LIFE CYCLE BASED GREENHOUSE GAS FOOTPRINTS OF METAL PRODUCTION WITH RECYCLING SCENARIOS

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Abstract

Life cycle assessment (LCA) is a recognized tool to evaluate various processing routes for metal production. Declining ore grades and higher specific energy requirements for primary metal production put greater emphasis on recycling. Greenhouse gas (GHG) emissions of steel and aluminium metal production were quantified with recycling scenarios using material recovery facility (MRF) data from the database of SimaPro LCA software. The GHG footprint of the MRF is relatively minor compared with that of associated transport during collection (i.e. 10 times more than MRF) of kerbside recyclable material. Additionally, if the bulk recyclable material is sent overseas (i.e. Australia to China) from the MRF for further processing, the GHG footprint of shipping can significantly be large compared with the sum of the collection and MRF (assuming electricity is from same source). Thus opportunities exist for reducing GHG from secondary metal production if it is processed close to the MRF.

Introduction

It is a challenge to recycle metal significantly although theoretically infinitely possible due to their elemental nature. Leaks from the metal stocks in society occur through corrosion, wear and dispersive uses, or via land filling or similar activities that return metals to the earth. In order to provide a technically sound and transparent assessment of metal recycling, a methodology such as life cycle assessment (LCA) should be used. By taking a life cycle perspective, the beneficial recycling properties of metals can be evaluated in a manner that enables appropriate comparisons with other materials or product systems that do not have recycling loops. In practice, mixtures of primary and secondary metals are often used in new products, and also at the end-of-life stage of various processing methods used.

The difficulty of introducing recycling into LCA is to set the right boundaries for the different flows ending in different product systems. It is question of which observed material flow belongs to the first product system and which one to the second or subsequent systems. Recycling can be part of any product LCA. However, it is often a complex issue which requires specific considerations. As pointed out by Yellishetty et al. (2011) and Birat et al. (2006), LCA practitioners are left with much freedom in allocation of environmental burdens to account for recycling, thus making subjective judgments on recycling and allocation of credits to recycling. This often makes it difficult to compare the results of LCA studies conducted by two different practitioners even on the same processes. As LCA is often used to define policy in government, business and society circles, it should be based on a sound, objective and unbiased description of recycling.

Collection, transport and separation of materials in various streams of recyclable products are the first important steps for further recovery of metals. Although there were attempts to quantify various contributions from each of these steps on environmental impact in the context of end-of-life vehicle in the US (Gallon and Binder, 2006) but the application of this approach is limited in Australia.

Residents of local government areas (LGA) in Australia generally use a 240 L recycling bin with yellow lid for kerbside collection on a fortnightly basis filled with so call commingled waste. The recycling contractor uses trucks to collect glass, plastic, juice and milk cartons, aluminium, steel, newspaper and cardboard waste. In the group of metals, aluminium is in drink cans such as Coke cans, and steel is including pet food cans, tinned fruit cans, empty and clean paint and oil tins, and aerosol cans. The recyclable items on the kerbside are taken to a sorting factory, called a Materials Recovery Facility (MRF). Here they are emptied on a conveyer and sorted into different categories of glass, plastic, metals, paper etc. Typically, it takes about one hour to sort a truck load of recyclables. Each type of recyclable is put into separate bins, bundled or bailed up and stored until there are sufficient quantities to be transported to the manufacturer for reprocessing and reuse.

These are then sold to the manufacturing companies, who use machines to crush, shred, wash and melt the materials (reducing the need to use natural resources) so that they can be used to make new items, such as new bottles, plastic garden furniture, new cans, packaging and recycled paper products. Aluminium cans are sent to aluminium producers such as Alcan and steel materials are sent to a local steelmaker such as BlueScope Steel or OneSteel.

One company such as Visy has over 25 MRFs in Australia and recycles over 2.2 Mt of waste annually. In one particular year, this waste stream supplied over 22,000 t of steel, about 7,000 t of aluminium and 20 t of copper (Visy, 2013). Visy operates similar recycling based industrial operation in the United States with the trade name Pratt Industries.

A typical MRF has a capacity of 50 t material/h sorting or 300,000 t/year. MRFs are generally automated but with 6 to 10 people employed per plant. These plants have high productivity sorting lines that use gravity, air separation, magnetic, eddy current for aluminium cans and with hand sorting when necessary (Figure 1). An eddy current is an electric current induced within conductors by a changing magnetic field in the conductor. The eddy current separator uses a powerful magnetic field to separate non-ferrous metals from waste stream with a preceding step of magnetic separation of ferrometals. The eddy current separator is applied to a conveyor belt carrying a thin layer of mixed waste. An eddy current rotor sits at the end of the conveyor belt. Non-ferrous metals are thrown forward from the belt into a collection bin, while remaining waste stream fall off the belt due to gravity.

The objective of this paper is to describe the results of an LCA study of a material recovery facility (MRF). The greenhouse gas footprints of various products from a MRF have been estimated. Since there are several products from a MRF, the allocation of impact is an issue. This issue has also been described in this paper. Based on these results, the total GHG footprints of steel and aluminium have been estimated. For some unit processes within the life cycle boundary, GHG footprints were collected from the literature. The total GHG footprints of recycled steel and aluminium have been compared with primary metal production obtained from the literature.

Methodology

One simple case study has been selected for this paper based on the available primary data in the database of SimaPro LCA software (Australasian Unit Processes life cycle inventory data, 2013; PRe, 2013; Ecoinvent, 2013). This base data have been modified based on the expert judgment after comparing with existing in-house data available for other LCA study of metals by CSIRO.

Flow boundary for LCA

The typical boundary for an MRF is shown in Figure 1. This can be considered as a cradle to gate LCA. The collection starts from household kerbside (cradle), transported to a MRF and waste are separated, sorted and dispatched (gate). The metals are collected and bailed for further processing (extension of boundary to include reprocessing plant to final metal product). The GHG footprints of various unit processes of this MRF have been estimated. The GHG footprints of steel and aluminium metals have also been estimated for this study. It can be argued that the primary production chain (i.e. mining, processing, smelting, refining, manufacturing steps) should be considered precede the household use phase. This integration of whole life cycle would be undertaken in future, however, for this study, MRF has only been considered.

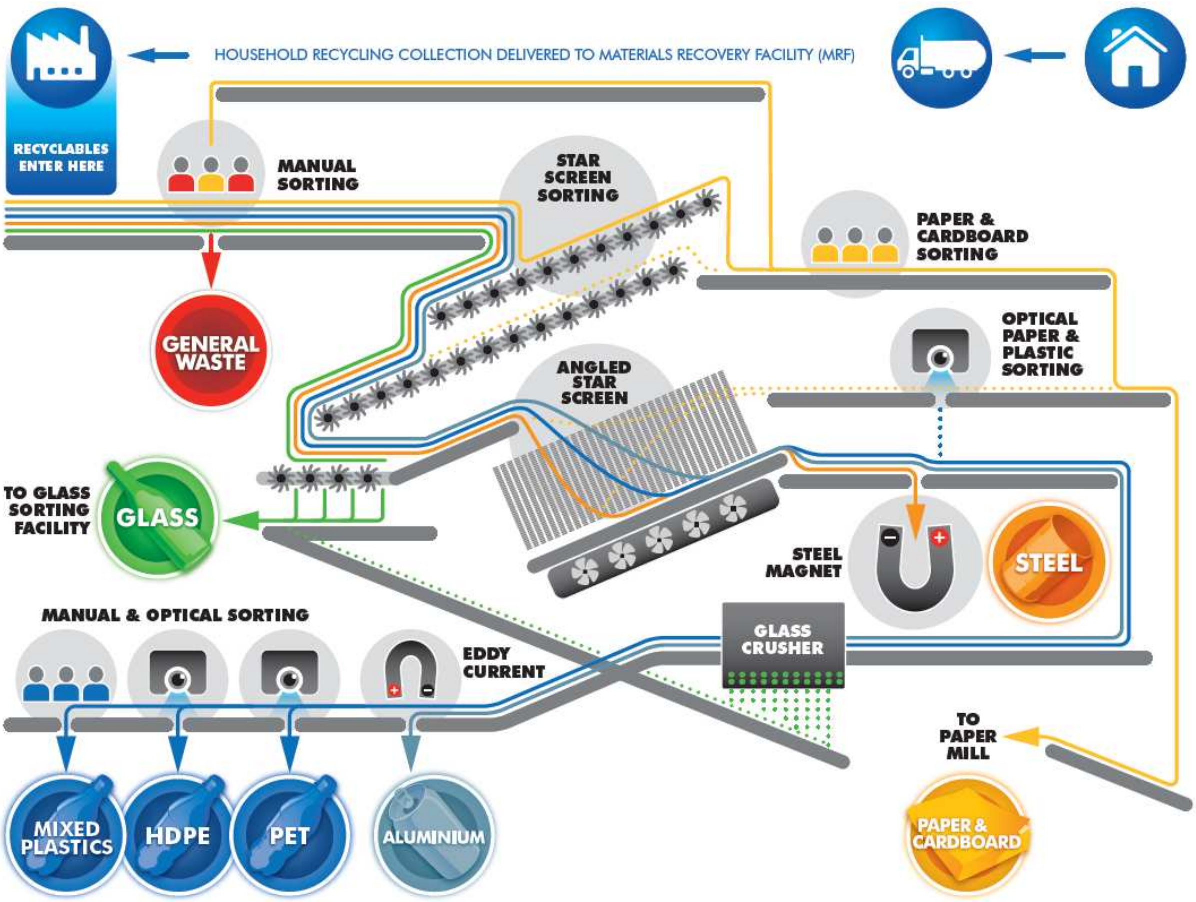

Figure 1. Schematic of a MRF (Visy, 2013)

Life cycle inventory

A load of 150 m^3 or 18 t of material assuming 8.7 t/m^3 average density (Australasian LCI, 2013) is assumed to be sorted by a MRF for this case. The primary data for MRF is shown in Table 1 and for transport scenario is shown in Table 2.

Table 1. LCI for MRF process

Activity or unit process	Value	Unit	Comments
Trommel screen	8	hours	Assumed to be running 8 hours per day
Front-end loader	2	hours	Running time
Conveyor	40	hours	5 conveyors running 8 hours each per day
Glass breaker	4	hours	Assumed runs half time
Magnetic separator	8	hours	Running time
Eddie current separator	4	hours	Running time

Table 2. LCI for transport of average 150 m^3 recyclables of the 30 councils in an Australian City (Australasian LCI, 2013)

Activity or unit process	Value	Unit	Comments
Collection time	28.97	hours	Door to door kerbside collection
Unloading time	0.77	hour	At a collection point
Traversed distance for recyclables in the suburban areas	393	km	Distance travelled during kerbside collection
Transit distance for bulk recyclables	0.67	km	From collection point to the MRF

A scenario has been assumed for processing of the recyclable metals. Generally, given the small amount of metal produced currently from MRF, it is reprocessed locally. However, if the recycling amount is increased and with the increased loss of Australian local remanufacturing and reprocessing facilities, there is a risk of exporting these recovered metals to China for further reprocessing. Although this is not the case for steel and aluminium but for other e-waste, generally recovered materials in Australia are sent overseas for reprocessing. If the recyclable materials are sent to China for reprocessing, the shipping distance for example, between Melbourne to Shanghai is assumed to be 8,100 km. International shipping freight has been assumed as the transport mode.

Allocation issues

MRF generally produce multiple products. Therefore, a method for allocating a proportion of the energy consumed and GWP to individual product is required. The impact can be allocated on a mass basis or economic basis (ISO, 2006). A mass based impact allocation has been used here.

Results and Discussion

The above inputs have been used in a SimaPro LCA model to generate results for further analysis. The results are presented in the next sections and their implications are discussed.

MRF Analysis

The typical composition of MRF output stream is shown in Figure 2 and their relative contributions on potential revenue in terms of estimated price is shown in Figure 3.

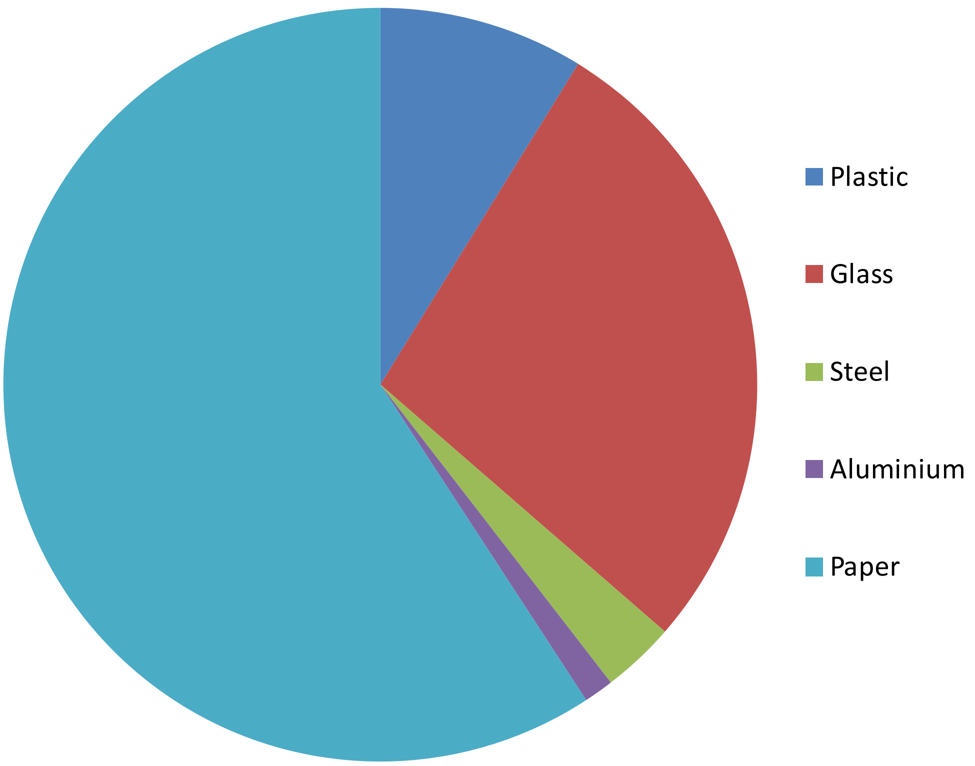

Figure 2. Typical material composition of MRF plant product stream

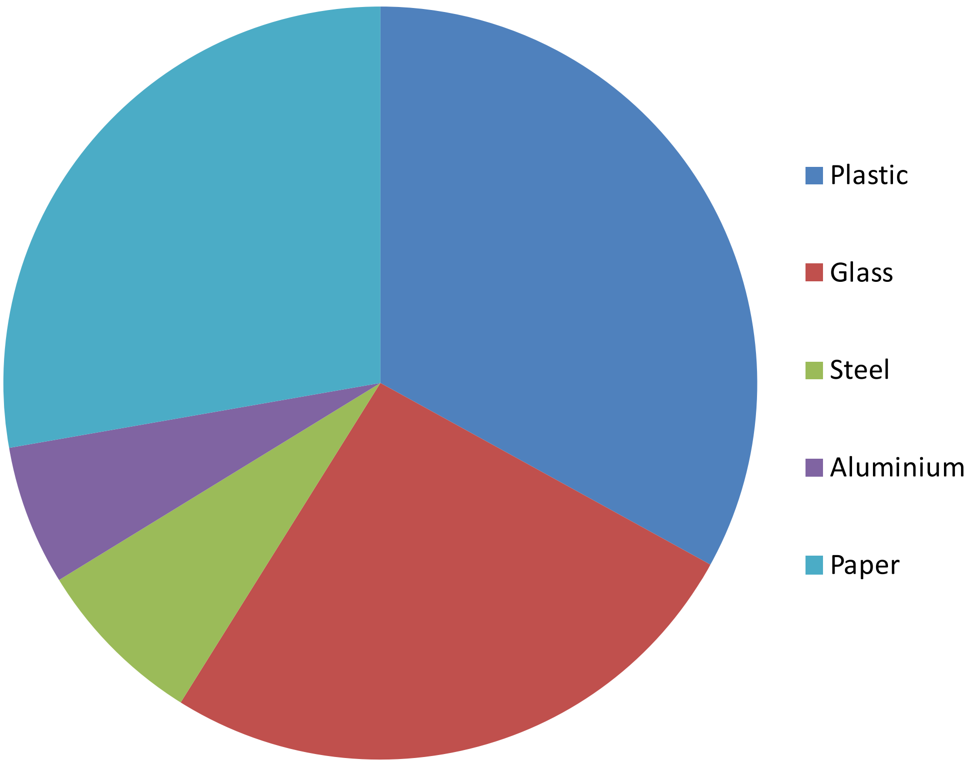

Figure 3. Typical revenue distribution of an MRF product stream

Although the steel and aluminum metals are only 4.4% based on weight but about 14% revenue would come from the value of these recovered metals. Therefore economic allocation would produce different results.

The GHG footprint of the MRF is shown in Figure 4 with the contribution from various unit processes such as trommel screen, front-end loader, conveyor, glass breaker, magnetic and eddy current separator.

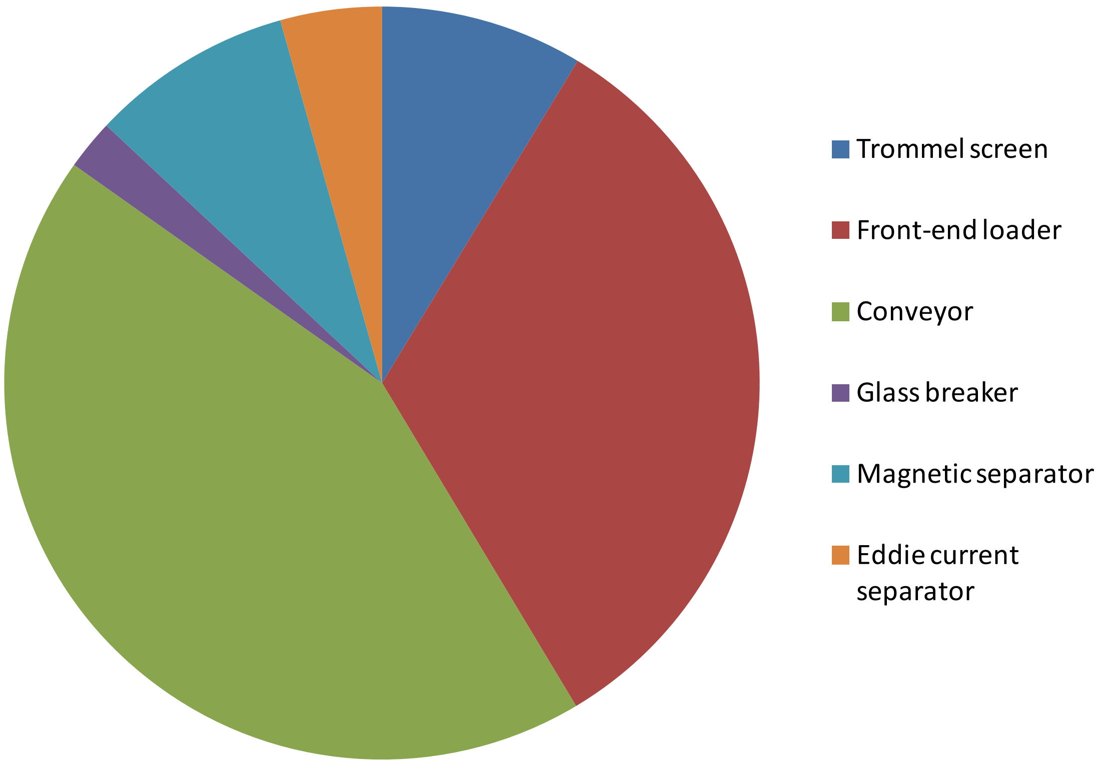

Figure 4. GHG footprints of MRF by contributing processes (16.5 kg CO_2-e/t of product produced). Except front-end loader (diesel energy), other unit processes are electricity based energy source.

The major contributing process of MRF on total GHG is conveyor motors (ca 43%) since there are five of them (relatively large number compared with other equipments). Front-end loaders (FEL) contribute to about 33%. Since the FELs have generally low productivity due to small specific material delivery rate and although diesel is used in FELs but the contribution is significant. Remaining GHG emissions are from the other equipments.

Steel

The result for GHG footprint of recycled steel is shown in Figure 5. The distribution from various stages along with international shipping contribution is shown here. For comparison, the GHG footprint of blast furnace and basic oxygen furnace steel making route is shown (Mathieson et al., 2012).

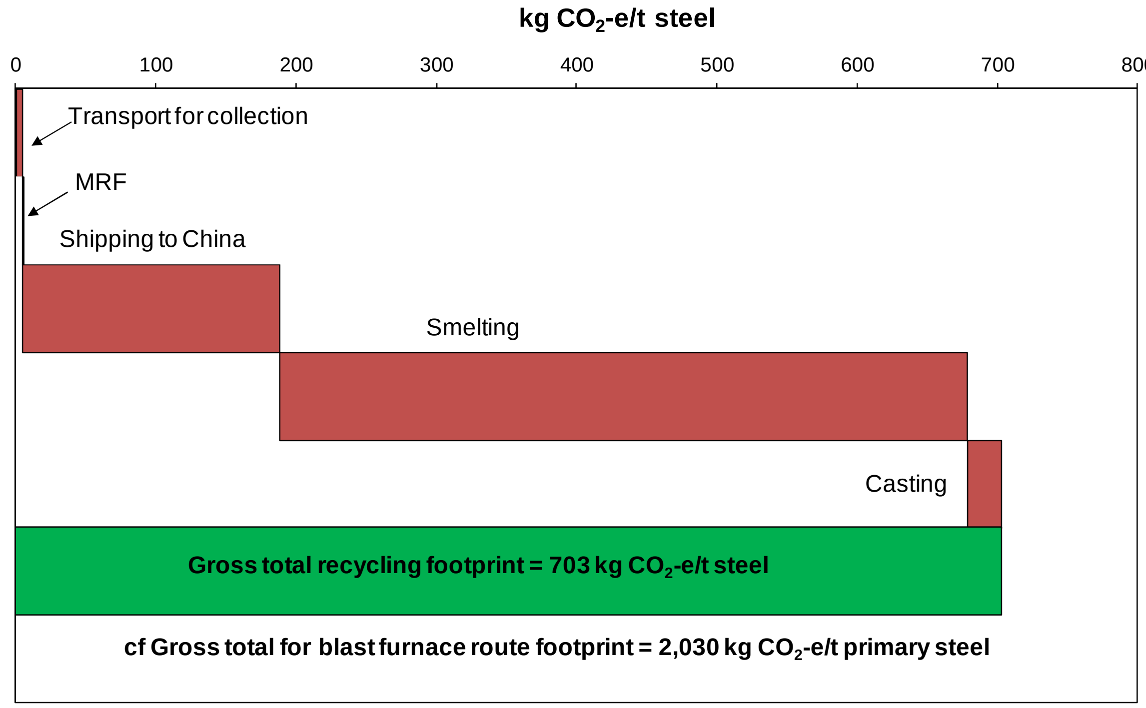

Figure 5. The GHG footprint of recycled steel scenario

This result shows that the collection of material and contribution from the MRF is relatively small part of the overall GHG footprint of recycled metal. This result also shows that if the sorted output product is sent overseas such as China for reprocessing, the contribution from shipping can be high. The GHG footprint of recycled steel is about one third of the primary steel production.

Aluminium

The results for GHG footprint of recycled aluminium is shown in Figure 6. The distribution from various stages along with international shipping contribution is shown here similarly to steel. For comparison, the GHG footprint of aluminium production in Australia from coal based electricity is shown (Norgate et al., 2007). It was assumed smelting of scrap aluminum requires 95% less energy (Aluminium International Today, 2013). The GHG footprint of aluminium casting is obtained from a previous study for remelt ingots scenario (Koltun et al., 2009).

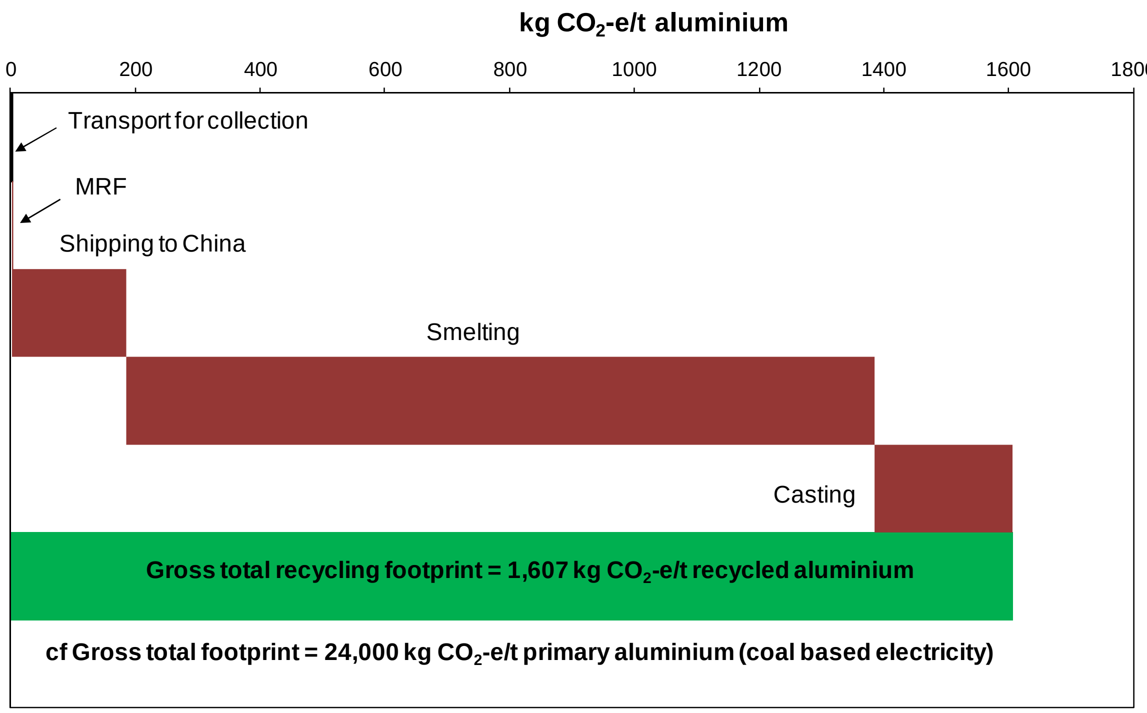

Figure 5. The GHG footprint of recycled aluminium scenario

This result shows that the collection of material and contribution from the MRF is a relatively small part of overall GHG footprint of recycled aluminium similar to steel. This result also shows that if the recovered metal is sent to China for reprocessing, the contribution from shipping can be high similar to steel. The GHG footprint of recycled aluminium is about 7% of the primary aluminium production. The difference between steel and aluminium is very high use of electricity during primary aluminium production. The contribution of electricity on the GHG footprint of primary aluminium is over 90%. The implication of this finding suggests that the impact of aluminium recycling is higher than that of steel because of higher use of electricity in the production of aluminium.

Comparison and assessment with some literature data

Norgate (2013) compiled specific energy input from the literature for waste collection. Fuel consumption for collecting and transporting waste materials (including metals) to a material recovery facility (MRF) is largely dependent on the duration of the collection route, which in turn depends on the source of the waste, e.g. city centre or suburban or regional areas, the lower the population density, the greater the transport distance between collection points. Another issue that affects collection energy is the type of collection system, e.g. single-stream (all materials combined) or dual stream (two streams, i.e. one for paper fibre and the other for commingled plastic, metal and glass).

The average GHG footprint of waste collection is reported to be 45.3 kg CO_2-e/t of waste. This is over two and a half times higher than that of MRF shown in Figure 4. It is difficult to compare since transport distances and scenarios may be different as mentioned above.

The average specific GHG of scrap metal sorting is estimated to be 33.8 kg CO_2-e/t scrap metal and 40.9 kg of CO_2-e/t scrap metal for shredding based on the specific input energy compiled by Norgate (2013). The GHG footprint of steel is estimated to be 520.3 kg CO_2-e/t steel and 494 kg CO_2-e/t aluminium based on the average specific energy inputs found in Norgate (2013). This result is the average of several values collected by Norgate (2013) from the open literature.

Conclusions

The GHG footprint of the MRF is relatively minor compared with that of associated transport during collection (i.e. 10 times more than MRF) of kerbside recyclable material. Additionally, if the bulk recyclable material is sent overseas (i.e. Australia to China) from the MRF for further processing, the GHG footprint of shipping can significantly be large compared with the sum of the collection and MRF (assuming electricity is from same source). Thus opportunities exist for reducing GHG emissions from secondary metal production if it is processed close to the MRF that can avoid large contribution of GHG emission from transport. This LCA result is preliminary and indicative only. Further uncertainty analysis of the key variables will be undertaken in future.

Acknowledgements

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