste contains valuable metals that can be recovered and reused in new products (Robinson, 2009).

2.1 Significance of batteries

Most electronic devices now contain batteries, and as we move towards a society increasingly reliant on these devices, the amount of batteries reaching end-of-life will increase. In 2013, lithium-ion batteries accounted for 24% of sales by weight of portable batteries in Australia, and their use is expected to continue to grow as they enable new applications and replace other chemistries in existing applications (O'Farrell, 2014). A breakdown of growth in different areas is shown in Figure 1 (Global X, 2012), where it is clear that the increase of lithium-ion batteries is significant. Overall, lithium-ion batteries reaching end-of-life are forecast to grow by over 300% from 2012-13 to 2019-20, from 1750 tonnes to 5700 tonnes, and this increase in waste must be dealt with.

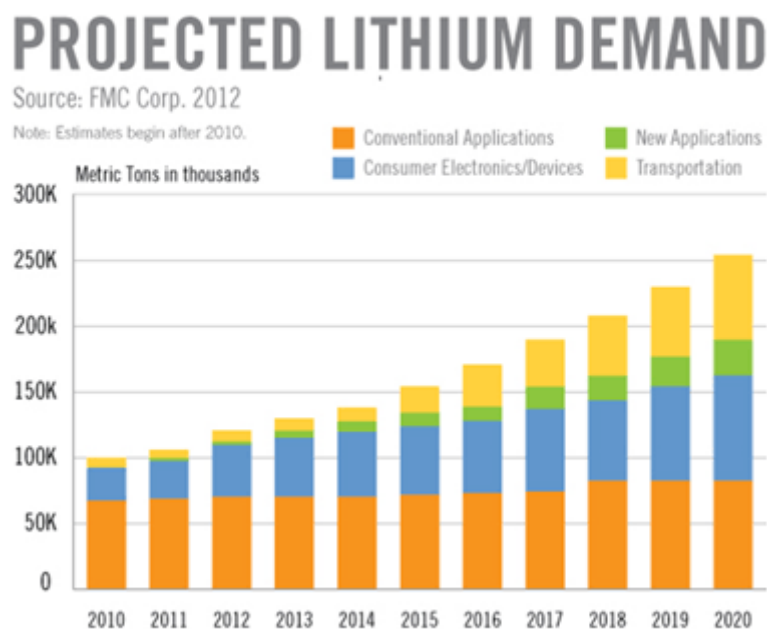


Figure 1 - Projected lithium demand

Growth can also be seen in particular devices. One example of this growth is in use for mobile phones. Figure 2 shows the increase in the presence of lithium-ion batteries in mobile phones collected for recycling between 2005 and 2013 (MobileMuster, 2013).

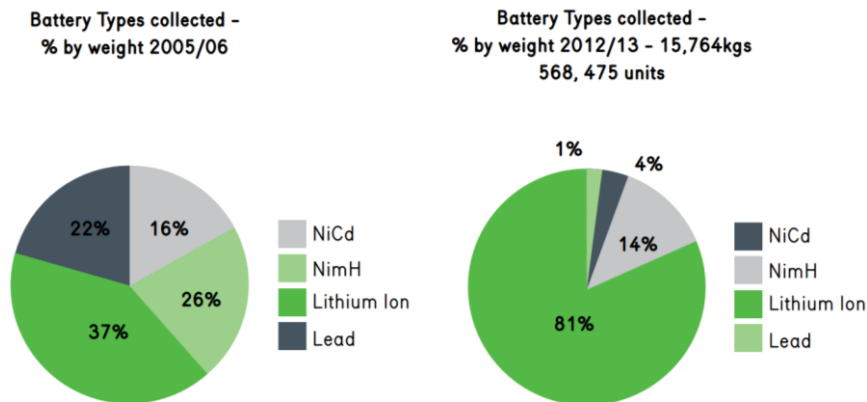


Figure 2 - Mobile phone battery chemistry

End-of-life lithium-ion batteries end up in a variety of places depending on the country, regulations, and participation of consumers. In 2012-13 in Australia, only 1.8% of portable lithium-ion batteries were recycled. The remaining fraction was landfilled. Smaller batteries were much more likely to end up in landfill (O'Farrell, 2014). While allowing e-waste to be disposed of in landfill is not ideal generally, it poses a particular threat when considering the composition of the batteries. Batteries may contain hazardous materials that require specialised facilities to recover. If batteries end up in landfill, they may leach toxic metals such as mercury, lead and cadmium into soil and groundwater; while if they are incinerated, the metals are released into the air and into the ash that is subsequently landfilled. Lithium-ion batteries contain less toxic materials than other battery types (Gold Peak Industries, 2007). However, if they end up in landfill, the materials within the battery can never be recovered. As a result, more raw materials are extracted to manufacture new batteries, using large amounts of energy and creating emissions that are harmful to the environment. Repair and reuse of spent portable batteries is impossible, so the only way of recycling is the recovery of valuable materials (Georgi-Maschler et al., 2012).

2.2 Battery recycling

As an alternative to landfill, battery recycling provides environmental benefits (Bernardes et al., 2004). It enables the recovery of non-renewable resources, reduces the environmental impacts of mining and manufacturing virgin materials, and reduces the environmental impacts of landfill (Lewis, 2010). For lithium-ion batteries in particular, this results in an overall reduction in energy consumption (Gaines et al., 2010), reduces greenhouse gas emissions, and results in 51.3% natural resource savings (Dewulf, 2010). It is also economically viable to recover materials from spent lithium-ion batteries. Xu et al. (2008) mentioned that the valuable materials in lithium-ion batteries are cobalt, copper and nickel, and that recycling of spent lithium-ion batteries may result in economic benefits. This economic drive has been identified in many sources. Kumar (2014) states that the battery recycling market is largely price-driven. Recycling companies operate as a business, aiming to maximise profit from sale of recovered materials.

There are many different techniques that can be used to recycle portable batteries. These can be broken into three general categories: mechanical processes, pyrometallurgical processes, and hydrometallurgical processes. Often a combination of these techniques is used one after the other for the recovery of different materials. These processes are described in further detail below.

2.2.1 Mechanical

Mechanical processes are used for two purposes. The first is to dismantle batteries and liberate the components. Usually, this is done by crushing or shredding. Crushing is often done with a high speed rotating blade or hammer. On impact with the blade, the parts fracture. The fractured parts are rebounded back onto the blade if not a sufficient size, and eventually pass through when the appropriate size has been reached (T. Zhang et al., 2014).

Mechanical processes are also used for separation of crushed components. This is done by sorting materials according to their physical properties. For example, magnetic separation extracts ferrous components from a shredded feed, and air ballistic separation sorts light and heavy materials according to their projectile properties. Sieving can also be used to separate materials by particle size. Separation by density is achieved using vibrating tables and controlled air flow. Mechanical processes such as shredding or crushing must always be used as a pre-treatment for hydrometallurgical processes. Although, it is possible to use entirely mechanical processes to recycle lithium-ion batteries. These processes consume a considerable amount of energy (Al-Thyabat et al., 2013).

2.2.2 Pyrometallurgical

Pyrometallurgy is the recovery of metals using high temperatures (Bernardes et al., 2004). Pyrometallurgy includes a range of thermal treatments such as pyrolysis, smelting, distillation and refining. Pyrolysis is the decomposition of organic material through the application of intense, indirect heat in the absence of oxygen (Ecoreps, n.d.). This process can be used to deactivate batteries and eliminate electrolyte and organic matter such as plastic and paper (SNAM, n.d.). In vacuum pyrolysis, heating occurs in a vacuum to decrease boiling temperature and avoid secondary chemical reactions. Smelting uses heat and chemical reduction to obtain metals, leaving slag and gases behind. Often gas cleaning is used with smelting to ensure harmful chemicals are not released. Distillation treatment can be used to separate metals thermally (SNAM, n.d.). The metals are evaporated at different temperatures and then condensed. Distillation can also be performed using a vacuum. Higher temperatures are not required since the reduction in pressure also reduces the evaporation temperature. Thermal treatment can also be used to refine metals to very high purities by eliminating unwanted materials (SNAM, n.d.).

Due to these high temperature requirements, large amounts of energy are consumed, (Espinosa et al., 2004). Pyrometallurgy can also lead to harmful gas emissions including carbon dioxide and carbon monoxide, dust from scrap metals, sulphur dioxide, and volatile organic compounds (Tes-Amm, 2008), so pyrometallurgical techniques are often associated with high control of air emissions (Bernardes et al., 2004). Lithium and organic compounds are not recoverable by pyrometallurgical processes alone (Al-Thyabat et al., 2013). This is because organic materials such as paper, plastic and the battery electrolyte are burnt, and lithium is always left in the slag. This slag contains metals which, when landfilled, can leach to the environment (Defra, 2006). However, it can be treated hydrometallurgically to recover materials such as lithium.

2.2.3 Hydrometallurgical

Hydrometallurgical processes recover metals using acids or bases to leach metals into a solution, which is then purified to obtain metals. These processes must be preceded by a mechanical process such as crushing or shredding to liberate the materials. For this reason, where processes are referred to as hydrometallurgical, this includes the mechanical pre-treatment. Hydrometallurgical processes are considered suitable for the recovery of metals from lithium-ion batteries because high recovery of metals with good purity is possible, and there are low energy requirements and minimised air emissions (Huang et al., 2009; Jha et al., 2013). Leaching is normally performed with sulphuric acid, hydrochloric acid or nitric acid, and between temperatures of 50-80°C (Jha et al., 2013). The recovery of the metal is done by precipitation with an alkaline solution such as sodium hydroxide or by electrolysis, where a current is passed through the leach solution (Xu et al., 2008). Although leaching has low energy consumption, special equipment may be needed to treat toxic gases such as chlorine, depending on the leaching agent (Xu et al., 2008). Wastewater requiring further treatment is also produced (Tes-Amm, 2008). Hydrometallurgical processes may also be used to extract materials from the slag produced in pyrometallurgical processes. However, the heat treatment reduces the efficiency of the subsequent hydrometallurgy (Al-Thyabat et al., 2013).

2.2.4 Environmental impacts

It is not possible to recycle WEEE without causing any environmental impacts (Hischier et al., 2005). However, it has been confirmed that overall, the recycling of batteries gives an environmental benefit (Bernardes et al., 2004). Depending on the processes used, many different environmentally detrimental effects can be caused. The major disadvantage of pyrometallurgical processes is their association with high atmospheric emissions control, since dioxins, chloride compounds and mercury can be generated in the process (Bernardes et al., 2004). Pyrometallurgical techniques also consume large amounts of energy, since operating temperatures are in the range of 800-1000°C (Espinosa et al., 2004). Mechanical processes may also use large amounts of energy in the form of electricity, and hydrometallurgical processes generate hazardous waste that requires further treatment (Espinosa et al., 2004). Furthermore, no recycling process is capable of recovering all materials, and some waste will always be landfilled (EPA, 2013).

The extent of the environmental effects varies with the types of batteries being recycled, since each battery type uses different materials for the cathode, anode, and electrolyte (Bernardes et al., 2004). Additionally, the environmental effects of recycling one battery type vary with the process used (i.e. mechanical, hydrometallurgical, pyrometallurgical or a combination). Regardless of the recycling method used, batteries must always be transported to a recycling facility. The magnitudes of the environmental effects resulting from transportation are dependent on the location of the recycling facility and the means of transport.

2.2.5 Comparison

A summary of the advantages and disadvantages of the processes described is shown in Table 1 (Gaines, 2011; Riba, 2013; Tes-Amm, 2008; Vadenbo, 2009; Xu et al., 2008).

Table 1 - Comparison between recycling processes

Process	Advantages	Disadvantages
Mechanical	Material composition remains the same	High energy requirements Risk of explosion Requires uniform feed
Hydrometallurgical	Low energy requirements High recovery and purity Minimised emissions	Gas treatment required Wastewater produced Sensitive to process input
Pyrometallurgical	Simple operations Sorting often not necessary Can take any input or mix of inputs in high volume	Lithium cannot be recovered Plastic/paper not recovered High energy requirements High emissions control

2.3 Previous work

Currently, only general comparisons have been made regarding the environmental effects relating to different battery types. Yu et al. (2012), found that the environmental impact of lithium-ion batteries is lower than nickel metal hydride batteries. This particular study focused on the entire life-cycle, and did not indicate what assumptions were made for the end-of-life scenario of the batteries. Similarly, the Environmental Protection Agency (EPA) in America has conducted a life cycle assessment of lithium-ion batteries for vehicles. One finding was that the recycling of lithium-ion batteries has negative environmental impacts when considering the entire life cycle. One limitation of these studies is the exclusion of battery transport in the analysis. Depending on different factors, transport can have a large effect on the environmental impacts. Overall, the environmental impacts of different recycling methods for lithium-ion batteries are not well documented.

General comparisons have also been made between the different recycling processes for lithium-ion batteries. Espinosa et al. (2004) compiled an overview of the current processes for recycling of batteries, including the processes used and materials recovered by eleven different companies. Bernardes et al. (2004) conducted a similar study, however with more detail on collection and sorting. Xu et al. (2008) reviewed the processes and technologies for the recycling of lithium-ion rechargeable batteries, and Al-Thyabat et al. (2013) also conducted a critical review of processing operations, listing the drawbacks of current methods being used. The limitation of all these papers is the lack of a comparison between the technologies. Each process is described, but the information is not presented in such a way that any conclusions can be drawn regarding a preferential process. Another drawback of the current reviews is the focus on laboratory-scale processes still in development. It is unclear whether any of the reviewed processes are currently being used commercially, and if not, whether it is likely they will be used in the future. The studies performed also hold limitations in their scope. No research has been done comparing the environmental effects of the recycling processes alone. Although the effects of a battery's entire life cycle is useful, the results from previous research cannot be an indication of the environmental effects of recycling since a significant proportion of the emissions and waste occur in the battery manufacturing stage (Parsons, 2007). If the effects of specific processes can be compared, then this information can be used by battery collectors to choose the process that is the most beneficial.

2.4 This project

This research seeks to fill the gaps in the current literature. The processes currently being used commercially will be identified, and compared in terms of the techniques used and the materials recovered. The environmental effects relating to these processes will then be investigated and compared if sufficient data is available. This will be done focusing on three environmental impact categories: global warming potential, human toxicity potential and terrestrial ecotoxicity potential. Lastly, the current technological trends relating to lithium-ion batteries will be investigated with a focus on how these trends might affect the current recycling infrastructure. This thesis will investigate the following hypotheses:

1. Current recycling of lithium-ion batteries is driven by financial gains and this will be evident in the materials recovered and the processes used for recovery.
2. The environmental effects associated with mechanical and hydrometallurgical processes will be lower than those associated with pyrometallurgical processes. The literature indicates that high temperatures result in high energy use and increased harmful emissions. Also, recycling will have lower environmental impacts when compared to landfill.
3. Considering the Australian context, the transport associated with recycling will cause significant contributions to the overall environmental impacts, due to the large distances that must be travelled.
4. Lastly, the current technological trends towards changing battery chemistries will mean that legislation is required to ensure an environmental benefit from recycling.

Chapter 3 – Methodology

Different methodologies were applied to different areas of the research.

3.1 Surveys

To compare the different processes currently used for recycling lithium-ion batteries, data from recycling companies was needed. This was an iterative process that involved surveys, email correspondence, and site visits. Initially, several recycling companies were identified through suggestions made by MobileMuster and the ABRI. The preliminary round of surveys requested information regarding the processes used for recycling batteries and the materials recovered. The results of these surveys were compiled and used to construct a secondary survey that requested inventory data (inputs and outputs) associated with the recycling process. Unfortunately many companies were unable to provide detailed data due to privacy of the information requested. A wider range of companies were then included in the analysis and email correspondence was used to obtain a wide representation of the data. An example of an initial survey sent to recyclers can be found in Appendix A, and an example of a secondary survey can be found in Appendix B.

3.2 Life cycle assessment

For the environmental component of analysis, life cycle assessment (LCA) was used. LCA is used to assess the environmental aspects and potential impacts of a product, process, or service over its entire life cycle. LCA was chosen because it provides a means of comparison despite the potential differences in environmental impact of the different battery types and processes. According to ISO 14040 Life cycle assessment – Principles and framework (1998), LCA consists of four stages:

1. **Goal and scope definition** – Describe the system to be studied such as the product, process or activity, determine the goal of the study, and describe the scope including impact assessment methodology and system boundaries.
2. **Life cycle inventory analysis (LCI)** – Identify and quantify relevant inputs and outputs to the system. These may include the use of resources and releases to air, water and land associated with the system.
3. **Life cycle impact assessment (LCIA)** – Associate inventory data with specific environmental impacts. Evaluate the significance of potential environmental impacts using the results of the LCI.
4. **Life cycle interpretation** – Interpret the results of the inventory analysis and impact assessment phases in relation to the objectives of the study. Provide conclusions and recommendations.

A LCI requires a detailed set of data on the inputs and outputs of the system. The surveys did not provide sufficient information to produce a complete LCI. However, some secondary inventory data was available and was analysed using LCIA techniques to create a general comparison between two methods for recycling lithium-ion batteries. For this analysis, the ISO Standards 14040-14043 (1998) for LCA were followed, and LCA software GaBi was utilised where necessary.

Chapter 4 – Background

This chapter provides a description of portable lithium-ion batteries in terms of their use, applications, composition and toxicity. Before investigating the processes used, it is important to understand the product itself. This information will help to determine the reasons why lithium-ion batteries are recycled, how recycling processes work, and why they are used.

4.1 Product description

Lithium-ion batteries consist of a cathode, anode, electrolyte, separator and casing. For portable sizes, a lithium oxide is most commonly used for the cathode, and graphite for the anode. The electrolyte consists most commonly of lithium hexafluorophosphate (LiPF_6) dissolved in an organic solvent such as ethylene carbonate, dimethyl carbonate or diethyl carbonate. A microporous polymer is used for the separator, and a polyvinylidene fluoride (PVDF) binder holds the active materials to the copper and aluminium current collectors (Schmidt, 2003). During charging, lithium ions travel from the cathode to the anode, via the electrolyte and through the separator. The ions are stored in the graphite anode until the battery begins to discharge. When the lithium ions travel back to the cathode during discharge, energy is released (Templeton, 2014). The structure of a prismatic portable lithium-ion battery is shown in Figure 3 (Chat with the designers, 2013).

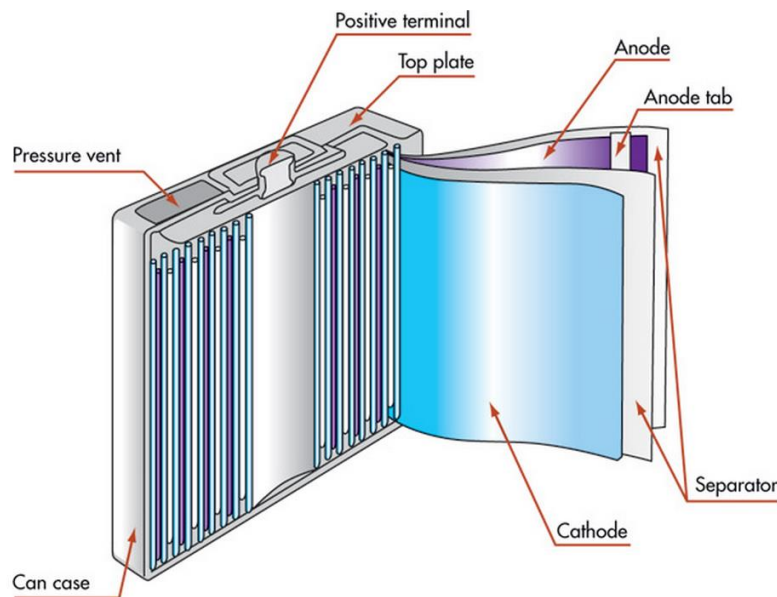


Figure 3 – Portable lithium-ion battery structure

4.2 Applications and types

Lithium-ion batteries are characterised by their high energy density, high efficiency and long life (Scrosati and Garche, 2010). These aspects make them very suitable to power a range of devices, and are commonly used to power portable devices such as mobile phones, laptops, tablets, MP3 players, digital cameras, cordless power tools, and handheld game consoles, although they are also being used in larger applications such as electric vehicles. In 2013, 5 billion lithium-ion batteries were sold globally (Van Noorden, 2014).

Under the name lithium-ion, there are many different forms of batteries. These types most commonly vary with their cathode material. The most common cathode material for portable batteries is lithium cobalt oxide (abbreviated to LCO) because it gives the highest energy density. This material has a high capacity and is usually used for portable electronic device such as mobile phones, laptops, and tablets (Battery University, 2014). Other cathodic materials include lithium manganese oxide (LMO), lithium iron phosphate (LFP), and lithium nickel manganese cobalt oxide (NMC), all three of which are considered safer than LCO batteries and are more commonly used in power tools, electric vehicles, and medical applications. Two other chemistries also available are lithium nickel cobalt aluminium oxide (NCA), and lithium titanate (LTO). NCA batteries are less commonly used in the consumer market due to safety and cost, but are candidates for electric vehicle batteries (Battery University, 2014). Despite the nearly universal use of a graphite anode in lithium-ion batteries (Amine et al., 2014), lithium titanate batteries use lithium titanate as the anode, most commonly with a LMO cathode. These batteries are characterised by their high safety and life span, but improvements are sought in energy density and cost.

4.3 Composition analysis

The composition of batteries varies between sizes, applications, and cathode material. The materials included in different types of batteries have been recorded for different studies, though no single research paper contains data consistent with what is required for this research. Several composition analyses were collated to form a composition breakdown of a typical lithium-ion portable battery. This is shown in Table 2, and has been used for all subsequent analysis requiring composition.

Table 2 – Portable lithium-ion battery composition

Component	Wt. %
LiCoO ₂	27.5
Steel	20.2
Ni	4.3
Cu	9
Al	5.5
Graphite	16
Electrolyte	3.5
Polymer	14

The base composition was taken from a review of lithium-ion recycling processes (Xu et al., 2008), and more detailed composition analyses (Accurec, 2014; Georgi-Maschler et al., 2012) were used to refine the composition. This is not representative of all lithium-ion portable batteries, as the composition may vary. A lithium cobalt oxide cathode material was assumed, and an iron-nickel alloy casing material, although other materials such as aluminium may be used for the casing. The electrolyte is most commonly the lithium salt LiPF₆ dissolved in an organic solvent such as ethylene carbonate, dimethyl carbonate or diethyl carbonate. The compositions of industrial or automotive lithium-ion batteries contain further differences. However, these types of batteries are excluded from the scope of the project.

4.4 Toxicity classification

In general, lithium-ion batteries are considered significantly safer than other battery chemistries such as nickel metal hydride and nickel cadmium, and are not dangerous during use if operated according to manufacturers' instructions. Additionally, since the lithium is in the form of lithium ions, the risks associated with lithium metal are not present. In Australia, waste batteries are classified as hazardous waste, and have regulations involving imports and exports. Furthermore, lithium-ion batteries are classified as Class 9 – Miscellaneous dangerous goods by the United Nations (IATA, 2013). These classifications mainly refer to transport and do not provide any guidelines on treatment of waste batteries. The toxicity of lithium-ion batteries themselves has not been defined clearly in the literature, and there are opposing views as to their safety.

There are different safety issues depending on the individual components of the battery. Lithium cobalt oxide is toxic to the central nervous system in humans and repeated ingestion may damage kidneys (STREM Chemicals, 2011). Graphite is slightly hazardous if it comes in contact with the skin, is ingested, or is inhaled. Copper and steel may cause respiratory irritation if their dust is inhaled, but there is no risk from the aluminium content (BAK, n.d.). If the electrolyte is released, no toxic or poisonous gases are produced. However, the electrolyte is not stable with respect to water and will decompose to make small amounts of skin irritants (Gold Peak Industries, 2007). There is also a fire risk with lithium-ion batteries if a short circuit occurs. The solvents used are organic and will burn if a flame is applied. The hazards presented in safety data sheets for lithium-ion batteries focus on effects to human health. The environmental effects associated with end-of-life lithium-ion batteries are explored further in subsequent sections of this report.

Chapter 5 – Lithium-ion battery recycling processes

In order to determine the environmental effects of battery recycling, it is necessary to first understand what processes are used. There are multiple different techniques for recycling batteries, giving a range of recovered materials and recycling efficiencies. When deciding where batteries are sent for recycling, it is important to look at all options available and understand the differences between them. This chapter will examine where, why and how lithium-ion batteries are currently recycled, and present some general comparisons between the processes used.

5.1 Where lithium-ion batteries are recycled

Currently, there are no facilities in Australia for recycling lithium-ion batteries. A list of prominent battery recycling companies globally was collected from the ABRI, MobileMuster, and additional general research. A list of the companies identified and their locations is shown in Table 3. Company names have been altered for privacy reasons, and the naming convention is representative of the recycling methods used by each company. ‘P’ indicates a pyrometallurgical process, ‘M’ a mechanical process, ‘H’ a hydrometallurgical process, and ‘C’ a combination of pyrometallurgical and hydrometallurgical processes. It should be noted that this is not a complete representation of all the facilities used to recycle batteries worldwide.

Table 3 - List of companies currently recycling lithium-ion batteries

Company name	Country	Continent
Company P1	Germany	Europe
Company P2	France	Europe
Company P3	USA	North America
Company M1	Switzerland	Europe
Company M2	France	Europe
Company M3	Finland	Europe
Company H1	South Korea	Asia
Company H2	USA	North America
Company H3	Singapore	Asia
Company C1	Belgium	Europe

An assessment of the companies that recycle lithium-ion batteries showed that Europe contains the most facilities, with some companies operating in Asia and North America. None of these facilities operate with the sole purpose of recycling lithium-ion batteries. Most companies originally recycled other battery types, and have applied their processes to lithium-ion batteries as they have grown in use. This means that often the processes have not been designed specifically for lithium-ion batteries (Riba, 2013). Furthermore, some companies only ‘recycle’ batteries by consuming them in the smelting process for recovery of cobalt, nickel and copper. Although this ensures some materials are recovered, a dedicated recycling processes is required to recover more materials from lithium-ion batteries (Umicore, 2010).

5.2 Why lithium-ion batteries are recycled

Lithium-ion batteries are recycled for several reasons, the most prominent being the recovery of valuable materials and to adhere to environmental laws. These are further discussed below.

5.2.1 Value of materials

Battery recycling is largely price driven (Kumar, 2014). Recovered materials are sold and profit can be made if the costs of extracting materials are lower than the sale price. Hence materials are only recovered if they are economically viable to recover. Table 4 shows the recoverable metals from lithium-ion batteries, their current price per tonne in Australian dollars, and the associated value per tonne of batteries. The value per tonne of batteries was calculated using the composition determined in section 4.3. The price per tonne of lithium carbonate was used to represent the price of lithium, as it is often found in this form.

Table 4 - Estimated value of materials per tonne of portable lithium-ion batteries

Material	Current price (\$AUD/tonne)	Source	\$AUD available per tonne of batteries
Nickel	18684.00	(LME, 2014)	803.40
Cobalt	37210.00	(LME, 2014)	6160.10
Aluminium	2464.00	(LME, 2014)	135.55
Copper	8168.00	(LME, 2014)	735.10
Steel	567.00	(LME, 2014)	114.60
Lithium carbonate	~7300.00	(Fox-Davies, 2013)	142.90
Manganese (for Mn cathode batteries)	2745.00	(InvestmentMine, 2014)	458.20

The results indicate that some materials are highly valuable and may be worth recovering. The most valuable materials identified were cobalt, nickel and copper, which is consistent with what was found in the literature. It also explains why batteries may be consumed in a smelting process for recovery of these metals.

5.2.2 Regulations

To prevent lithium-ion batteries from ending up in landfill, collection regulations and recycling rates for batteries are mandated in some places. While most countries have regulations regarding imports and exports of hazardous waste through the Basel Convention, the recycling processes themselves are often not the focus of these regulations. For example, in Australia, there are regulations on classifying hazardous waste, transporting hazardous waste, and whether or not waste can be imported and exported. Similar regulations exist in Asia, where in Singapore (a signatory to the Basel Convention), the Hazardous Waste Act for the control of imports, exports, and transit exists (Singapore Government, 2000).

In America, the laws regarding the collection and recycling of batteries vary between states, and only New York and California laws cover lithium-ion batteries (Gaines, 2014). Lithium-ion batteries are classified as miscellaneous hazardous waste meaning there are specific requirements regarding packaging, labelling, and shipping. Most states prohibit the disposal of batteries containing lead and cadmium in landfill, while in only some states all rechargeable batteries are prohibited from being disposed of as municipal waste. In California, retailers who sell small, non-vehicular rechargeable batteries must provide consumers with a free system for returning these batteries for reuse, recycling, or proper disposal (Call2Recycle, n.d.).

Europe is the only region where the laws extend further into recycling. The European Commission (EC) provides legislation aimed at minimising the environmental effects of batteries through the EU Battery Directive. In regards to recycling of batteries, Directive 2006/66/EC (European Parliament, 2013a) 'On batteries and accumulators and waste batteries and accumulators' provides legislation on the recycling of batteries. It defines recycling as the "reprocessing in a production process of waste materials for their original purpose or for other purposes, but excluding energy recovery". It states that for lithium-ion batteries, the recycling to yield similar products or products for other purposes must reach at least 50% by average weight. This target affects the materials that are recovered. Recovering only copper, nickel and cobalt would achieve a recycling efficiency of approximately 30%. To achieve the target efficiency, more materials must be recovered.

5.3 How lithium-ion batteries are recycled

Multiple techniques exist for recycling lithium-ion batteries, and different recycling processes have different environmental outcomes. Generally, battery recycling processes can be broken down into three categories: mechanical, pyrometallurgical and hydrometallurgical processes, as described in the literature review. Usually a combination of several kinds of recycling processes should be implemented to recover all target materials (Xu et al., 2008). This section lists the recoverable materials through recycling, the challenges of recycling and disassembly, and the results from the surveys sent to recycling companies.

5.3.1 Recoverable materials

The materials recovered may vary depending on the cathode material and current prices. In terms of regulations, the EU Battery Directive does not enforce which materials are recovered, only a target percentage of materials recovered. Since there is no obligation to recycle certain materials, it is commonplace for the most valuable materials to be recovered. Assuming the composition determined in section 4.3, the recoverable metals from lithium-ion batteries are copper, nickel, aluminium, lithium, steel, cobalt and manganese. These materials constitute approximately 57% of the battery's composition. In addition to these metals, plastic, carbon, and fluorine may also be recovered, depending on the process. The recovery of pure electrolyte components has not yet been realised but is considered feasible (Georgi-Maschler et al., 2012). However, the electrolyte makes up only 3.5% of the battery by weight and its recovery is not required to meet the EU Battery Directive target of 50% recycling efficiency. This target can be met through the recovery of metals alone.

5.3.2 Challenges of recycling and disassembly

Opening a lithium-ion battery is potentially hazardous, due to the presence of residual lithium atoms at the anode (Sonoc and Jeswiet, 2014), i.e. the battery is not fully discharged. These lithium atoms may react violently with moisture on exposure. Several approaches can be taken to deactivate the battery. The batteries can be crushed in an inert atmosphere, cryogenically cooled by immersion in liquid nitrogen (Georgi-Maschler et al., 2012), pyrolysed by heating to high temperatures (Accurec, 2014; Georgi-Maschler et al., 2012; SNAM, n.d.), or mechanically treated in brine solution (Vadenbo, 2009). There is a larger risk involved if lithium-ion batteries are consumed in recycling processes they were not intended for. Most processes designed for a specific chemistry are intolerant of contamination by other chemistries (G&P, n.d.). Explosions have occurred when lithium-ion batteries have been unknowingly included in recycling processes for lead-acid batteries (Gaines, 2014).

5.3.3 Survey results

Two rounds of surveys were used to identify the processes different companies use to recycle batteries, what materials they recover, and an indication of the efficiency of the processes used. Examples of these surveys can be found in Appendix A and Appendix B. The surveys were followed up with email conversations to clarify the content. The survey results, research on company websites, and information in the literature were used to form descriptions of the processes used by each company. These are shown below.

Company P1

Batteries are first sorted manually by cobalt content, and the state of health is checked. Damaged batteries and those with a high state of charge are discharged. For high-cobalt batteries, a mechanical pre-treatment is performed, including an impact mill stage and then magnetic separation. Plastic that has been separated is sent to incineration with energy recovery. Any cables recovered are sent to specialised companies for separation of recoverable materials. Vacuum pyrolysis is then performed at 500-600°C in order to remove the electrolyte and pyrolyse the plastic separators. This process works to deactivate the batteries. Both a wet scrubber and neutralisation are used to prevent emissions. The batteries are then fed to an electric arc furnace for recovery of metals.

Company P2

After manual sorting, batteries are first pyrolysed, which removes organic material such as paper, plastic and electrolyte, and deactivates the batteries. The batteries are then distilled thermally, which separates and extracts different metals. These metals are purified in a thermal refining treatment and cast into ingots. While Company P2 has plans to invest in hydrometallurgical treatments, they currently only use thermal treatments.

Company P3

Company P3 do not consider their process as ‘recycling’ batteries. Lithium-ion batteries are consumed in a smelting process for the recovery of cobalt, nickel and copper.

Company M1

The batteries are sorted by chemistry before processing. Lithium-ion batteries are fed into a crushing unit in batches. The batteries are crushed in a controlled atmosphere of carbon dioxide (Vadenbo, 2009). The released lithium is neutralised with moist air, and gas treatment is performed through a scrubber. The individual recoverable materials are separated in a multi-stage separating plant. No survey response was received from Company M1, so it was assumed that the multi-stage separating plant utilises only mechanical processes.

Company M2

Batteries are first sorted by chemistry. Lithium-ion batteries are crushed and the gases collected are treated. The shredded fractions undergo a physical treatment of several stages. First magnetic separation is performed to extract steel, then air ballistic separation to separate the light plastic fractions from the heavier metal fractions. The material is then sieved and density separation is performed.

Company M3

Company M3 uses a patented 'Dry Technology' to process lithium-ion batteries. The end product is a lithium-ion battery powder, which is then sold to metal refineries. The process consists of a two-phase crushing line. Dusts and gases generated are collected and returned into the recovered product. Magnetic separation and sieving can be performed if required by the customer.

Company H1

Company H1 first treats lithium-ion batteries mechanically, then uses chemical processes to extract materials. Further details of these processes were not provided by Company H1. However, it is likely that the mechanical processes referred to are crushing or shredding, and the chemical processes are leaching and solvent extraction.

Company H2

First, lithium-ion batteries are crushed under a liquid brine solution to prevent emissions and to reduce reactivity of processed batteries. If necessary, the batteries are treated cryogenically to remove residual energy (Georgi-Maschler et al., 2012). Mechanical processes separates the batteries into metal solids, metal-enriched liquid and plastic fluff. The metal solids are sold and the metal-enriched liquid is solidified using filtering technology and then sent off-site to be further purified hydrometallurgically.

Company H3

Batteries are first shredded under inert conditions, and charged batteries are deactivated in the process. Sieving and magnetic separation are used to separate ferrous, non-ferrous, and cathode powder fractions. Materials are then extracted hydrometallurgically, using solvent extraction and electrolysis. Paper and plastics are sent to landfill. This process has been patented by Company M2, who have partnered with Company H3 for the recycling of lithium-ion batteries.

Company C1

Lithium-ion batteries are fed into a smelter along with nickel metal hydride batteries. Gas cleaning is utilised and fluorine from the electrolyte is collected. The smelting process produces a slag that contains the lithium from the batteries. This slag is used as an addition to concrete. The smelter output is then treated hydrometallurgically for extraction of materials.

The surveys also requested that the companies provide a list of the materials recovered in the process. These results are shown in Table 5, along with a summary of the processes used by each company. It was determined that Company M3 and Company P3 do not perform full recycling of the batteries, and the data from these two companies was not used in further analysis.

Table 5 - Survey results: Processes and recovered materials

Company name	Processes used	Location	Copper	Aluminium	Nickel	Lithium	Cobalt	Manganese	Steel/iron	Plastic	Carbon	Fluorine
Company P1	Disassembly/sorting Mechanical treatment Vacuum pyrolysis Vacuum evaporation	Germany	Y	Y	Y	N	Y	Y	Y	Y (i)	Y (a)	N
Company M1	Crushing Neutralisation Gas treatment Separation	Switzerland	Y	Y	Y	N	Y	Y	Y	Y (u)	N	N
Company H1	Mechanical processes Chemical processes	Korea	Y	Y	N	N	Y	N	Y	Y (r)	N	N
Company M2	Crushing, gas treatment Magnetic separation Air ballistic separation Sieving Density separation	France	Y	Y	Y	N	Y	N	Y	Y (r)	Y	N
Company H2	Cryogenic cooling Shredding Hydrometallurgical (data Riba/ future)	USA	Y	Y	Y	Y	Y	Y	Y	Y (l)	N	N
Company P2	Sorting Pyrolysis Distillation Refining	France	Y	Y	N	N	Y	N	N	N	N	N
Company H3	Shredding Sieving Magnetic separation Solvent extraction Electrolysis	Singapore	Y	Y	Y	N	Y	Y	Y	Y (l)	N	N
Company C1	Smelting Gas cleaning Hydrometallurgical refining	Belgium	Y	N	Y	Y (c)	Y	N	Y	N	N	Y
Company P3	Smelting (batteries consumed in general metal recovery process)	USA	Y	N	Y	N	Y	N	N	N	N	N
Company M3	Crushing Gas treatment Separation	Finland	Lithium-ion battery powder recovered Sold to metal refineries					Mechanical process Pyrometallurgical process Hydrometallurgical process Combination hydro and pyro				

(r) = further recycled, (l) = landfill, (c) = addition to concrete, (a) = reused as reduction agent, (i) = incinerated, (u) = unspecified

Where possible, the data provided directly from the companies was used. Otherwise, information on the recovered materials was compiled from secondary sources. This presented some inconsistencies that were attributed to changes in recycling processes between publication dates of the literature and when the surveys were completed. If plastics were incinerated or landfilled they were not considered recovered.

5.4 Interpretations of results

The survey results and process descriptions are interpreted below in terms of the recovered materials and processes used.

5.4.1 Recovered materials

The results of the surveys showed a large range in the materials recovered by different recycling companies. It was found that all companies who provided data recovered cobalt and copper. This result was expected, as it was shown in section 5.2.1 that the most valuable material in the battery is cobalt, and copper also has significant value. Steel, nickel, and aluminium were also found to be commonly recovered. This is intuitive from the sale price of nickel (approximately \$800 AUD per tonne of batteries). Although steel represents the lowest value component of the battery, it is one of the simplest to extract, since it can be separated magnetically. Therefore, the lower sale price can be justified. Aluminium has a value similar to steel at \$135 AUD per tonne of batteries, although is still recovered despite its lower price. This is likely due to the demand for recycled aluminium considering the high cost and energy requirements of producing aluminium from raw materials.

Most companies claimed to recover plastic, and it was found that the only companies that did not recover plastic used pyrometallurgical processes at the first stage of recycling. This is consistent with the process descriptions, as organic materials such as plastic and paper are burnt during smelting and can therefore not be recovered. Of the companies that did recover plastic, the materials were treated in various ways. One company sent plastic to be incinerated for energy recovery, and two companies sent recovered plastics to landfill. This result was unexpected due to the presence of recycling processes for polypropylene and polyethylene. Two companies sent recovered plastic to external facilities for further recycling, and the remaining company did not specify what happened to plastic after recovery.

Of the eight companies who provided data, two claimed to recover lithium in the recycling process. These both use hydrometallurgical processes, as lithium cannot be recovered by pyrometallurgical processes alone. Currently, one company uses the lithium-containing slag from pyrolysis as an addition to concrete. According to the EU Battery Directive, recycling is the reprocessing of waste materials for their original purpose or for other purposes. For this reason, the use of lithium-containing slag as an addition to concrete was not considered to be recycling of lithium. These batteries contain only a small amount of lithium and unlike aluminium and steel, it is not as easy to recover, nor does it have such large demand. Recycled lithium is as much as five times the cost of lithium produced from the least costly process (Kumar, 2014). Recovery of lithium is therefore currently not economically viable.

Approximately half the companies recovered carbon in the process, with this material sometimes being reused as a reduction agent in the pyrometallurgical processes. Only one company recovered fluorine, and this was done through gas cleaning of the smelting emissions. Lastly, it was found that manganese was recovered by approximately half of the companies. Manganese is only present in the composition if the cathode contains it, so it cannot be assumed that the remaining companies do not recover manganese, as they may not collect manganese-containing batteries. Overall, the main finding from the recovered materials is that those most commonly recovered have the highest value. This reinforces the previously mentioned point that the battery recycling business is a price-driven industry.

5.4.2 Processes used

The survey results showed that a wide range of processes are currently used to recycle lithium-ion batteries. These processes vary in the types of materials that are recovered, and a summary of these results is shown in Table 6.

Table 6 - Survey results: Distribution of processes used

Process	Number of companies	Average number of recovered materials
Hydrometallurgical	3	6
Pyrometallurgical	2	5
Mechanical	2	7
Pyrometallurgical/hydrometallurgical	1	5

Of the processes investigated, it was found that pyrometallurgical processes and a combination of hydrometallurgical and pyrometallurgical processes recover the lowest number of materials. Pyrometallurgical processes are the most flexible in terms of the mix of battery types that can be fed to the process (Gaines, 2011). However, the types of materials recovered are often not altered with changing battery compositions. This can result in a lower number of materials being recovered, due to the mix of battery chemistries present. Hydrometallurgical processes on the other hand tend to be more specialised, and hence require more sorting (Gaines, 2011). The advantage is that more materials can be recovered. Only one company used a combination of hydrometallurgical and pyrometallurgical processes. The number of materials recovered was similar to those from pyrometallurgical processes. This result is surprising, as it indicates that no additional metals are recovered by adding the hydrometallurgical step after the pyrometallurgy. Since only once company performed this combination, further information would be needed to determine whether the combination is beneficial in terms of recovered materials. Solely mechanical processes were found to recover the largest number of materials.

5.5 Chapter summary

The aim of this chapter was to examine where, why and how lithium-ion batteries are currently recycled. A list of companies recycling lithium-ion batteries across the world was compiled and surveys were sent to these companies requesting information on their processes and recovered materials. It was found that most of these companies are located in Europe, with some facilities also in Asia and North America. Of these areas, all have laws regarding transportation of batteries, but only Europe enforces recycling efficiency targets (50% for lithium-ion batteries). An assessment of the potential value of a tonne of lithium-ion batteries was then carried out and the most valuable materials identified were cobalt, nickel and copper. The results from the surveys confirmed that recycling is a price-driven industry, since the most commonly recovered materials by recycling companies were also the most valuable materials. As in any business, the potential value must be weighed against the necessary costs. Hence recycling can also be thought of as a cost-driven industry. Lithium was only recovered by one company, due to its low value and difficulty to recover, and only two companies recovered plastic for further recycling. The remaining companies eliminated plastic through pyrometallurgical techniques, or recovered plastic only to have it incinerated or sent to landfill. Finally, it was found that according to the survey results, mechanical processes recover the largest number of materials, followed by hydrometallurgical processes, with both pyrometallurgical and a combination of pyrometallurgical and hydrometallurgical processes recovering the least number of materials.

Chapter 6 – Factors influencing recycling location decisions

There are currently no facilities in Australia capable of recycling lithium-ion batteries; batteries collected for recycling are sent overseas for processing. At present, lithium-ion batteries collected by the ABRI are sent to Company H3 in Singapore, Company H1 in Korea, and Company C1 in Belgium (ABRI, 2014). The aim of this chapter is to present some factors that may influence the decisions regarding where batteries are sent for recycling, and provide some insight as to which companies provide the most benefits. These benefits will be assessed in terms of cost, recycling efficiency, and environmental effects, with the largest focus on environmental effects.

6.1 Costs

In general the financial costs associated with collection, sorting and recycling of portable batteries are considerably higher than the costs of landfill (Defra, 2006). However, there are also financial costs associated with handling environmental impacts, which are prevented by recycling. The recycling companies who provided information on recovered materials and processes were also asked whether they charged a fee to collectors for recycling, or whether they paid the collectors for the spent batteries. Five companies responded, but were not able to provide exact amounts charged as it is considered internal information. The responses are shown in Table 7.

Table 7 - Survey results: Payment methods

Company	Processes	Location	Payment plan
Company P1	Pyrometallurgical	Europe	Pays for high cobalt batteries Charges for low cobalt batteries
Company P2	Pyrometallurgical	Europe	Charges for batteries
Company C1	Pyrometallurgical/ hydrometallurgical	Europe	Pays for lithium-ion vehicle batteries
Company P3	Pyrometallurgical	USA	Pays for batteries based on metals found
Company H1	Hydrometallurgical	Asia	Pays for batteries containing cobalt

It should be noted that Company P3 do not consider their treatment as ‘recycling’ batteries, as the batteries are consumed in a smelting process for cobalt, nickel and copper recovery. The results from this survey question show the strong relationship between recycling and value of materials. Most companies indicated that they buy waste batteries for processing. However, in Europe, where collection and recycling efficiency targets are enforced, recycling companies may be willing to accept batteries that do not contain cobalt in order to meet these targets. The result is that they must charge a fee for this service. In America, there is no directive for battery recycling targets. This means that the process is entirely cost driven, indicating that companies are more likely to pay for batteries, and only accept batteries containing valuable materials. Overall, it can be seen that generally batteries containing cobalt are bought by recyclers, but current information on the price is not available publicly.

The concept of paying for waste disposal may be received differently within different cultures. Some places, such as in Europe, have a market prepared to pay for better environmental outcomes from waste disposal. This system will not necessarily be as successful in places where there are different cultural norms, since recycling rates are influenced by economic and moral motives (Hage et al., 2009).

In terms of the process costs of recycling (as seen by the recycling companies), it is likely that there are also variations. Although no data was available on the costs of the recycling processes, it can be assumed that the processes containing more stages are likely to be associated with higher costs. This is because more materials can be recovered with more steps. The higher cost may be offset by increased opportunity of material sale. However, the most valuable materials as determined in section 5.2.1 were cobalt, nickel, and copper, which can all be recovered using mechanical, hydrometallurgical or pyrometallurgical techniques alone.

6.2 Recycling efficiency

The surveys sent to recyclers also requested the recycling efficiency by weight associated with the processes used. Most companies were unable to provide this due to privacy reasons. Using the assumed composition and materials recovered, a maximum possible recycling efficiency was calculated for each company. These efficiencies were optimistic, assuming that any recovered plastic was further recycled (unless otherwise specified), and assuming the recovered materials were 100% pure. Carbon was not included in these calculations, due to the uncertainty of which companies did or did not recover carbon, and if so, where it was subsequently sent. Additionally, if manganese was recovered, this was not included in the calculation due to the assumption of cobalt-containing cathodes. The results are shown in Table 8, along with any recycling efficiencies that were provided by the companies.

Table 8 - Estimated recycling efficiencies

Company	Location	Processes	Max calculated efficiency	Provided efficiency
Company P1	Europe	Pyrometallurgical	55.6%	64.9%
Company P2	Europe	Pyrometallurgical	31.1%	-
Company M1	Europe	Mechanical	69.6%	-
Company M2	Europe	Mechanical	69.6%	-
Company H1	Asia	Hydrometallurgical	65.3%	-
Company H2	North America	Hydrometallurgical	57.5%	-
Company H3	Asia	Hydrometallurgical	55.6%	-
Company C1	Europe	Pyrometallurgical/ Hydrometallurgical	50.1%	> 65%

Only Company P1 and Company C1 provided indications of the recycling efficiency, and in both cases, these estimations were higher than the calculated efficiencies. It is difficult to compare these provided values with the estimates made since efficiency can be calculated in multiple different ways. Given that the provided efficiencies are higher than the calculated values, it is possible that these values represent the efficiency of the recovered materials only. For example, the efficiency provided by Company C1 (65%) may refer to the fraction of copper, nickel, steel, cobalt and fluorine that is recovered from the batteries as a percentage of the total amounts. It is also possible that carbon (which accounts for approximately 16% of the battery's composition), was included in their calculations. It is also possible that energy recovery was considered recycling in the calculation. For example, in some cases plastic was recovered but then incinerated with energy recovery. However, this is excluded from the definition for recycling given by the EU Battery directive.

According to these results, all European recycling companies excluding Company P2 have achieved the minimum recycling efficiency of 50% for lithium-ion batteries specified by the EU Battery Directive. It is suspected that Company P2 recovers additional materials that were not mentioned in the survey. Company P2 was contacted to confirm this, but no response was received. If Company P2 is not considered, it can be seen that two European companies recovered 50-55% of materials and two recovered almost 70%. This indicates that there may be a difference in motivations for recycling. The two companies with the lower efficiencies, Company P1 and Company C1, both pay for batteries rather than charging a fee. This indicates that the processes are cost driven, and the recycling efficiency is met only because it is required. It may be that profit is maximised when only the most valuable materials are recovered, to reduce the number of processes and hence cost. Outside of Europe, there are no requirements for recycling efficiency. Regardless, according to the estimations, the remaining three companies included in the analysis recover above 50% of materials. Assuming these processes are purely cost driven, it can be deduced that this efficiency is necessary in order to maximise profit.

Generally, it was found that on average the pyrometallurgical processes have the lowest efficiency, 43%, including the efficiency calculated for Company P2. The only combination of hydrometallurgical and pyrometallurgical processes had a calculated efficiency of 50%, and hydrometallurgical processes had an average efficiency of almost 60%. Purely mechanical processes had the highest average efficiency with 70%. There are uncertainties in these values as mentioned above, indicating that they may not accurately represent the recycling efficiencies for these processes. Nevertheless, there is a clear difference shown in the results. It is suspected that the differences can be largely attributed to the differences in recovery of plastic. The composition analysis found that lithium-ion batteries are approximately 14% plastic. If this plastic is recovered and then further recycled, as specified by some companies, it greatly affects the calculated recycling efficiency.

6.3 Environmental effects of battery recycling

Although it is known that recycling batteries is beneficial to the environment through the prevention of raw materials extraction, the actual processes of recycling still have negative effects. Ideally, these effects should be minimised to reduce the overall impact on the environment. The environmental effects of recycling lithium-ion batteries were evaluated in respect to transport, energy consumption and specific processes. A comparison was then made with the disposal of batteries in landfill. Where possible, analysis was made quantitatively using LCA principles. If the data was insufficient for the application of LCA, a more qualitative evaluation was performed.

6.3.1 Life cycle assessment

The ISO standards for LCA require the definition of product, goal, scope and functional unit. The product has been defined as portable lithium-ion batteries, and the goal is to provide a comparison between different processes for recycling these batteries. The functional unit is 1 tonne of batteries, and all values are given in terms of this unit. The scope of the analysis includes the end-of-life phase of the product life cycle only, and collection has been excluded from the analysis. Finally, three impact categories have been chosen for the focus of this analysis: global warming potential (GWP), expressed in kilograms of carbon dioxide equivalent (kg CO₂-eq); human toxicity potential (HTP); and terrestrial ecotoxicity potential (TETP); both expressed in kilograms of dichlorobenzene equivalent (kg DCB-eq). All characterisation was performed using the CML 2001-April 2013 database. Normalisation of the results was also performed to compare the results to a reference value: the impact of one person in one year. For this normalisation, the 'World, Year 2000' factors were used in GaBi LCA software. This gives the results in terms of person equivalents (PE), or the impact potential per person per year, without specific reference to one region.

6.3.2 Transport

All recycling methods require transport of the waste batteries to a recycling plant. The emissions associated with transport depend on the transport means and distance travelled. In the Australian context, transport should be taken into consideration since waste lithium-ion batteries currently must be exported for processing. In order to make a general comparison of the transport to different continents, an analysis of the environmental effects was performed using LCA principles.

Four general locations were chosen: Europe, Asia, North America and Australia. It was assumed that the distances travelled by road were equal for each location, and that batteries collected in Australia were shipped from Sydney. The largest ports in the general areas were identified and the distances to these ports from Sydney were calculated (Sea Distances, 2014). GaBi LCA software was then used to estimate the environmental impacts on the chosen impact categories resulting from the shipping of 1 tonne of batteries to each location. For the analysis, the transport option chosen was 'EU-27 – Container ship including fuel'. The results are given in the units associated with each impact category, but also in the units of person equivalents (PE), a fraction of the same impacts caused by one person in one year. The results are shown in

Table 9.

Table 9 - Environmental impacts due to transport per tonne of lithium-ion batteries

Location	Distance (by sea)	GWP 100		HTP		TETP	
<i>Units</i>	<i>km</i>	<i>kg CO₂-eq</i>	<i>PE</i>	<i>kg DCB-eq</i>	<i>PE</i>	<i>kg DCB-eq</i>	<i>PE</i>
Europe (Rotterdam)	21428	306	7.3e-12	14.1	5.5e-12	0.0446	4.1e-14
North America (Houston)	17112	245	5.9e-12	11.2	4.4e-12	0.0356	3.3e-14
Asia (Singapore)	7914	113	2.7e-12	5.2	2.0e-12	0.0165	1.5e-14
Australia (Sydney)	0	0	0	0	0	0	0

The results showed that environmental effects of recycling batteries can be reduced by choosing recycling locations closer to Australia. For example, if batteries are recycled within Australia instead of being shipped to Europe, approximately 300kg CO₂-eq can be saved. It can also be seen that the impacts on terrestrial ecotoxicity are significantly lower than those on human toxicity. These impacts can be compared since they are the same units. Several assumptions were made for the analysis, giving an approximate result. A more accurate analysis could be performed by including specific locations of ports and recycling facilities, and by investigating the exact methods of transport currently used.

6.3.3 Energy consumption

The surveys sent to recycling companies also requested the energy use of the processes used. Some companies provided information, and the remaining information was found through research. For Company H1 and Company P2, no information was available. Values for energy consumption of typical processes found in the literature have been included to broaden the results. The energy consumption results and the processing stage they correspond to are shown in Table 10.

Table 10 - Available energy consumption data

Company	Process	Energy use/tonne (kWh)	Source	Limitation
Company P1	Pyrometallurgical	40	Correspondence	Vacuum pyrolysis stage only
Company M1, 2004	Pyrometallurgical	800	Defra	Whole process
Company M2, 2004	Hydrometallurgical	140	Defra	Whole process
Company H2	Hydrometallurgical	173	(Gaines, 2013)	Shredding stage only
Company H3	Hydrometallurgical	707	Correspondence	Shredding stage only
Company M2	Mechanical	42	Correspondence	Crushing stage only
Company M3	Mechanical	300	(AkkuSer, 2011)	Whole process
Argonne literature	Mechanical	70	(Gaines, 2014)	Size reduction and physical separation
Company C1	Combination*	222	(Gaines and Dunn, 2012)	Smelter and gas cleaning only
Argonne literature	Combination*	1390	(Gaines and Dunn, 2012)	Smelting and gas clean up only

**Combination refers to a method including both pyrometallurgical and hydrometallurgical processes*

The results show that there is a huge variation in the data provided for energy consumption. For pyrometallurgical processes, the two values cannot be compared since they refer to different parts of the process. The hydrometallurgical values also contain inconsistencies. It was found that Company M2's operations in 2004 consumed less energy in the whole process than both Company H3 and Company H2's process does in the shredding stage only. Additionally, Company H3 supposedly consumes four times as much energy than Company H2 for the same process.

Within the mechanical processes, similar inconsistencies are present. If the size reduction and physical separation stages of the theoretical mechanical treatment described by Argonne are assumed to represent the total energy consumption, then this value is significantly lower than the energy consumption given for the whole process performed by Company M3. Company M2's energy consumption was given for the crushing stage only and is hence lower than the other values given for the entire processes. The largest inconsistency is shown in the data available for a combination of hydrometallurgical and pyrometallurgical processes. These values were given for the same section of the process and were found from the same source. However, they vary significantly, with the value for Company C1 representing less than a sixth of the energy consumed by the theoretical combination process from Argonne.

The inconsistency of the data may be due to several reasons. Firstly, the equipment used by different companies for the same purposes may vary in operation and age. These factors could have an effect on the overall energy efficiency of the equipment. The values also correspond to different portions of the entire process used by each company, and have been obtained from a wide range of sources. Although these factors could contribute to inconsistencies in the data, they cannot be the only cause. It is also possible that the recycling companies are not keeping track of their energy consumption effectively. This could be the reason that information was only available for one section of the process or not available at all.

Overall, these values cannot be used to form a comparison between the energy consumption of different recycling processes. The literature indicates that hydrometallurgical processes have a lower energy consumption than pyrometallurgical and mechanical processes. In order to confidently confirm this differentiation, the energy consumption of these processes must be understood in more detail than is currently available. In terms of the environmental effects of energy consumption, the energy generation is also significant. The location of recycling facilities and the associated energy mix has a strong influence on comparative results (Defra, 2006).

6.3.4 Pyrometallurgical processes

The largest environmental concern associated with pyrometallurgical processes is the energy consumption. Pyrolysis is the first stage of the processes used by Company P1 and Company P2, and this process occurs at 500-600°C (Accurec, 2014). The energy consumptions of different processes were discussed in section 6.3.3.

Pyrometallurgical processes are also associated with high emissions control to prevent releases to air. The initial round of surveys asked recycling companies to indicate whether CO₂ was emitted in the recycling process, and to list any other waste produced. The two companies using pyrometallurgical processes only who participated in the survey were Company P1 and Company P2. Company P1 indicated that CO₂ was emitted in the auto-thermal vacuum pyrolysis stage only, and Company P2 indicated that CO₂ was emitted in all three of the stages: pyrolysis, distillation and refining. The emissions occur from the organic materials such as plastic and paper that are burnt during thermal treatment. The emissions control systems such as scrubbers also deal with emissions such as hydrogen fluoride, which occurs from the combustion of the electrolyte and PVDF binder contained in the battery. Vacuum metallurgy has the advantages of high efficiency and better environmental properties since it does not require secondary off-gas or wastewater treatment (Huang et al., 2009).

The survey results did not provide enough detailed information to calculate the environmental impacts directly. As an alternative, inventory data for a pyrometallurgical process was found through a LCA conducted by the Department for Environment, Food and Rural Affairs (Defra) in the UK (2006). This data shown in Table 11 is associated with a pyrometallurgical process that was used by Company M1 in 2004. Company M1 currently recycles lithium-ion batteries mechanically, but it was assumed that the following data is representative of a pyrometallurgical process.

Table 11 – Life cycle inventory, pyrometallurgical process

Flow	Quantity	Unit
INPUTS		
<i>Raw material inputs</i>		
Waste batteries	1	tonne
NaOH (30%)	350	kg
<i>Electricity consumption</i>		
Electricity, national grid (Switzerland)	800	kWh
<i>Water consumption</i>		
Process water, main supply	1000	L
OUTPUTS		
<i>Product output</i>		
Steel, to steel industry	270	kg
Co-powder (cobalt oxide 60%, carbon 40%), to cobalt industry	192 (Co = 74.9)	kg kg
Non-ferrous metals, to metal industry	240	kg
MnO ₂ -powder, to recycler	10 (Mn = 6.3)	kg kg
<i>Emissions to air</i>		
Dust	0.208	kg
SO ₂	0.048	kg
<i>Emissions to water</i>		
Water to sewer	1000	L
SO ₂	40	kg
Cl	40	kg
<i>Solid wastes</i>		
Plastics, to incinerator	200	kg

It should be noted that there is a discrepancy of approximately 9% between raw material inputs and process outputs provided and, for this process, input exceeds output. The difference in mass between process inputs and outputs is considered to be due to losses of salts and oxygen, which leave the system with the wastewater and waste gas scrubber (Defra, 2006). These values also assume a mixture of waste lithium batteries, not one particular cathode material (Defra, 2006). The inventory data was entered into GaBi LCA software and the effects on the three chosen impact

categories was calculated. The results are shown in Table 12. The aforementioned discrepancies were not taken into account in the impact assessment.

Table 12 - Life cycle impact assessment, pyrometallurgical process

Process	GWP 100 (kg CO₂-eq)	HTP (kg DCB-eq)	TETP (kg DCB-eq)
Electricity generation	36.4	3.07	0.0891
Processing	0	0.0558	0
Plastics incineration	645	0.402	0.00499
<i>Total</i>	681	3.53	0.0941
Total (PE)	1.63e-11	1.37e-12	8.61e-14

The results indicate that the electricity generation and burning of plastics have the largest impact overall. Incineration of plastics contributed most to the global warming potential and electricity generation contributed most to human and terrestrial toxicity. The effects of electricity generation vary country to country and these effects could be reduced by implementing a larger proportion of renewable energy generation. As for the incineration of plastics, it is not necessary to consume plastic components in the heat treatment. Company P1 has shown through their processes that plastic components may be recovered mechanically preceding the pyrometallurgical process. It should be noted that the inventory data was obtained through secondary sources and may not fully represent the actual inputs and outputs of Company M1's system in 2004. Additionally, this data was obtained from one company only and does not provide a good representation of all pyrometallurgical processes.

6.3.5 Hydrometallurgical processes

The literature shows that hydrometallurgical processes are considered to have significantly lower energy requirements when compared to pyrometallurgical processes. Often the solutions used to leach materials are heated, but generally to temperatures less than 80°C. A comparison between processes in terms of energy consumption can be found in section 6.3.3.

The initial surveys sent to recycling companies requested information on what stages of the process emit CO₂. Two companies, Company H1 and Company H3, provided information for hydrometallurgical processes. Company H3 indicated that CO₂ was emitted, but did not specify whether this occurred at the shredding or hydrometallurgical stage of treatment. Company H1 indicated that CO₂ was not emitted at all during the recycling process. Since leaching and precipitation are being performed, wastewater is also produced from hydrometallurgical processes.

The survey results did not provide enough detailed information to calculate the environmental impacts directly. As an alternative, inventory data for a hydrometallurgical process was found through the LCA conducted by Defra in 2006. This data is shown in

Table 13 and is associated with a hydrometallurgical process that was used by Company M2 in 2004. Company M2 currently recycles lithium-ion batteries mechanically, but it was assumed that the following data is representative of a hydrometallurgical process.

Table 13 - Life cycle inventory, hydrometallurgical process

Flow	Quantity	Unit
INPUTS		
<i>Raw material inputs</i>		
Waste batteries	1	tonne
Reagent	25	kg
<i>Electricity consumption</i>		
Electricity, national grid (France)	140	kWh
<i>Water consumption</i>		
Industrial water	0.72	m ³
H ₂ SO ₄ (92%)	126	L
Lime	116	kg
OUTPUTS		
<i>Product output</i>		
Cobalt salt (as CoCO ₃), to cobalt producer	340 (Co = 180)	kg kg
Lithium salt (as Li ₂ CO ₃), to lithium producer	198 (Li = 30)	kg kg
Iron and steel, to steel industry	165	kg
Non-ferrous materials, to reprocessor	150	kg
<i>Emissions to air</i>		
SO ₂	4.5	g
VOC	2.5	g
<i>Emissions to water (sewer)</i>		
Solid suspension	12	g
Chemical oxygen	30	g
Total hydrocarbon	0.01	g
Cu+Co+Ni	0.05	g
Fluoride	0.03	g
Water to sewer	337	kg
<i>Solid wastes</i>		
Paper and plastic, to refining	130	kg
Residue to landfill	202	kg
Gypsum (as CaSO ₄ , H ₂ O), to landfill	339	kg

It should be noted that there is a discrepancy of approximately 13% between raw material inputs and process outputs provided and, for this process, output exceeds input. The difference in mass between process inputs and outputs is considered to be due to the water use within the process. Apart from the direct emission to sewer, the water input ends up in various output fractions, such as cobalt salts, lithium salts, residues and gypsum (Defra, 2006). The inventory data was entered into GaBi LCA software and the effects on the three chosen impact categories were calculated. The results are shown in Table 14. The aforementioned discrepancies were not taken account in the impact assessment.

Table 14 - Life cycle impact assessment, hydrometallurgical process

Process	GWP 100 (kg CO₂-eq)	HTP (kg DCB-eq)	TETP (kg DCB-eq)
Electricity generation	16	1.36	0.0169
Processing	0	0.000783	9.87e-6
Landfill gypsum	817	0.754	0.493
Landfill residue	487	0.449	0.294
<i>Total</i>	1320	2.57	0.803
Total (PE)	3.16e-11	9.95e-13	7.35e-13

The results indicate that a large proportion of the environmental effects of processing lithium-ion batteries hydrometallurgically come from the landfill of gypsum and residue and from electricity consumption. As was the case for pyrometallurgical treatment, the impacts due to electricity generation may be reduced by implementing a larger proportion of renewable energy. The composition of the waste to landfill was not specified in the inventory data, so the impacts were modelled using a general landfill process. It may be possible to further treat the residue, resulting in less materials ending up in landfill. It should be noted that the inventory data was obtained through secondary sources and may not fully represent the actual inputs and outputs of Company M2's system in 2004. Additionally, this data was obtained from one company only and is not representative of all hydrometallurgical processes.

6.3.6 Mechanical processes

Mechanical processes have two uses. They may be used for crushing or shredding batteries (which is often a step preceding hydrometallurgical processing), and they may be used for separation (i.e. magnetic, sieving, density). All these processes consume energy. A comparison between processes in terms of energy consumption can be found in section 6.3.3. Crushing and shredding processes are often performed first, to liberate battery components. This process is often performed in a controlled atmosphere to contain any emissions occurring from the electrolyte. It is also done to prevent the batteries from exploding. The initial surveys asked recycling companies to list what waste was produced during the process. Company M2 was the only company using solely mechanical processes to provide a response. Company M2 stated that volatile organic compounds (VOCs) were emitted during the crushing stage, and that these emissions must stay below a certain permitted amount. Company M2 also indicated that during the mechanical separation stage, dust was emitted, which contained mainly carbon.

One of the benefits of mechanical processing only is that it is the closest recycling process to direct recovery. Direct recovery is when the battery materials are separated but not processed, and the outputs are battery-grade materials, i.e. the casing is recovered and reused as casing for new batteries. This is best achieved by remanufacture and is the most efficient way to treat end-of-life batteries because it saves the most energy by reducing the number of processes required (Gaines, 2013). In general, low temperature recycling technologies are especially beneficial. There are lower energy requirements and less material transformation. This results in a more direct reuse/recycling of materials used in batteries (EPA, 2013). The downside of direct recovery is that it requires as uniform feed as possible, i.e. all the same battery type (Gaines, 2011).

The survey results did not provide sufficient data to perform a life cycle assessment for mechanical recycling processes. Furthermore, no inventory data could be found in the literature that was similar to those used for the pyrometallurgical and hydrometallurgical impact assessments. This prevents a clear comparison being made between all different processes.

6.3.7 Landfill

In Australia, only 1.8% of portable lithium-ion batteries were recycled in 2012-2013. The remaining fraction were hoarded or sent to landfill, with the hoarded batteries likely reaching landfill eventually. As well as contributing to the sheer volume of waste that reaches landfill, this can also have a toxic effect on the environment. In landfill, greenhouse gas emissions are produced from organic components such as paper, plastic and electrolyte solvents (Defra, 2006). Furthermore, heavy metals may be leached to the environment, and can reach soil or water depending on the condition of the landfill. The LCA conducted by Defra compared landfilling to recycling for a mix of battery types and found that in all impact categories, recycling causes significantly less damage to the environment. It was found that in the three impact categories used for this project (GWP 100, HTP and TETP), there was an overall environmental benefit from recycling batteries (Defra, 2006). In contrast, landfilling caused damaging effects on the environment in all impact categories. This study included all portable battery chemistries, not just lithium-ion, and did not break down the results by battery type. For this reason, the results cannot be used directly in this study to say that recycling lithium-ion batteries is better than landfilling in terms of environmental benefits. This is primarily due to the fact that lithium-ion batteries have a lower toxicity than the mix of batteries used in Defra's LCA (i.e. nickel cadmium included), and may cause less damage by landfilling when compared to a mixture of battery types.

To estimate the environmental effect of landfilling batteries, GaBi LCA software was used. Due to limitations in the software, only the impacts due to the nickel, copper, and aluminium content of the batteries were assessed. The results were calculated with 5% of heavy metals being leached to soil, as this is the fraction assumed by Defra. The impacts to the chosen three categories are shown in Table 15.

Table 15 - Life cycle impact assessment, landfill

Process	GWP 100 (kg CO₂-eq)*	HTP (kg DCB-eq)	TETP (kg DCB-eq)
Landfill	10.04	5.77e-3	578
Landfill (PE)	2.4e-13	2.24e-9	5.28e-10

** The values for GWP 100 were not available through GaBi. Consequently, these values were calculated using emissions associated with landfill of mixed plastics and characterisation factors from the IPCC (IPCC, 2007). The calculations considered only carbon dioxide, methane, and nitrous oxide.*

It can be seen immediately that the values for human toxicity potential and terrestrial ecotoxicity potential are significantly higher than the corresponding values for both pyrometallurgical and hydrometallurgical processes, shown in Table 12 and Table 14 respectively. These comparisons are further discussed below. The results for metals and plastics shown in Table 15 were calculated separately. It was found that the metals did not contribute to the global warming potential, so these results are related only to the plastic content. For toxicity, only metals were considered. The limitations of this model lie in the exclusion of all components of the battery; the landfilling of plastic, electrolyte, cathode and casing are not included in the analysis. These components would all contribute to land use and the organic components would cause greenhouse gas emissions as they break down.

Although copper, nickel and aluminium were included in the GaBi process, it was found that only copper and nickel contributed to the environmental effects. This was a result of the characterisation database employed. CML 2001-April 2013 does not consider aluminium leached to soil as having an effect on toxicity. Through the survey results of this project, it was found that copper and nickel are both commonly recovered from recycling processes, due to the value that can be obtained from selling the recovered materials. It is apparent that these are also the materials in the battery that cause the most damage to the environment through landfill. Overall, this is a beneficial result, as it means that through recycling, the most harmful materials do not reach landfill. The environmental effects calculated above are compared to those of recycling scenarios in Table 16. Transport was not considered.

Table 16 - Comparison of processes in terms of environmental impacts per tonne of batteries

Process	GWP 100 (PE)	HTP (PE)	TETP (PE)
Pyrometallurgical	1.63e-11	1.37e-12	8.61e-14
Hydrometallurgical	3.16e-11	9.95e-13	7.35e-13
Landfill	2.4e-13	2.24e-09	5.28e-10

For global warming potential, landfill actually showed a lower impact than both pyrometallurgical and hydrometallurgical processes. This result can be explained by the number of processes required to recycle batteries, many of which involve carbon dioxide emissions. For both human toxicity potential and terrestrial ecotoxicity potential, landfill showed a significantly worse outcome when compared to recycling. Here, the effect on the environment is between three and four orders of magnitude higher when batteries are landfilled.

These results also take a very conservative approach. The landfill estimations did not include several components of the batteries, such as the casing, cathode, anode and electrolyte. Had these components been included in the evaluation, the impact due to landfill would have increased. Additionally, the estimations for recycling can be considered close to a maximum. Normally in a LCA, recycling effects are negative. This is due to the savings in natural resources and emissions that are associated with raw materials extraction and processing, and if taken into account, would decrease the impacts associated with recycling. If both these uncertainties are taken into account, the gap between the effects of recycling and landfill only increases.

6.4 Life cycle interpretation

The results of the life cycle impact assessment were used to make some comparisons between different end-of-life scenarios. Since there was a lack of data available for solely mechanical processes, they have been excluded from this comparison.

6.4.1 Comparison between recycling processes

First of all, the data was used to compare hydrometallurgical and pyrometallurgical processes, based on the impact assessment performed in sections 6.3.4 and 6.3.5. The data used for the impact assessment was obtained through secondary sources and was not representative of processes currently used by recycling companies when processing lithium-ion batteries. The results are shown in Figure 4, excluding transportation.

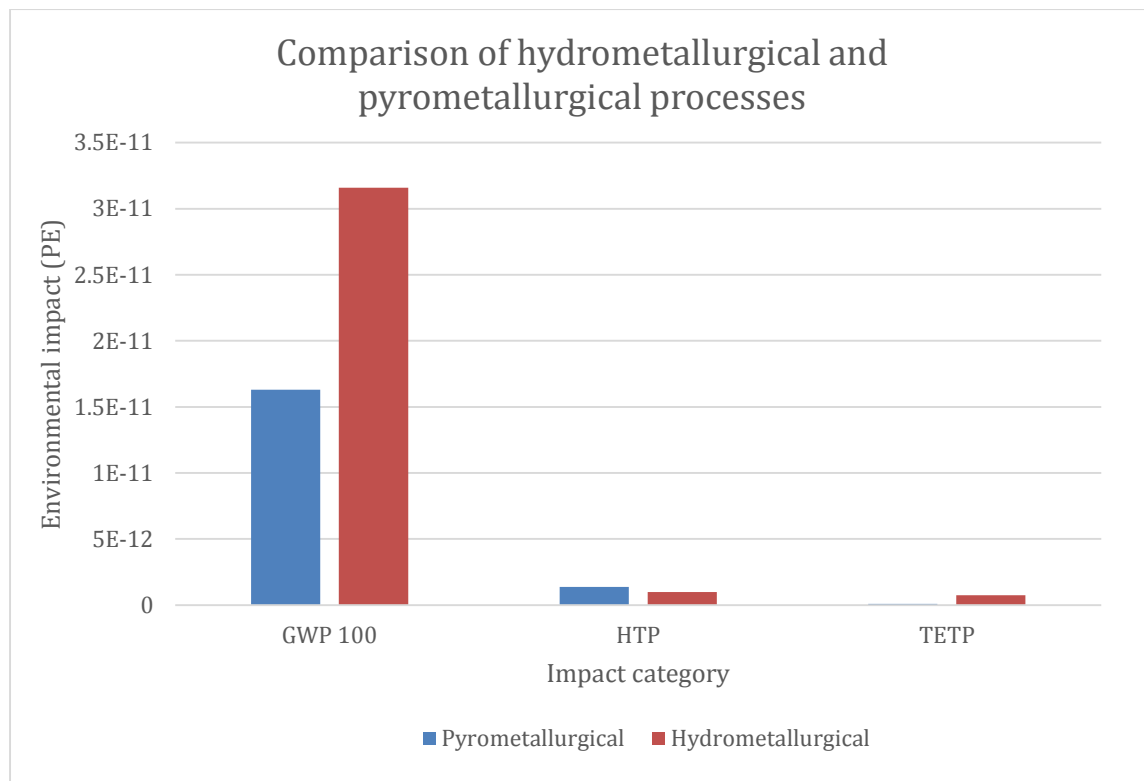


Figure 4 - Environmental impacts of hydrometallurgical and pyrometallurgical processes per tonne of lithium-ion batteries

It should be noted that these impact categories cannot be compared to each other, as they represent impacts to different environmental categories. The results indicate that within the category of global warming, hydrometallurgical processes have a larger impact per tonne of batteries recycled. As shown in section 6.3.5, the largest contribution of hydrometallurgical processes to global warming was the effect of landfilling waste produced during the process. In contrast, the largest contribution to global warming from the pyrometallurgical process was the incineration of plastics. This is important to note, as the amount of waste going to landfill may differ between hydrometallurgical processes. Additionally, the composition of the waste to landfill was not specified. It is also possible that the pyrometallurgical process produced waste for landfill that was not mentioned in the inventory data. The effects on human and terrestrial toxicity were similar in magnitude in each case, and presented a lower impact when compared to global warming potential in terms of the effects of one person in one year.

6.4.2 Comparison between locations

As shown in section 6.3.2, there are environmental effects associated with transporting end-of-life batteries by sea from Australia to other countries for recycling. To determine the significance of the impacts from transportation, four locations were compared for hydrometallurgical and pyrometallurgical processes. The transport environmental impacts were added to the recycling impacts, and it was assumed that all batteries were shipped from Sydney, Australia. The results are shown in Figure 5 and Figure 6.

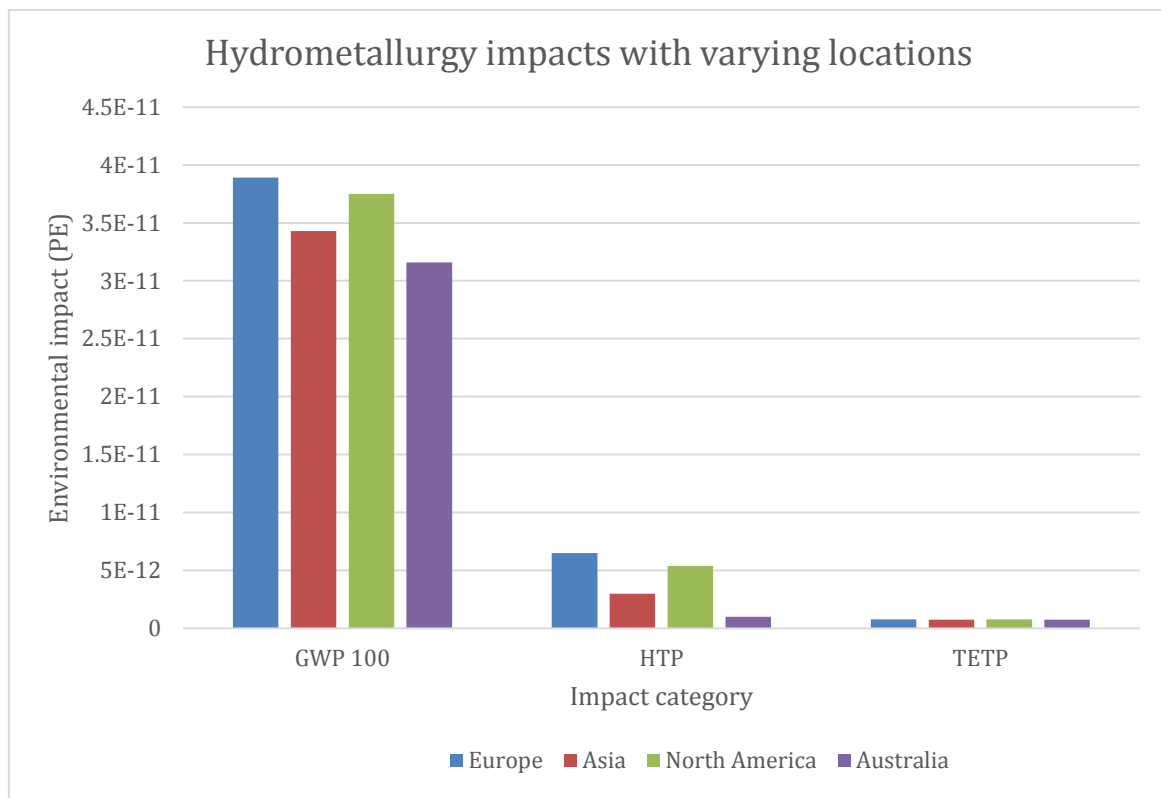


Figure 5 – Effect of location on environmental impacts of hydrometallurgical processes per tonne of lithium-ion batteries

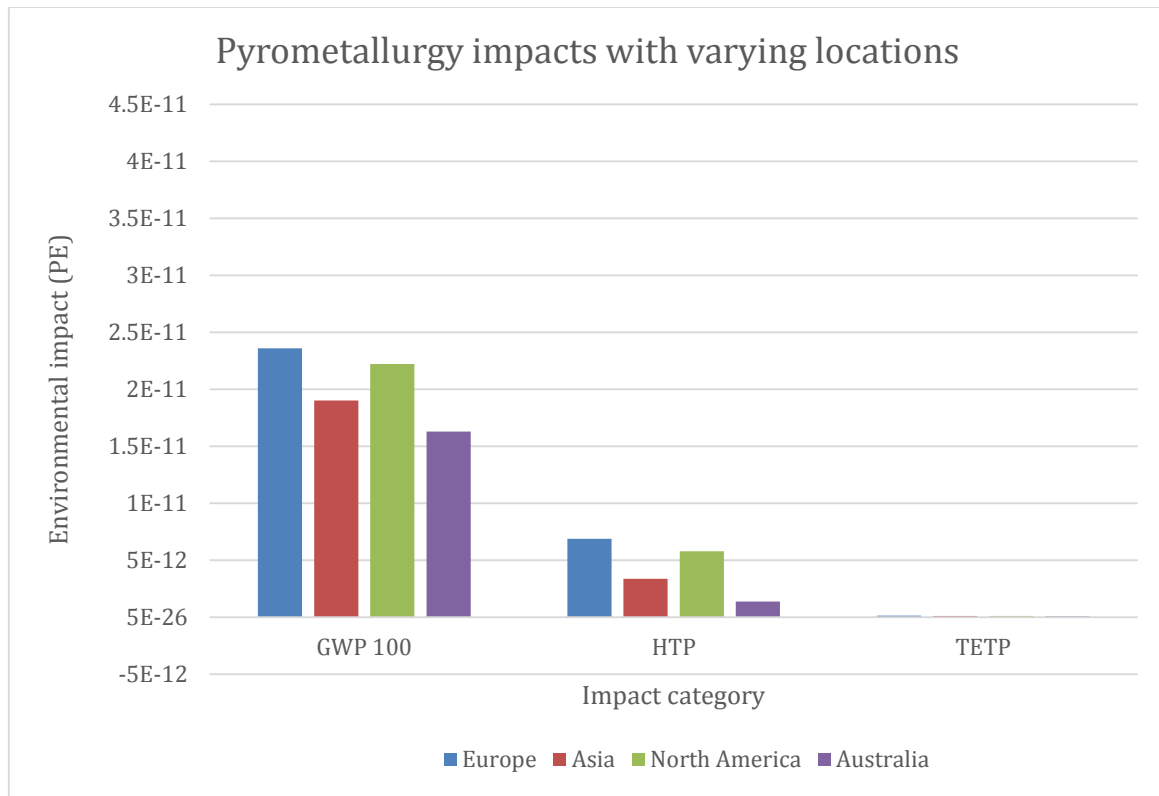


Figure 6 – Effect of location on environmental impacts of pyrometallurgical processes per tonne of lithium-ion batteries

It should be noted that although all values are in units of person equivalents (PE), this does not mean that the impacts from different categories can be directly compared. However, it can be seen that the transport does directly affect the environmental impacts associated with recycling, with larger distances causing larger environmental impacts. This was mainly seen with the global warming and human toxicity impacts categories, while terrestrial ecotoxicity showed less variation. The largest increase in environmental effects was seen by transporting batteries to Europe. This scenario causes an increase of 45% in impact on global warming potential for pyrometallurgy, and an increase of 550% on human toxicity potential for hydrometallurgy, when compared to the same processing within Australia.

6.5 Chapter summary

The aim of this chapter was to present some factors that may influence the decisions regarding where batteries are sent for recycling by organisations like the ABRI and MobileMuster. These factors were cost, recycling efficiency and environmental impacts. The cost comparison was made by asking recycling companies whether they charge a fee for their services or if they purchase spent batteries. It was found that most commonly the batteries are only purchased if they contain cobalt. This further strengthens the concept that battery recycling is price-driven.

The efficiencies of each recycling process were also estimated based on the number of materials recovered and assuming maximum recycling efficiency. These results showed that pyrometallurgical processes are the least efficient, with an average recovery rate of 43% by weight. Using a combination of hydrometallurgical and pyrometallurgical processes gave an efficiency of 50% and hydrometallurgical processes alone an efficiency of 60%. Purely mechanical processes had the highest average efficiency with 70%. It was suspected that the differences could largely be attributed to whether or not the process was capable of recovering plastic.

The environmental effects associated with lithium-ion battery recycling were then investigated. First, the data available regarding energy consumption was compiled from a range of sources. The results showed many significant inconsistencies and it was determined that the values were not sufficiently reliable to form a comparison of energy use between processes. In saying this, the literature indicates that pyrometallurgical processes use significantly more energy than hydrometallurgical processes, which was supported by inventory data through Defra. This inventory data was also used to quantify the environmental impacts of a pyrometallurgical and hydrometallurgical process. For the pyrometallurgical process it was found that the electricity generation and incineration of plastics contributed the most to the environmental impacts. It was also determined that these impacts could be reduced by sourcing renewable energy and by recovering plastic mechanically for further recycling before incineration is performed. The same analysis was then performed using the inventory data for a hydrometallurgical process. Again, the electricity generation caused a large contribution to the impacts, as well as the effects due to the disposal of residue to landfill. The composition of residue was not given so it cannot be determined whether further treatment was possible. All inventory data used was obtained through secondary sources and reflected processes that are no longer used by the companies who provided the data. The same analysis could not be completed for a purely mechanical process due to a lack of availability of inventory data.

The effects of recycling were then compared to an estimation of the effects of landfilling the same amount of batteries. For global warming potential, landfill actually showed a lower impact than both pyrometallurgical and hydrometallurgical processes, but for both human toxicity potential and terrestrial ecotoxicity potential, landfill showed a significantly worse outcome when compared to recycling. Here, the effect on the environment is between three and four orders of magnitude higher when batteries are landfilled. These results were considered conservative and a more accurate assessment would be expected to provide a clear difference between landfilling and recycling.

The impact assessment results were also used to compare hydrometallurgical and pyrometallurgical processes. The results showed that hydrometallurgical processes have a larger impact on global warming than pyrometallurgical processes. For the toxicity impact categories, the results were similar in magnitude. The impacts were then assessed including different transport scenarios. The most extreme scenario (shipping batteries from Australia to Europe) resulted in clear increases in environmental impacts. For example, a pyrometallurgical process in Europe causes a 45% increase in global warming impacts and a hydrometallurgical process in Europe causes a 550% increase in human toxicity impacts when compared to the same processes in Australia. Hence, the environmental effects are influenced by transport and can be reduced by choosing locations closer to Australia.

Overall, the results from this chapter showed that even though recycling is beneficial overall, there are detrimental environmental effects associated that may be reduced. These environmental effects can vary and should be considered when choosing a recycling process for lithium-ion batteries. However this may be difficult considering there is a general lack of transparency in this area that greatly inhibits an accurate assessment of the environmental effects of recycling.

Chapter 7 – Future trends and their effect on current recycling infrastructure

The purpose of this chapter is to identify the current trends in lithium-ion battery technology. These trends are analysed in terms of their effects on the current recycling infrastructure. The trends identified were: changes in applications of lithium-ion batteries, changes in composition, changes in available resources and changes in recycling processes. It was found that some trends directly influence others, and this has been acknowledged where possible. These results will be useful for forming a picture of the way we will recycle lithium-ion batteries in the future, and preparing for these forecasted changes.

7.1 Trend: Changes in applications

The use of electric vehicles is set to increase dramatically in the coming years (Amine et al., 2014). In 2013, approximately 150 000 electric vehicles were sold. This is estimated to reach 40 million per year by 2050 (UK ERC, 2014). The contribution of electric vehicle batteries to overall battery sales is shown in Figure 7 (EPA, 2013). Lithium-ion batteries are a promising energy storage technology for this application (Sonoc and Jeswiet, 2014), and it is estimated that more than 70% of electric vehicles likely to be introduced in 2015 will use lithium-ion batteries (Kumar, 2014).

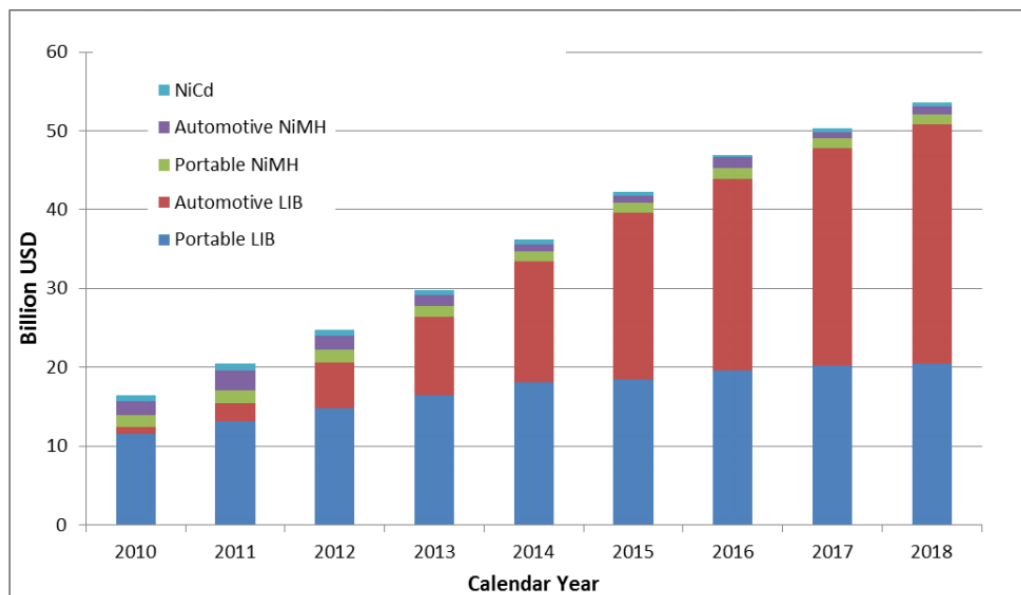


Figure 7 - Projected growth in battery sales

This projected increase in demand has an effect on recycling for multiple reasons. These are discussed in further detail below.

7.1.1 Effect on recycling due to collection

Electric vehicle batteries can weigh hundreds of kilograms (Nelson, n.d.). This size makes them more difficult to hoard, increasing the probability that they are recycled (FOE Europe, 2013). The application of lithium-ion batteries to vehicles also affects the way they are collected at end-of-life. Automotive lead-acid batteries are currently commonly recycled. This is mainly due to the fact that they are usually changed professionally and hence are not hoarded or disposed of to landfill. Although lithium-ion batteries do not present as severe environmental issues as lead-acid automotive batteries, it is likely that a similar system will develop for electric vehicles. The overall effect on recycling is an increase in the volume of lithium-ion batteries collected, due to both their size and increased likelihood of collection.

In regards to Europe alone, the rate of collection is likely to increase for electric vehicle batteries owing to additional laws. The European Commission provides regulation regarding electric vehicles through the ELV Directive 2000/53/EC. It stipulates that owners should be able to dispose of their cars to an authorised treatment facility without any cost (European Parliament, 2013b). The producer of the vehicle shall carry all, or a significant share of the costs. It also requires that the materials of all vehicles should be reused and recycled (excluding energy recovery) to 80% by an average weight per vehicle.

7.1.2 Effects on recycling due to processing

The move towards larger lithium-ion batteries also presents opportunities for reuse. It is not viable to repair portable batteries due to their size (Georgi-Maschler et al., 2012). Vehicle batteries are easier to disassemble, making them more suitable for this end-of-life scenario (Riba, 2013; H. Zhang et al., 2014). When lithium-ion batteries are classified as end-of-life, the bulk of materials are still active and the battery retains 80% of its original capacity (H. Zhang et al., 2014). Because of this reduced capacity, refurbished automotive lithium-ion batteries can only be used directly in lower-performance applications (Gaines, 2014). This is also known as repurposing. Refurbishment of lithium-ion batteries used in electric vehicles is still in the pilot stage (H. Zhang et al., 2014), though once lithium-ion batteries are disposed of on a large scale, the percentage of batteries that undergo refurbishment can be expected to rise (EPA, 2013). Already, Nissan plans to reuse batteries to store energy from photovoltaic panels or to store backup power for buildings (Gaines, 2014). It is profitable for electric vehicle manufacturers to support remanufacture because it avoids significant costs associated with producing new batteries (Standridge and Corneal, 2014). Due to the delay in return of materials for recycling caused by remanufacture, recycling companies might begin to include remanufacture in their operations. Research is also being conducted on the possibility of rejuvenating end-of-life vehicles batteries by replacing the electrolyte, which degrades through use (EPA, 2013). The major benefit of this technique is that the batteries may be able to be reused in the vehicle itself (EPA, 2013).

Repair and reuse also offers the most benefits environmentally. It replaces the largest number of processes required for manufacture of new products, saving more energy and resources than other end-of-life scenarios (H. Zhang et al., 2014). Although this may not be the reason that lithium-ion vehicle batteries are remanufactured at end-of-life (i.e. financial drivers are most prevalent), it still results in lower environmental impacts when compared to materials recovery.

7.1.3 Effects on recycling due to chemistry

Batteries used in electric vehicles must store more energy per unit volume and weight than is possible with current lithium cobalt oxide battery technology, and must be capable of undergoing many thousands of charge-discharge cycles (Amine et al., 2014). The result of these changed performance requirements is a need for new cathode materials. Lithium-ion batteries containing a cobalt cathode are unlikely to be the battery technology of choice for electric vehicle manufacturers as chemistries that are safer and better optimised for automotive applications are developed (UK ERC, 2014).

Alternative cathode materials for lithium-ion vehicle batteries are lithium iron phosphate (LFP) and lithium manganese oxide (LMO). While these cathode materials do not contain cobalt, there are other alternatives to lithium cobalt oxide that still do contain cobalt. For example, electric vehicle batteries may use lithium nickel manganese cobalt oxide (NMC), or lithium nickel cobalt aluminium oxide (NCA) (Gaines, 2013). These cathodes are already being used in the automotive industry, with the Nissan Leaf using a NMC cathode (Cole, 2012), and the Tesla Model S with a NCA cathode (MacKenzie, 2013). As we can see, there are a range of cathode materials that will be used in lithium-ion electric vehicle batteries. The potential reduction in cobalt use may decrease the motivation for recycling, and this effect is further discussed in section 7.2.

7.2 Trend: Changes in composition

The main drivers for changes in composition are cost and performance. Emerging cathode materials for lithium-ion batteries are starting to replace the use of cobalt with cheaper materials. Lithium-ion batteries have a high energy density, and their performance can also be improved by incorporating nanotechnology into the design. However, even the theoretical capacity of lithium-ion batteries is not capable of supplying electric vehicles with an 800km range, or powering a mobile phone for several days (Van Noorden, 2014). To enable higher performance applications, improvements are being sought in energy density, voltage, safety, and cycle life through new battery chemistries (Amine et al., 2014).

Two new technologies currently in research are lithium-sulphur batteries and lithium-air batteries. Lithium-sulphur batteries can theoretically store five times the energy of lithium-ion batteries at a lower cost (Shwartz, n.d.). The major challenge in commercialising lithium-sulphur batteries is obtaining a reasonable cycle life. The anodes of these batteries are commonly lithium metal, and the cathodes are made mainly from sulphur, which is obtained at low cost as a waste product from petroleum processing (Baum, 2014). According to manufacturers, lithium-sulphur batteries have less environmental impacts when compared to other technologies such as lithium-ion (Oxis Energy, n.d.). This is thought to be because metals such as nickel and cobalt are no longer required, and the sulphur used is recycled. The recyclability of the battery itself has not been researched.

The other emerging technology for replacing lithium-ion is the lithium-air or lithium oxygen battery. This chemistry offers the highest theoretical energy density, and unlike other battery technologies, is competitive with liquid fuels in this respect (Amine et al., 2014). Lithium-air cathodes consist mainly of carbon, with a metallic lithium anode. Much like lithium-sulphur batteries, the price of lithium-air batteries is expected to be lower than that of lithium-ion batteries, since metals are not used in the cathode. Lithium-air batteries are still in experimental stages, and problems such as low power output, poor cyclability, and low energy efficiency are being tackled. Some researchers believe the technology is not capable of reaching a desirable number of cycles (Van Noorden, 2014). Both lithium-sulphur and lithium-air batteries contain lithium metal as the anode, much like primary lithium batteries currently used for watches and hearing aids. These batteries are recyclable, and are accepted for example, by Company H2. This similarity suggests that it will also be possible to recycle lithium-sulphur and lithium-air batteries. A comparison of the energy densities of some of the battery types mentioned is shown in Figure 8 (Van Noorden, 2014).

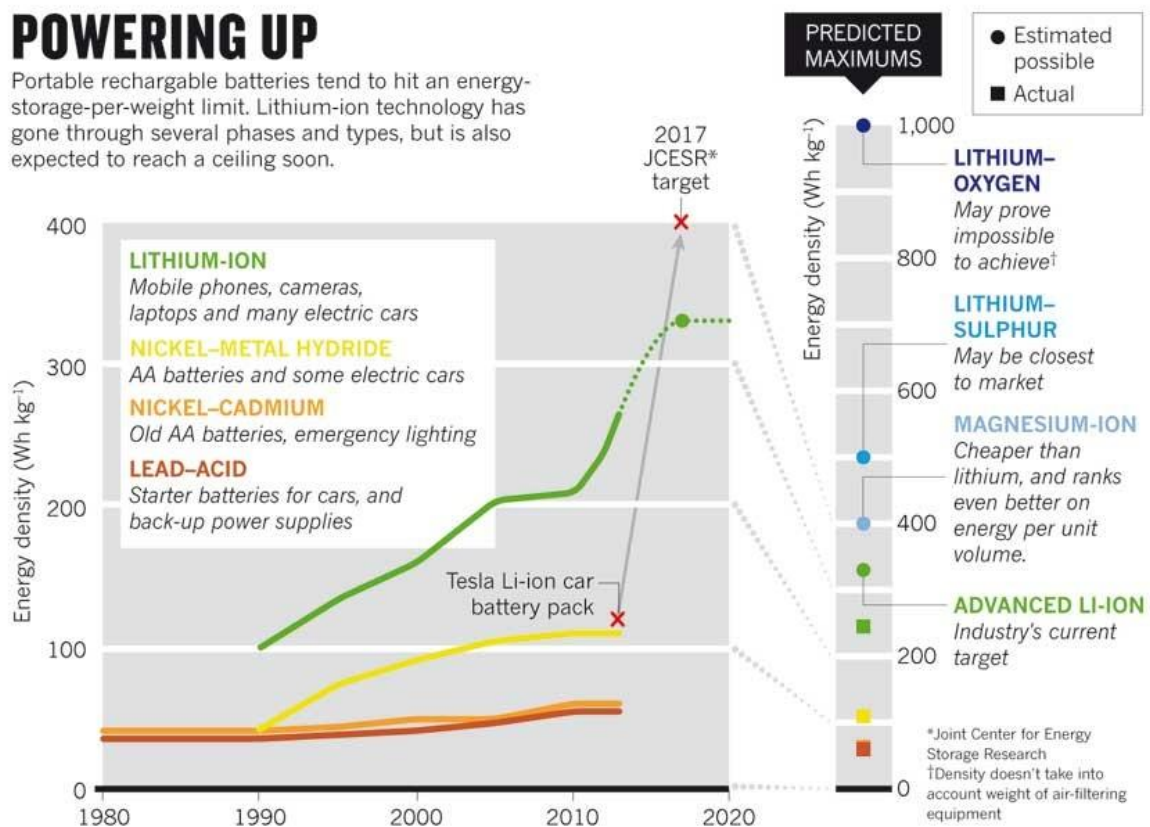


Figure 8 - Estimated and actual energy densities of different battery chemistries

It is believed that these advancements in lithium-ion battery types will prevent batteries from being recovered for monetary value due to the reduction of valuable materials like cobalt and nickel (Kumar, 2014). This concept was supported through the results of the surveys and contact with recycling companies. Through correspondence, Company P3 stated that “as long as lithium-ion batteries contain appreciable amounts of cobalt, nickel, and to a lesser extent, copper, we will be interested in processing them through our smelter. The trend in recent years is to lower cobalt content, which is making end-of-life batteries less of an interesting feed stock for us.” Recycling in the long term will therefore be mainly for ecological benefits and adherence to environmental laws (Kumar, 2014; Riba, 2013).

The effects of changes in composition will vary with location, depending on the laws present. Where regulations are in place requiring batteries to be recycled, it is likely that the recycling companies will need to begin charging a fee for recycling, since this return will not be made through sale of recovered materials. We already see this happening in Europe where laws are in place to ensure minimum recycling efficiencies. In places where there are less regulations, it is possible that the recycling of lithium-ion batteries will decrease. To ensure that batteries are still recycled, legislation enforcing collection and recycling targets will be needed. These laws play a large role in the recycling of batteries and in the future, could even dictate what materials are recovered through recycling (Riba, 2013).

7.3 Trend: Changes in available resources

The survey results showed that lithium is not commonly recovered by recycling companies, with only one company doing so. This was due to the low lithium content within the batteries, meaning it is currently not economically viable to recover. Lithium can be recycled an infinite amount of times (Battery University, 2014). However, only 1% of lithium consumed worldwide was recovered in 2011 (Riba, 2013). The fact that lithium is not commonly recycled, combined with its expected increase in demand makes it an important material to focus on in terms of material availability for lithium-ion batteries. For this reason, only lithium was focused on in this section.

Although lithium only makes up a small percentage of the composition of lithium-ion batteries, it is a necessary component for achieving high energy density. Given these properties, there is currently no substitute for lithium-based electric vehicle batteries (UK ERC, 2014). Lithium supply and price are influenced by additional demand in consumer electronics, geo-political relationships, environmental impact of mining, and new applications (Kumar, 2014). It was shown in section 7.1 that there is a large expected growth in lithium-ion batteries for electric vehicles. These batteries contain approximately 4kg of lithium (Battery University, 2014), so it is anticipated that this growth will greatly increase the demand for lithium. Lithium is widely considered a 'critical' metal, which refers to metals with the highest availability concern (UK ERC, 2014).

Batteries currently account for 27% of lithium consumption worldwide. Due to the projected increase in demand for lithium-ion batteries in electric vehicles, this fraction is expected to increase to 40% by 2020 (Kumar, 2014). Lithium reserves are not in immediate short supply (AEA Technology, 2010). However, it is predicted that there will be a shortage of lithium between 2021 and 2023 if lithium is not recycled (Sonoc and Jeswiet, 2014). The resultant increase in demand for lithium has been studied by the UK Energy Research Centre (UK ERC, 2014). The report showed that estimates for an increase in demand lay in the range of 150% to over 700% (UK ERC, 2014). These estimates were for the year 2030 and were represented as a percentage of supply in 2012. The projection, including past demand of lithium is shown in Figure 9 (UK ERC, 2014).

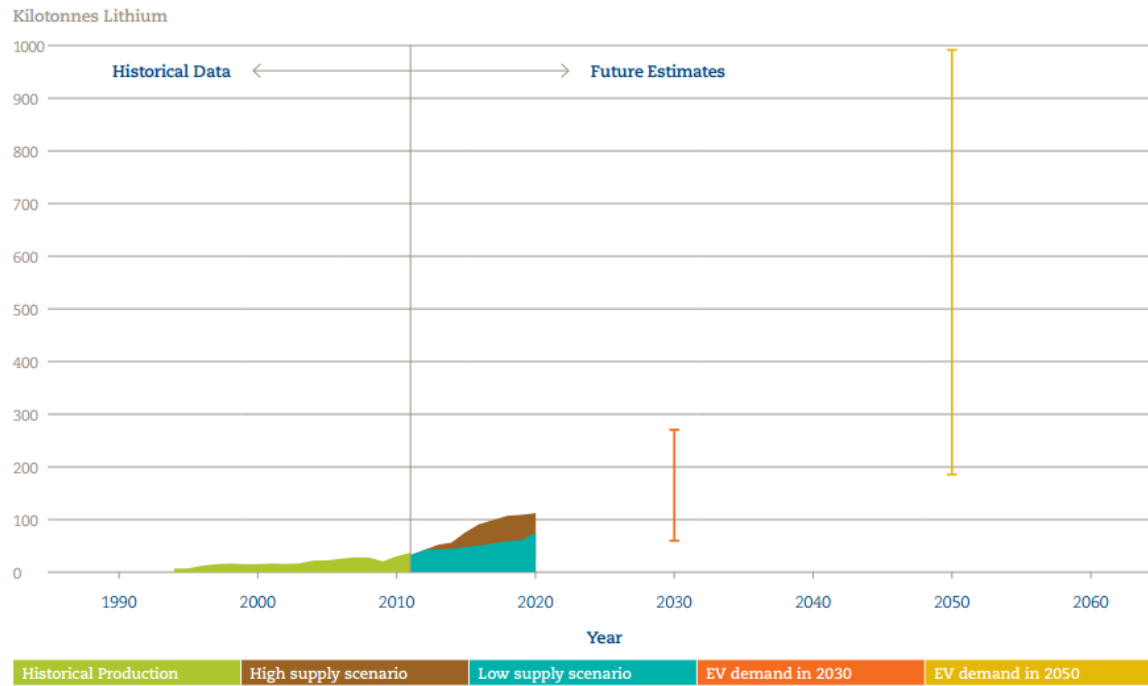


Figure 9 - A comparison of historical lithium production, future supply estimates and future demand estimates

It was noted in the ERC's report that the ranges presented in Figure 9 are large and reflect a wide range of assumptions in the literature and uncertainty in future demand estimates. The projections were therefore considered an illustration of the implications of assumptions in the literature, rather than a useful forecast of critical metals availability. Regardless, it does show the general consensus of a significant increase in lithium demand in the future.

The most prominent effect of this increase in demand would be an increase in the price of lithium due to scarcity (UK ERC, 2014). It has been estimated that as lithium supplies approach a point of shortage, prices could increase by ten times their current value (Standridge and Corneal, 2014). It is not currently economically viable to recover lithium, and as shown previously, the recovered materials from batteries are largely dependent on their value. An increase in the price of lithium will therefore create a need for recycling, likely improving collection rates and recycling efficiencies. Regulations may also be necessary if large-scale recovery of lithium is needed (Vadenbo, 2009).

7.4 Trend: Changes in recycling processes

As previously shown, changes are being sought in battery composition to reduce cost and improve battery performance. Often the research into new chemistries does not focus on how these batteries will eventually be recycled. Feeding batteries with new chemistries into existing hydrometallurgical or pyrometallurgical processes results in a reduced product value (Gaines, 2011), and hence developments are needed to ensure maximum materials recovery for lithium-ion batteries. The processes in development also focus on improving the environmental effects relating to emissions and energy efficiency, and ensuring lithium is recovered.

There are developments occurring in all current methods for recycling. However, there is a focus on low temperature methods such as mechanical and hydrometallurgical processes (Sonoc and Jeswiet, 2014), and on combinations of mechanical, hydrometallurgical and pyrometallurgical

processes (Georgi-Maschler et al., 2012). Mechanical and hydrometallurgical techniques are capable of recovering more materials and use less energy than pyrometallurgical techniques. The research being conducted in this area relates to different solvents, different mechanical pre-treatments, and performance testing of recovered materials for suitability to reuse in battery manufacture (Hanisch, n.d.).

It may be that as lithium supplies deplete, there will be a change from purely pyrometallurgical processes. This is already being seen from the French recycler Company P2, who is investing in the installation of hydrometallurgy treatment units to optimise its recycling process. It is not clear whether the purpose of this change is to recover lithium, although there is an opportunity to extract an additional 158.6 tonnes of lithium per year in Europe by treatment of the slag produced by pyrometallurgical processes (Riba, 2013). The Lithium-ion Battery Recycling Initiative (LiBRi) is currently developing a process for doing so (TU Clausthal, 2014). An environmental assessment carried out by Vadenbo (2009), indicated that there might also be environmental benefits in recovering lithium from slag. Overall, the advances in lithium-ion battery recycling point towards a system where different batteries have specific recycling processes, each dedicated to the specific chemistry.

7.5 Chapter summary

The purpose of this chapter was to identify the current trends in lithium-ion battery technology. These trends were then analysed in terms of their future effects on the current recycling infrastructure. The trends identified were: changes in applications of lithium-ion batteries, changes in composition, changes in available resources, and changes in recycling processes. The largest trend in the changes in applications was the increasing use of electric vehicles. The amount of sales is expected to increase dramatically in the coming years. As well as increasing the volume of batteries at end-of-life, this will also cause an increased likelihood of recycling due to the size of the batteries and methods of collection. It is also expected that these batteries will be refurbished for reuse at end-of-life. This may cause a change in the processes performed by recycling companies in order to gain maximum profit from spent batteries. Additionally, the increase in number of these batteries is expected to greatly affect the supply of lithium in the future. This increase in demand for lithium is likely to cause an increase in collection rates and recycling efficiency in order to maintain the availability of lithium.

Improvements to battery operation are constantly being sought out, which is resulting in changes in chemistry. With the aim of reducing costs, there is a current trend towards cathode materials that do not contain cobalt. The result of this is an overall threat to recycling as the profitability is reduced. This threat is more significant in countries outside of Europe, where there are no laws regarding target recycling efficiencies. To ensure that batteries are still recycled elsewhere, legislation similar to the EU Battery Directive may be required. Factors such as increasing lithium demand and increasing awareness of the environment are also leading towards changes in the processes used for recycling lithium-ion batteries. The processes in development are most commonly aimed at increasing the number of materials recovered, and improving the environmental effects relating to emissions and energy consumption. These processes are also becoming more specialised, focusing on one particular chemistry to achieve the best results.

Chapter 8 – Conclusions

The aim of this project was to investigate the different processes that are currently used for recycling lithium-ion batteries, and to compare these processes focusing on the associated environmental impacts. First, lithium-ion batteries were described in terms of their components, chemical composition and applications. The determined composition was then used for subsequent analysis. The current methods for recycling lithium-ion batteries were then identified, and compared in terms of their processes and recovered materials. This was done by examining where, why and how lithium-ion batteries are currently recycled. In the next stage of the project, different recycling processes were compared based on factors influencing decisions regarding where lithium-ion batteries are sent for recycling. This was done by investigating the associated costs and recycling efficiencies, and by evaluating the environmental effects of recycling lithium-ion batteries. In the final stage of the project, the current trends in lithium-ion battery technology were identified and the effects of these changes on current recycling infrastructure were explored.

Interpretation of the survey results showed that batteries are recycled to gain value from the recovered materials, and to adhere to laws imposing recycling targets. Most of the companies that recycle batteries are located in Europe, with some facilities also in Asia and North America. The methods used vary greatly, along with the materials recovered by each of the processes. The most commonly recovered materials are cobalt, nickel and copper, which are also the materials with the highest value per tonne of batteries. One unexpected result was the fact that multiple companies recover plastic only to send the waste to landfill or incineration rather than further recycling. Through comparison of the recovered materials of different processes, it was found that according to the survey results, purely mechanical processes recover the largest number of materials, while processes involving pyrometallurgical treatments recover the lowest number of materials.

The recovered materials were used to estimate recycling efficiencies for each company, and the results showed clear differences between the recycling processes. According to the results, pyrometallurgical processes are the least efficient, with an average recovery rate of 43% by weight. Using a combination of hydrometallurgical and pyrometallurgical processes gave an efficiency of 50% and hydrometallurgical processes alone an efficiency of 60%. Purely mechanical processes had the highest average efficiency with 70%. It was suspected that the differences could largely be attributed to whether or not the process was capable of recovering plastic. The cost comparison then showed that recycling companies will most commonly buy batteries that contain cobalt and may charge a fee if they do not contain cobalt. This confirmed the concept that battery recycling is a price-driven industry.

The environmental impacts evaluation showed which stages of the recycling processes contribute most to the environmental impacts. The largest contributors are electricity generation, incineration of plastics, and landfilling of residue. Using the impact results, the processes were then compared, showing that hydrometallurgical processes have a larger impact on global warming potential than pyrometallurgical processes. For the toxicity impact categories, the results were similar in magnitude. The impacts were then assessed including different transport scenarios. The most extreme scenario (shipping batteries from Australia to Europe) resulted in clear increases in environmental impacts. The effects of recycling were then compared to an estimation of the effects of landfilling the same amount of batteries. For global warming potential, landfill showed a lower impact than both pyrometallurgical and hydrometallurgical processes, but for both human toxicity potential and terrestrial ecotoxicity potential, landfill showed a significantly worse outcome when compared to recycling. The results for energy consumption were not sufficiently reliable to form a clear comparison. However, the literature indicated that pyrometallurgical processes use significantly more energy than hydrometallurgical processes.

By examining current technological trends, it was determined that the expected increase in electric vehicle sales will influence the development of new battery chemistries, recycling rates and recycling methods. The size of electric vehicle batteries results in a higher lithium content, which is expected to have an effect on lithium supplies. For this reason, there will be a greater need for lithium recovery, so recycling and collection rates are expected to increase. Lithium-ion vehicle batteries are also more likely to be repurposed for reuse at end-of-life, due to the remaining capacity once they are deemed unfit for use in electric vehicles. The trend for changes in composition is towards battery chemistries with lower cobalt content. The result of this is an overall threat to recycling as the profitability is reduced, especially where there are no laws regarding target recycling efficiencies. Factors such as increasing lithium demand and increasing environmental awareness are also leading towards changes in the processes used for recycling lithium-ion batteries. The processes in development are most commonly aimed at increasing the number of materials recovered, and improving the environmental effects relating to emissions and energy consumption.

Overall, these results can be used to influence decisions regarding where lithium-ion batteries are sent for recycling. The environmental effects of different processes were compared but a definitive comparison could not be made. This is largely due to the lack of detailed data available on the inputs and outputs of battery recycling processes, both directly from recycling companies and from the literature. This lack of transparency greatly hinders the possibility of a detailed environmental assessment. However, it is clear that there are environmental impacts associated with battery recycling, and that these impacts must be considered in order to ensure the best possible environmental outcomes from battery recycling. In terms of environmental effects, it is suggested that the most beneficial processes are those that utilise low temperatures, and are capable of recovering both plastic and lithium.

In terms of future work, there are many opportunities to further understand the environmental implications of lithium-ion battery recycling. Foremost, due to the omission of an environmental assessment for mechanical processes in this project, it would be greatly beneficial to perform a full LCA including all processes. In order to complete this task, it would be necessary to work closely with the recycling companies to ensure that a complete inventory is obtained. Due to this lack of data surrounding mechanical processes, their effect as a pre-treatment to pyrometallurgical processes could also not be examined. It would be interesting to see if there is an overall environmental benefit by adding this one stage to the process, preventing plastics from being incinerated. Considering the expected increases in lithium-ion battery use and advancements in processes, it would likewise be useful to explore the feasibility of commercialisation for the processes currently in development. These processes are focused on high material recovery and environmental benefits, so it is important to understand how they fit into a future of changing battery composition and increasing environmental awareness. Finally, to ensure that batteries with new chemistries are still recycled, it would be useful to see how adaptable the current processes are to the expected changes in composition.

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Appendix A

Initial survey example

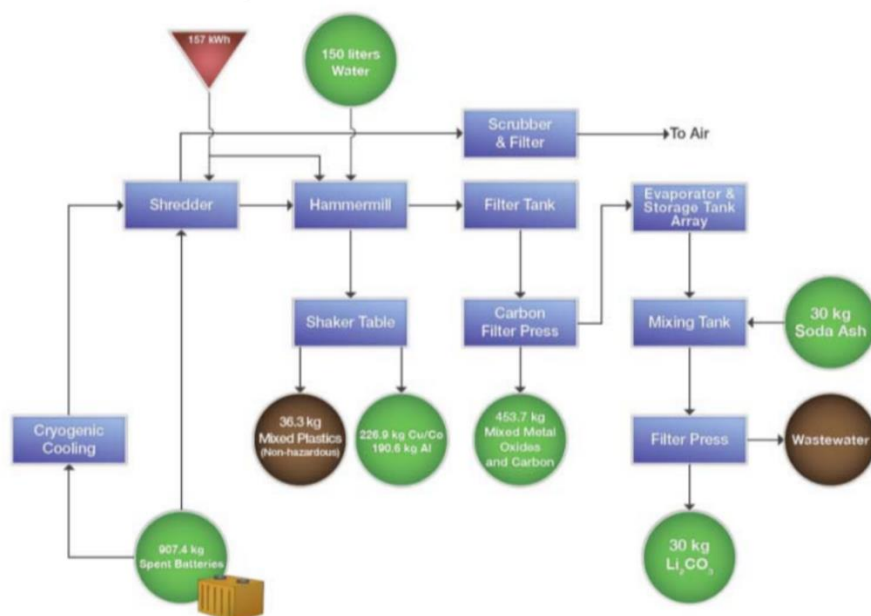
Please fill in the table below in as much detail as possible. Information provided on the company website has been completed. However, please highlight any incorrect information, and provide a correction. Thank you.

Battery type	Recycling efficiency (% by weight of recovered materials)	Processes used	Materials recovered	Materials sent to.. (e.g. sold, landfill)	Waste produced
Lithium ion		1. Shredding	Plastic fluff		
		2.			
		3.			
		4.	Copper Aluminium Cobalt		

After manual disassembly, are larger lithium ion batteries recycled using the same processes as smaller batteries?

What recycling fee does Company H2 charge per tonne of lithium ion batteries?

Does the following flow chart accurately represent the process flow of lithium ion batteries currently used by Company H2?



Appendix B

Secondary survey example

Process inputs and outputs survey

Please complete the following survey in regards to the recycling processes your company uses for *lithium ion portable batteries only*. The survey has been divided into sections corresponding to information provided in the initial survey, where general processes were specified. Data already given or made available on the company website has been filled in for convenience.

For quantitative questions, please provide answers in terms of the functional unit: 1 metric tonne (1000kg) of batteries. If other units are more appropriate, please specify what units are used. Additions can also be made to lists of materials where required.

Please include *all* materials that are inputs and outputs to the system, including waste to air and water. This is important for justification of results. If unsure about whether to include materials, add to the list and make a note. If unable to complete a section please state why (e.g. data not recorded).

Thank you for your participation!

Section 1 – Deactivation through pyrolysis

1. What is the purpose of this stage?	Deactivate batteries to prevent explosion			
	Recover materials			
	Eliminate organic matter and solvents			
	Other, please specify			
2. How is heat produced for pyrolysis?				
3. How much energy does this stage consumer per functional unit?				
4. What are the material inputs into the distillation stage? (i.e. fuel for heating, material for refining etc.) And what amounts of these materials are consumed per functional unit?	<i>Material</i>		<i>Amount</i>	
	Lithium ion batteries		1 tonne	
	Other			
	Other			
5. What materials are recovered, what amounts are recovered per functional unit and to what purity, and what happens to these materials (i.e. sent to distillation, sold)?	<i>Material</i>	<i>Amount</i>	<i>Purity (%)</i>	<i>Sent to</i>
	Cobalt			
	Copper			
	Aluminium scrap			
	Other			
6. What is the waste <i>emitted</i> in this stage, what amount is produced per functional unit, and where is it emitted to?	<i>Waste</i>		<i>Emitted to</i>	<i>Amount</i>
	Carbon dioxide		Air	
	Other			
	Other			
7. What is the waste <i>recovered</i> in this stage, how much is produced per functional unit, and what happens to these materials (i.e. sent elsewhere for further processing, send to landfill)?	<i>Waste</i>		<i>Amount</i>	<i>Sent to</i>
	Plastic			
	Electronics			
	Other			

Section 2 – Distillation

1. What is the purpose of this stage?	To separate materials through evaporation			
	Other, please specify			
	Other, please specify			
2. How is heat produced for distillation?				
3. What are the material inputs into the distillation stage? (i.e. fuel for heating, output material from pyrolysis etc.) And what amounts of these materials are consumed per functional unit?	<i>Material</i>	<i>Amount</i>		
4. How much energy does this stage consumer per functional unit?				
5. What materials are recovered, what amounts are recovered per functional unit and to what purity, and what happens to these materials (i.e. sent to refining, sold)?	<i>Material</i>	<i>Amount</i>	<i>Purity (%)</i>	<i>Sent to</i>
6. What is the waste <i>emitted</i> in this stage, how much is produced per functional unit, and where is it emitted to?	<i>Waste</i>	<i>Emitted to</i>	<i>Amount</i>	
	Carbon dioxide	Air		
	Other			
	Other			
7. What is the waste <i>recovered</i> in this stage, how much is produced per functional unit, and what happens to these materials (i.e. sent elsewhere for further processing, send to landfill)?	<i>Waste</i>	<i>Amount</i>	<i>Sent to</i>	

Section 3 – Refining

1. What is the purpose of this stage?	Refine metals for recovery			
	Other, please specify			
	Other, please specify			
2. How is heat produced for refining?				
3. How much energy does this stage consumer per functional unit?				
4. What are the material inputs into the distillation stage? (i.e. fuel for heating, output material from distillation etc.) And what amounts of these materials are consumed per functional unit?	<i>Material</i>		<i>Amount</i>	
5. What materials are recovered, what amounts are recovered per functional unit and to what purity, and what happens to these materials (i.e. sold)?	<i>Material</i>	<i>Amount</i>	<i>Purity (%)</i>	<i>Sent to</i>
	Cobalt			
	Copper			
	Aluminium ingots			
	Other			
6. What is the waste <i>emitted</i> in this stage, and how much is produced per functional unit, and where is it emitted to?	<i>Waste</i>		<i>Emitted to</i>	<i>Amount</i>
	Carbon dioxide		Air	
	Other			
	Other			
7. What is the waste <i>recovered</i> in this stage, how much is produced per functional unit, and what happens to these materials (i.e. sent elsewhere for further processing, send to landfill)?	<i>Waste</i>		<i>Amount</i>	<i>Sent to</i>