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ECO-EFFICIENT RECYCLING ALTERNATIVES FOR END-OF-LIFE CATHODE RAY TUBES

Jeremy Gregory[1], Randolph Kirchain[2]

[1]Laboratory for Energy and the Environment
[2]Department of Materials Science & Engineering and Engineering Systems Division
Massachusetts Institute of Technology, Cambridge, MA 02139

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Abstract

The disposal of end-of-life electronics is an issue receiving increasing scrutiny. This paper evaluates several alternatives for recycling cathode ray tubes (CRTs) within monitors and televisions. The primary recycling options for the glass, reuse in new CRTs or as a smelter flux, are examined from economic and environmental standpoints. The economics are modeled using a cost model that includes dependencies on processing costs, transportation logistics, and incoming material composition. The environmental consequences are modeled using life cycle assessment methodologies to determine the energy embodied in each processing alternative. Results map out preferred processing alternatives across a range of technological and operational conditions. The results indicate that economic costs and environmental consequences are heavily impacted by transportation. The preferred destination of CRT glass for a recycler is highly dependent on the proximity of the consumer to the recycler, although it is generally economically advantageous for the recycler to extensively dismantle the CRTs.

Introduction

Microelectronics and the products in which they are embedded symbolize the progress of the late 20th century. These innovations have created prosperity, but they have also driven new forms of consumption and inevitably associated waste streams. Although no recent data is available on the total amount of waste electrical and electronic equipment (WEEE) generated each year, a National Safety Council study estimated that 20.6 million computers became obsolete in 1998 and of those, only 11% were recycled [1]. Decreasing product lifespans and increasing consumption rates have likely increased the amount of WEEE generated since then.

This stream of materials is of concern not only because of its growing magnitude, but also because of its composition. Typical electronics goods comprise valuable non-renewable materials, such as precious and heavy metals, and toxic materials including Pb, As, and Hg. Waste cathode ray tubes (CRTs) have received particular scrutiny due to their lead content. Lead in electronics accounts for the largest fraction of lead entering the solid waste stream [2].

Regulators across the globe have begun to take an interest in WEEE. The most notable action is the EU directive 2002/96/EC, which requires original equipment manufacturers (OEMs) to collect and recycle their EoL products. In the US, several states have banned CRTs from landfills [3]. Other states choose to regulate CRTs as hazardous waste because of the lead content. Maine and California have recently implemented take-back policies for CRTs.

While there has been interest at the federal level in the issue of WEEE, the only action has been a proposed rule change by the EPA regarding the current classification of EoL CRTs as hazardous waste under the Resource Conservation and Recovery Act (RCRA) regulation [4]. The rule change would eliminate the hazardous waste classification for EoL CRTs and processed CRT glass that are to be recycled, reducing storage, labeling, and transportation requirements. As of the writing of this document, the EPA has yet to issue a final decision on the proposed change.

Because of the novelty of EOL electronics and the complexity of dealing with the commingled material content, the infrastructure for handling these products, including CRTs, continues to evolve, in terms of sophistication and effectiveness. The leaded glass in CRTs makes EoL processing particularly challenging. There are two main destinations for the CRT glass: smelters, which use the glass as a flux, or glass-to-glass (GtG) processors, who will sort and clean the glass and then send it to a CRT manufacturer for use in new CRTs. The glass typically has a low value because the primary material it is replacing, virgin silica, is a relatively low-cost material [5] and there is a surplus of secondary glass from containers [6]. Because of these challenges, CRTs are typically only processed for a fee, which can range from \$5-\$10 [5, 7].

The confluence of a relatively new recycling industry with emerging regulatory action and complex material processing issues motivates the research presented here, which aims to generate methods to facilitate data-driven decision-making for stakeholders in the WEEE recycling system. This is accomplished by developing models that characterize the economic and environmental implications of decisions made within the material recovery system, specifically the to-the-smelter vs. to-the-GtG processor decision. The objective of this work is to illuminate methods that will enable an economically robust recovery system in the face of environmental goals and regulatory change, thereby aiming for "eco-efficiency" – maximizing environmental benefit and minimizing costs.

Many techniques have been proposed to assess recycling using economic and/or environmental criteria [e.g., 8, 9]. Environmental issues are often calculated using some form of life-cycle assessment (LCA) and economic issues typically include processing costs. Other researchers have used operations research-type methodologies to examine efficient processing strategies [e.g., 10, 11]. While precise, the latter are highly data-intensive and, often, too complex for processors to use. The recyclability assessments suffer from the issue that they examine only fixed recovery scenarios. Although the assessments are useful for those scenarios examined it is difficult to understand how changes to the constraints affect the system. This is critical because of the dynamic nature of the system within which dismantlers operate.

This research seeks to fill this void by exploring the impacts of uncertainty and processing location on the economics that drive operational decisions and the associated environmental impacts. The desired outcome of the work is to develop a methodology that will enable these issues to be explored; material recovery from CRTs is presented here as a case study to demonstrate modeling capability. The modeling takes on the perspective of a processor dealing with multiple product types, not just one -- typical in other eco-efficiency models.

The paper is organized as follows. First, the CRT composition and recycling process are described. Then, the methodology and results for the economic analyses are presented, followed by methodology and results for environmental analyses. Finally, recommendations are given for stakeholders in the WEEE processing system regarding both the application of such modeling methods and with respect to the case at hand.

CRT Composition and Recycling Process

Figure 1 depicts the major components of a "bare" CRT; that is, a CRT that has had its housing and other exterior components removed. The glass typically accounts for over half of the weight of the monitor and the panel, or screen, glass accounts for over half the weight of the CRT [12]. The panel glass contains small amounts of Pb, on the order of 0-4% by weight, whereas the composition of funnel and neck glasses are on the order of 22-28% and 30% Pb, respectively [13]. The panel and funnel glasses are connected using a solder glass, or frit, that is approximately 85% Pb [7]. The panel glass and funnel glass have coatings on their inner surfaces: a phosphor coating and a conductive coating, respectively. The entire tube is a sealed vacuum, which must be compromised to remove the shadow mask and electron gun.

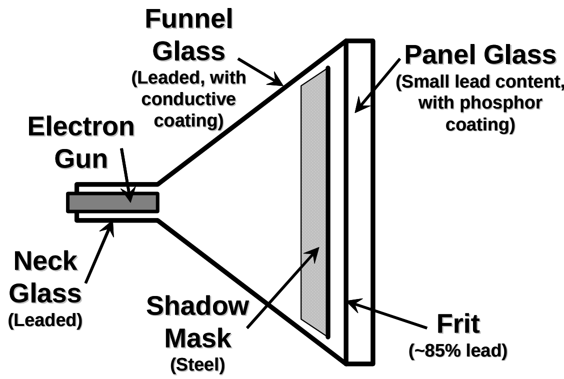

Figure 1. CRT composition.

CRT recycling can be accomplished using one of two major alternatives: shredding or disassembly. The former refers to shredding the entire monitor or television, typically minus the cord. The metals are separated using standard electro-magnetic and density methods, leaving two remaining material streams: glass and electronics shredder residue (ESR). ESR is predominantly plastics and as such, it can be sent to a processor who will recover most of the plastics, landfilled or incinerated. When shredded, the glass stream is a mixture of panel and funnel glass and is unsuitable for reuse in new CRTs. Firm-specific glass specifications vary considerably [6], driving strict uniformity constraints on secondary cullet, particularly for use in panel glass. As a result, the glass from the shredder is typically sent to smelters for use as a flux. The primary advantage of shredding is its high throughput and low labor costs; albeit, at a price of high capital costs.

Disassembly is an inherently labor-intensive process. Sequentially, housings (typically plastic) and the exterior components such as the yoke (coils of copper wire around the neck that act as an electromagnet) and banding (metal reinforcement around the funnel) are removed, leaving the bare CRT. If the glass will be used in new CRTs, the vacuum in the CRT is compromised (often by sawing off the neck, including the electron gun), the panel is separated from the funnel, and the shadow mask is removed. The glass is then crushed, cleaned, and sorted in an automated process that produces separate streams of glass that can be used in new CRTs. Some processors can produce several streams that vary according to glass composition and manufacturer type.

Although the disassembly process for CRTs is fairly straightforward, it rarely occurs at one location if the glass is to be used in new CRTs. This is because the equipment for cleaning and sorting the glass is quite specialized, thereby making the process reliant on large volumes. The more common scenario involves a first tier processor, referred to here as a dismantler, who will receive a whole monitor or television, disassemble it to a certain form, and then send the resulting material streams to other processors. The next level of processors, in this case, second tier processors, includes smelters and glass-to-glass processors.

Analysis Scope

There are many alternatives available to a first tier processor who is recycling CRTs. Figure 2 summarizes the alternatives specifically related to the CRT glass stream. The first decision that a processor must make is whether to shred or disassemble the monitor or television. If shredding is chosen, then the mixed glass cullet is most likely sent for use as a flux. However, if the processor disassembles the CRT, a decision must be made regarding the extent of dismantling and the destination for the glass. In most cases, these two decisions are interdependent and driven by three main factors: material, transportation, and processing costs.

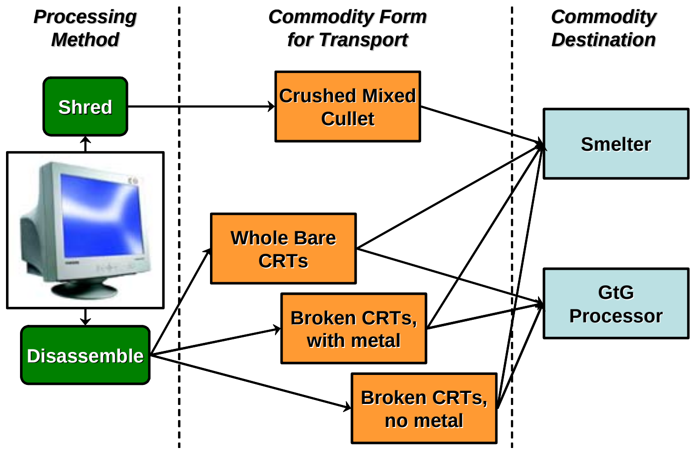

Figure 2. Recycling alternatives for CRT glass from Monitors/TVs for a first tier processor.

This study will focus on the disassembly route and, in particular, the decision of whether to send recovered glass to a smelter or GtG processor. The decision involves not only a glass destination, but also the form in which the glass should be processed. There are three main forms in which glass will be accepted, aside from whole CRTs: whole bare CRTs, broken CRTs with metal, and broken CRTs without metal [4, 7]. Material, processing, and transportation costs are all dependent on the form of the commodity that is shipped and the destination.

The dismantled CRT is the starting point for the study because initial disassembly is the same regardless of final destination. Specific questions examined include: What are the cost drivers? What conditions make a destination attractive? What is the most effective form for transporting CRTs (i.e. whole or broken)? Ultimately, it is important to determine which conditions trigger a change in processing decisions.

Economic Analyses

Modeling Methodology

The economic model developed calculates three components of the total cost associated with recovering glass from the CRT: transportation, material, and processing. The transportation cost is based on trucking rates for hazardous and non-hazardous materials, distances from dismantlers to glass consumers, and truck filling – by weight or volume. It is assumed that all transportation is by full trucks and that broken CRTs fill a truck by weight, whereas whole CRTs fill by volume. The hazardous/non-hazardous classification is included to enable examination of the proposed EPA rule change on CRT waste. Although transportation costs are dependent on the exact route, it is difficult to incorporate such specific cost information into a broad-based sensitivity analysis. Thus, average costs from the literature are used [4, 14, 15], shown in Table I; hazardous waste rates are 30% higher than non-hazardous rates [4]. Baseline distances are fictional and somewhat unimportant given that variation in distance is thoroughly examined.

The material costs as calculated in the model are dependent on 1) the type of monitor processed (and its material content) and 2) the rates that smelters and GtG processors charge for CRT-derived glass. The latter depend on the form of the glass. The rates in Table I are based on several sources in the literature [3, 4, 7, 16, 17]. Any metal collected after breaking the CRT (i.e., the shadow mask) provides positive revenue that is assumed to be \$0.33/kg (\$0.15/lb).

Processing costs include labor, allocated capital – equipment and facility, overhead and maintenance expenses associated with, when applicable, breaking or crushing the CRT and separating the metal. Breaking costs are applicable to both of the broken commodity forms and the metal separation costs are applicable only to the broken without metal commodity form. The values listed in Table I are representative estimates from industry sources; in future versions of the model these costs will be determined through process-based cost modeling.

Table I. Baseline input values for analyses. Values listed in SI units with US units in parentheses.

Transportation

Full truck by weight, kg (lb)	18,182 (40,000)
Full truck by volume, kg (lb)	13,636 (30,000)
Hazardous Transport Cost, $/km ($/mi)	$1.61 ($2.60)
Non-hazardous Transport Cost, $/km ($/mi)	$1.24 ($2.00)
Distance to Smelter, km (mi)	805 (500)
Distance to GtG, km (mi)	805 (500)

Material

	Destination	
Commodity Form	**Smelters**	**GtG**
Whole bare CRTs, $/kg ($/lb)	$0.22 ($0.10)	$0.22 ($0.10)
Broken CRTs, with metal, $/kg ($/lb)	$0.15 ($0.07)	$0.11 ($0.05)
Broken CRTs, no metal, $/kg ($/lb)	$0.15 ($0.07)	$0.00 ($0.00)

Processing

Breaking Costs, $/kg ($/lb)	$0.04 ($0.02)
Metal Separation Costs, $/kg ($/lb)	$0.04 ($0.02)

The mass profile of the CRT analyzed in this study is from [18], but the model is capable of analyzing processing of many different types of CRTs simultaneously. An annual processing volume of 50,000 CRTs is assumed. This has little impact on the total processing cost because dismantling activities have small economies of scale and are here assumed to be independent of volume. The processing volume has an impact on the total transportation cost.

Results

Figure 3 shows the total recycling costs for smelter and GtG destinations and the three commodity forms using baseline inputs and hazardous transportation costs. For this scenario in which the smelter and GtG are equidistant from the dismantler, the total recycling cost is smallest for broken CRTs without metal sent to the GtG facility. Indeed, the total cost decreases as more processing is done, despite extra processing cost. This is driven by two effects. First, transportation costs are less for the broken CRTs because they are filling trucks by weight. The second and larger cost driver is that the broken glass incurs much lower material fees. In fact, the broken CRTs without metal actually generate some revenue from material sales.

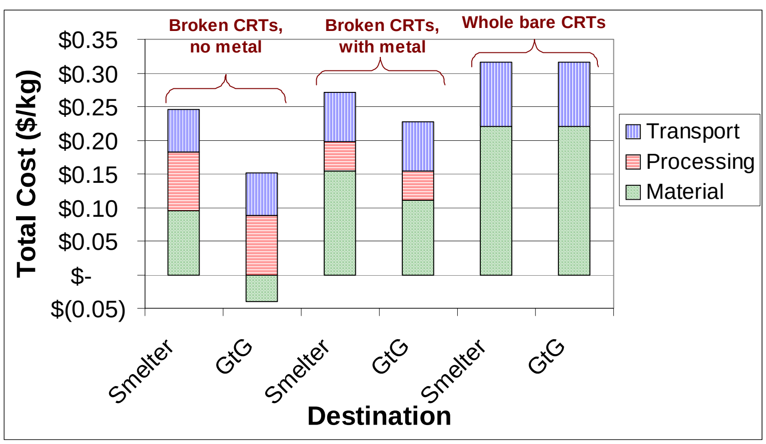

Figure 3. Total recycling costs for smelter and GtG destinations and three commodity forms using baseline inputs and hazardous waste transportation costs.

Although base-case results (cf. Figure 3) are informative, it is unlikely that these specific inputs will match the operating conditions for a particular dismantler. Thus, it is important to understand the implications of different processor conditions. To map the impact of transportation distance, two hypothetical dismantlers, one on the West Coast in the Silicon Valley and the other on the East Coast in New York City, are used as test cases. Distances from these dismantlers to glass processors are shown in Table II. The "current glass processors" are an actual GtG processor and smelter that accept CRT glass. Although there are nearly 20 lead smelters operating in the US, only a few currently accept CRT glass. However, the EPA rule change discussed previously, may lead other smelters to accept CRT glass. As such, the "potential glass processors" listed are actual smelters near the hypothetical dismantlers.

Table II. Distances from two dismantlers to glass processors. Distances are listed in km and miles in parentheses.

Dismantler Location	*Current Glass Processors*		*Potential Glass Processors*	
	GtG: PA	**Smelter: MO**	**Smelter: LA area, CA**	**Smelter: NYC area, NY**
San Jose, CA	4,830 (3,000)	3,220 (2,000)	564 (350)	
New York, NY	322 (200)	1,610 (1,000)		161 (100)

Given the number of input variables, it is difficult to explore the impact of changes transport, material, and processing costs on the glass destination decision. Figure 4 presents an attempt to resolve this difficulty. It is a map of the preferred destination for broken CRTs without metal for a range of distances and material costs (hazardous transportation rates). The x axis is the difference between the material fee for a smelter destination, which is variable, and the material fee for a GtG processor, which is fixed. Similarly, the y axis is the difference between the distance to a smelter, which is variable, and the distance a GtG processor, which is fixed. Choosing different baseline material costs or distances does not impact the results; it merely shifts the region of the map that is viewed by shifting the values on the axes. The region that is "preferred" is determined by the destination option with the lower associated cost.

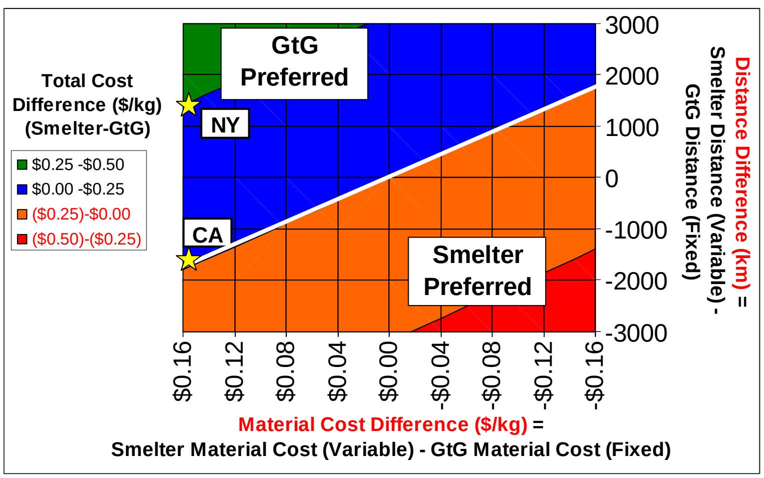

Figure 4. Map of preferred destination of broken CRTs without metal for a range of distances and material costs (hazardous material transportation rates).

The plot includes points that represent the current operating conditions for the New York and California dismantlers. For instance, the baseline difference in material costs is $0.15/kg and in distance is -1610 km for the California dismantler (the negative value indicates that the smelter is closer) and 1288 km for the New York dismantler. The close proximity of the New York dismantler to the GtG makes the destination a clear choice. For the California dismantler, the GtG processor is further away. However, the lower material fees make the total recycling costs nearly identical for both destinations. They also show that if the smelter decreased the cost it charges for glass it would become a more attractive alternative.

Figure 5 shows a map of preferred destination of whole bare CRTs using non-hazardous transportation rates. Unlike the previous figure, this plot shows that by using "Potential Glass Processors", smelting is clearly the preferred alternative for the California dismantler and is slightly the preferred alternative for the New York processor. Part of this is driven by the fact that the prices charged by the GtG processor and the smelter are the same, but it is also driven by the fact that the smelters are closer to the dismantlers.

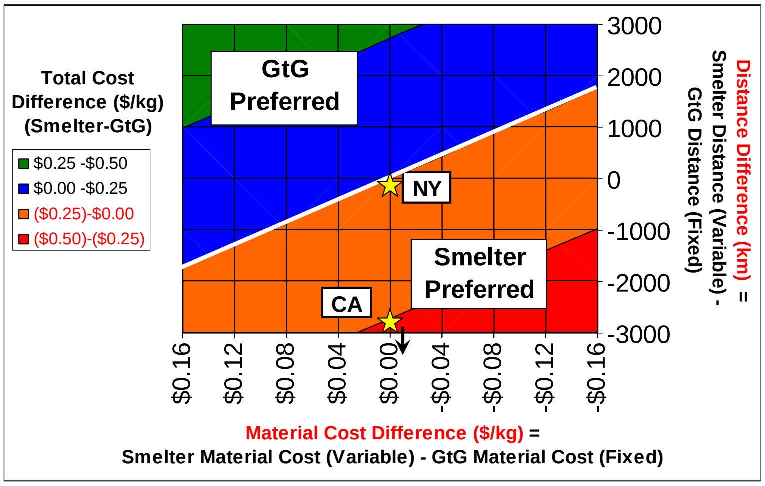

Figure 5. Map of preferred destination of whole bare CRTs for a range of distances and material costs (non-hazardous material transportation rates).

Plots similar to Figure 4 and Figure 5 can be created for all commodity forms and various transportation cost scenarios. Table III is a summary of the preferred alternatives across relevant scenarios. The additional distance to the GtG processor for the California dismantler makes it unattractive, with the exception of for the broken without metal commodity form. This additional distance would only increase if a smelter in California accepted CRT glass. For the New York dismantler, the close proximity to the GtG processor makes it an attractive alternative. It is interesting that if a New York smelter started to accept CRT glass, thereby making it closer to the dismantler than the GtG processor, it would only be the preferred alternative for one out of the three commodity forms.

Table III. Summary of preferred alternatives for both dismantlers for a given commodity form and transportation scenario, assuming baseline material, transport, and processing input costs.

Commodity Form	***CA Dismantler***		***NY Dismantler***	
	Current Processors	**Potential Processors**	**Current Processors**	**Potential Processors**
Broken CRTs, no metal	GtG	Smelter	GtG	GtG
Broken CRTs, with metal	Smelter	Smelter	GtG	GtG
Whole bare CRTs	Smelter	Smelter	GtG	Smelter

Environmental Analyses

<u>Modeling Methodology</u>

The environmental component of this analysis was computed using a simplified version of life cycle assessment (LCA). Using the disassembled bare CRT as a point of reference, the energy required to process the CRT glass was computed for each destination alternative and the material replaced. For example, the energy required to make glass from recycled (secondary) cullet was compared with the energy required to make glass from virgin materials (primary). For the smelter alternative, the CRT glass is replacing sand as a flux and thus, the energy required to acquire sand is compared with that to acquire the CRT glass. Data for these calculations came from the Ecoinvent® database within the SimaPro® LCA software. The database focuses on European data, but is expected to be representative for the processes considered.

Although environmental impact metrics are available, energy consumed was chosen as representative of a significant portion of the environmental impact and because it does not require any weighting scheme. In this study, the scope includes the energy required to acquire, process, and transport materials and the energy required to make the equipment and infrastructure that is used in those processes. In the case of secondary materials, the energy involved in sorting processes is included as well.

Packaging glass was used as an approximation of CRT glass because of data availability. The secondary glass production was based on 60% recycled glass content and 40% virgin content. This should be representative of CRT glass which is not made purely from secondary glass. The energy consumed in CRT glass going to a smelter is assumed to include the energy for sorting and transportation only. The transportation component of the LCAs for the secondary materials from the dismantler to the GtG processor or smelter is a variable component of these analyses so that the dependence of the environmental impact on this parameter can be studied.

Results

Figure 6 depicts the difference between the energy required for primary and secondary glass production as a function of the distance between the dismantler and the GtG processor. The distance between raw materials and new CRT glass production is assumed as a fixed distance.

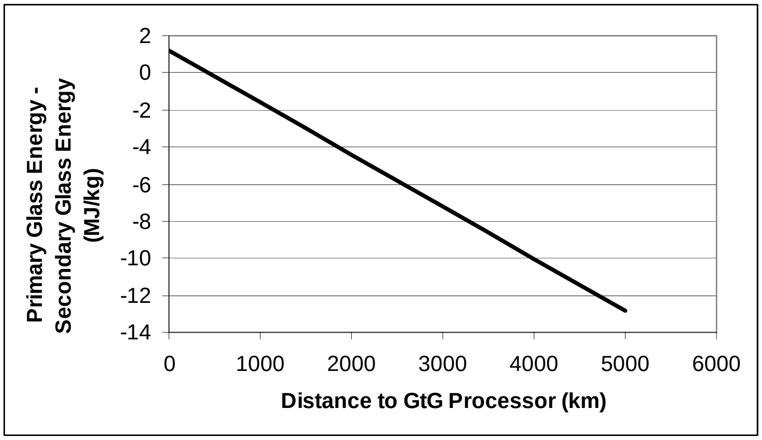

Figure 6. Energy difference between primary and secondary glass as a function of distance to GtG processor.

It is clear that transportation has a significant impact on the energy in secondary glass production. The difference between the primary and secondary glass production with zero secondary transportation distance is 1.2 MJ/kg. Every 1000 km of transportation adds 2.81 MJ/kg. Thus, the primary and secondary glass energy is equal at approximately 425 km. The New York dismantler is 322 km from the GtG processor, which is within this distance, but the California dismantler is 4,830 km from the GtG processor, which represents a secondary glass energy requirement of nearly 13 MJ/kg greater than the primary glass energy requirement.

Figure 7 is analogous to Figure 6, except it applies to a smelter rather than a GtG facility. It plots the energy difference between sand acquisition and secondary glass sorting and it plots this as a function of the distance from the dismantler to the smelter *and* the distance from the sand acquisition location to the smelter. The distance from the sand acquisition location to the smelter is included as an additional variable in this plot because this is an unknown. By contrast, the LCA for the primary glass production was based on a typical profile of a facility that is located a fixed distance from its raw materials sources.

The energy difference between the sand acquisition and secondary glass sorting is 0.22 MJ/kg (0.33 MJ/kg for acquisition and 0.11 MJ/kg for sorting) with no transportation. The magnitude of transport energy drives the sand/sorting difference, giving advantage to the closest facility.

As in the economic analyses, the objective in the environmental analyses is to compare the impacts of the smelter and GtG processor routes. Figure 8 accomplishes the comparison by plotting the energy difference between the smelter energy difference (sand acquisition, assuming 500 km of transportation – secondary glass sorting) and GtG energy difference (primary – secondary) as a function of distance to smelter and distance to GtG processor. If the

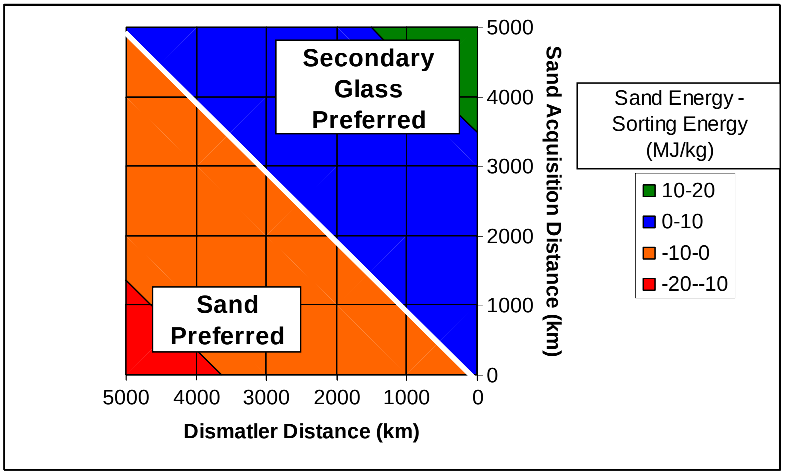

Figure 7. Energy difference between sand and secondary glass sorting as a function of distance from dismantler to smelter and distance from sand acquisition location to smelter.

environmental objective is to encourage recycling, then the smelter and GtG energy differences should be maximized. Figure 8 plots the difference between these two energy differences; the higher positive value or the smaller negative value is the preferred choice.

It is clear from Figure 8 that transportation distance is the dominant determinant of preferred route If the distance to the smelter is longer, then GtG is preferred, and vice versa. For the New York dismantler, the GtG route would be preferred from an environmental standpoint, whereas the opposite would be true for the California dismantler.

It is important to note that any energy-based metric does not assess the impact of toxicity. Thus, the effects of lead in the CRT glass are underassessed. Future research will include comparisons of different LCA metrics to discern which are best suited to analyses involving WEEE.

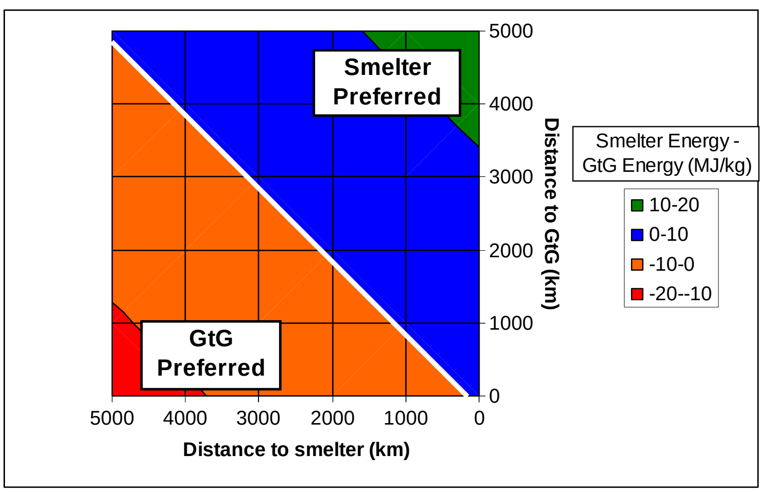

Figure 8. Energy difference between the smelter energy difference (sand acquisition, assuming 500 km of transportation – secondary glass sorting) and GtG energy difference (primary – secondary) as a function of distance to smelter and distance to GtG processor.

Conclusions

The results of this work show that the proximity of recovery destination is a large driver in the economic and the environmental impacts of dismantler processing decisions. Nevertheless, from an economic perspective, the closer alternative is not always attractive because of changes in material and processing costs. Economic analyses also indicate that it is generally advantageous for the recycler to spend resources to dismantle the CRT for maximum material revenue.

An important outcome of this work is that processors must continue to examine the range of alternatives for current and potential operational scenarios. Using a processor on the East and West Coasts as case studies, it was shown that the preferred destination for CRT glass can change depending on the commodity form that is transported and could also change if the EPA changes the classification of CRTs for recycling from hazardous to non-hazardous waste.

Future research will focus on three areas: data collection for regional conditions, utilizing process-based cost modeling to project technology-specific processing costs, and comparing outcomes of environmental analyses using different LCA metrics.

Acknowledgements

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