MODELING TEMPORAL ALUMINUM MATERIAL FLOWS AND GREENHOUSE GAS EMISSIONS TO EVALUATE METALS RECYCLING ALLOCATION IN LIFE CYCLE ASSESSMENT

by

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To my family.

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ABSTRACT

Although dynamic, time-dependent aspects mark its life cycle, aluminum has largely been treated as a static system in industrial ecology. Life cycle assessment (LCA) and material flows analysis (MFA) continue to expand beyond their initial purpose of providing single point-in-time results, but remain limited in their ability to capture the temporal nature of aluminum. As a result, this dissertation has developed more comprehensive and robust approaches for evaluating greenhouse gas (GHG) emissions and material flows of aluminum production, consumption, and recycling over time.

Dynamic MFA and LCA approaches are developed to analyze the effects of economic and technological trends on U.S. aluminum in-use stocks and global absolute and relative GHG emissions from primary aluminum production. A dynamic MFA model is developed to estimate in-use stocks and recovery from 1900 to 2007. Results show that 34% of apparent consumption since 1900 remains as in-use stocks in 2007. Time series analysis is used to quantify the relationship between gross domestic product and net additions to in-use stocks. A dynamic LCA is developed to quantify the spatial and temporal variation in the life cycle GHG emissions of global primary production, consumption, and trade from 1990 to 2005. Seven world regions are shown to have distinct GHG intensities; the largest difference in 2005 is between Asia (21.9 kg CO2-eq/kg) and Latin America (7.07 kg CO2-eq/kg).

The analysis of economic and technological trends is also used to provide a critical evaluation and counterargument for the metal industry's position that metals are: widely recycled, recycled many times over, and constrained in secondary production by scrap availability. The position that primary metal production is displaced by secondary production is put into question by analyzing the U.S. aluminum market.

Lastly, four LCA recycling allocation approaches are evaluated for their capacity to accurately reflect the temporal nature of aluminum. The recycled content approach is

recommended based on its ability to accurately account for the timing of material flows and GHG emissions, and to be used in a consequential LCA framework. Where appropriate, this approach should be extended with systems expansion methods that are based on sound economic theory.

CHAPTER 1

INTRODUCTION

1.1 Motivation

Like in other scientific disciplines, the primary tools of industrial ecology – life cycle assessment (LCA) and material flow analysis (MFA) – have been adapted to meet the evolving needs of their practitioners. LCA was first codified under the International Organization for Standardization (ISO) standard 14040 (1998) as a method that provided a static evaluation of the environmental performance of a system from raw material extraction to final disposal. In a similar way, MFA initially furnished a single point-in-time account of the masses of a material utilized throughout a specified system (Bringezu and Moriguchi 2002).

LCA and MFA were initially without dynamic elements and were firmly rooted in the natural and applied sciences from which they grew. Both tools emphasized the quantitative description of the physical inputs and outputs using material and energy balance principles of thermodynamics. LCA included the assessment of environmental impacts using approaches developed in the environmental sciences.

These analytic foundations of LCA and MFA remain; however, LCA in particular has taken on new complexities (e.g., spatial and temporal differentiation, rebound effects, scenario analysis) and has been extended to incorporate economics and other social sciences (Finnveden et al. 2009; Heijungs et al. 2009; Heller and Keoleian 2000). Many of these new extensions have come about from demands of sustainability analysis, which encompasses social and economic, as well as environmental, dimensions (Heijungs et al. 2009).

Advancing MFA and LCA methodology is crucial for a more comprehensive evaluation of aluminum flows and greenhouse gas (GHG) emissions. Focusing on aluminum is important because it is the second most-widely consumed metal

(International Aluminum Institute 2010a) and has a large electricity intensity of production. Producing a metric ton of primary aluminum requires 15,215 kWh of electricity on average (IAI 2010b). The combination of large global demand and high electricity consumption, as well as process emissions of high global warming potential perfluorocarbons (PFCs), results in primary aluminum production contributing to 0.93% of world GHG emissions in 2004 (McMillan and Keoleian 2009). Because of aluminum's importance as a material and as a source of GHG emissions, new MFA and LCA methods that can better characterize aluminum are valuable contributions for the overall development of nonrenewable natural resource and GHG policies.

This dissertation contributes to the evolution of LCA and MFA by expanding their methods to include dynamic, time-dependent elements. The advancements presented in this dissertation are responses originating from the research demands of National Science Foundation Materials Use: Science, Engineering, and Society grant #0628162. This project aims to develop novel methods and tools to evaluate changes in material flows arising from potential policy instruments to reduce greenhouse gas (GHG) emissions from automobiles and light duty trucks. The project establishes a predictive framework for understanding potential impacts of market responses on material flows and environmental emissions. This framework served as the genesis for research that incorporates temporal considerations in the evaluation of the industrial ecology of aluminum and other metals.

1.2 The Temporal Nature of Aluminum as a Metal and its Importance for Understanding Life Cycle Material Flows and Emissions

Time is an essential aspect of the life cycles of aluminum and aluminum products, as well as the aspects of many of their associated environmental emissions and impacts. In addition to the inherently temporal nature of a life cycle, intertemporal decisions shape where and when aluminum is produced, consumed, retired, and recycled. While LCA and MFA have largely ignored the temporal nature of aluminum and other metals, economic theory has long provided insight into the intertemporal decisions that guide the behavior of producers and consumers. If LCA and MFA are to advance in their ability to describe aluminum and other metals, temporal considerations must be addressed.

The life cycle of aluminum begins with the decision made by mining companies to extract mineral bauxite. The optimal rate of extraction across time is described by Hotelling's rule (Hotelling 1931). Hotelling's rule establishes that in the dynamic efficient extraction of an exhaustible resource the present value of marginal profit is equal in each time period; alternatively, the change in marginal profit between two time periods is set equal to the interest rate.

Other intertemporal decisions, such as the profit or net price expectations of primary aluminum producers, have shaped the geographic distribution of primary aluminum production. Because primary aluminum production is electricity intensive and electricity costs on average constitute more than one-third of input costs (Gagné and Nappi 2000),"the price of electric power is the major determinant of the international competitiveness of smelters" (Peck 1998, 14). The oil shocks of the 1970s led to restructuring in the United States and Western Europe – the initial regions of primary aluminum production – and growth of new capacity in regions with lower electricity costs (e.g., Latin America and Australia).

This expansion of production locations corresponded to a period of increasing global trade and economic growth in developing countries. The confluence of these trends has magnified the separation between where primary aluminum is produced and where it is consumed. Most notable is the case of China, which has experienced a large growth in aluminum consumption as the result of a surging economy. In 1990, Asia as a whole relied on imports for 59% of its consumption of primary ingot (International Aluminum Institute 2007; United Nations Statistics Division 2007). However, this trend was reversed due to significant smelter construction and by 2005, Asia as a whole relied on imports for only 27% of its consumption of primary ingot (International Aluminum Institute 2007; United Nations Statistics Division 2007).

Along with these geographic shifts came changes to the environmental impact of primary aluminum production. The pursuit of the lowest-cost source of electricity generation has contributed to the emergence of regionally distinct GHG intensities of production. In addition, changes in technology, such as decreased electricity intensity and improved alumina feeding technology, have further affected the regional character of primary aluminum emissions. Coupled with changing trade patterns, the GHG intensity of primary aluminum production in a given region is potentially different from the GHG intensity of aluminum consumed in the same region. However, life cycle inventory data have largely ignored spatial and temporal variation of GHG emissions intensities of production, as well as the influence of international trade. GHGs embodied in trade have been quantified using input-output tables (Ahmed and Wyckoff 2003; Munksgaard and Pederson 2001; Peters and Hertwich 2008; Weber and Matthews 2007), but this approach has not been used on the level of an individual commodity or used to provide annual estimates. Recent debate on levying carbon tariffs for goods imported from countries without emissions caps has brought regional variation in GHG intensity to the forefront. As a result, an improved quantification of the variability of primary aluminum emissions has important implications for the development of climate and international trade policies.

Aluminum supply and demand are also based in part on intertemporal considerations described by theories on stockholding behavior. Because aluminum is a storable commodity (i.e., a good that is not perishable), suppliers and consumers can choose to hold stocks based on expected changes in price and to buffer against stochastic shocks to price and other market conditions. Stocks also play a role in market price formation by equilibrating supply and demand. Stocks provide this necessary function due to the difficulty of adjusting aluminum production in the short term.

Aluminum demand is tied not only to overall economic activity, but also to new product development (Brubaker 1967). New applications that emerge and change the overall aluminum product mix over time have implications for consumption, composition of in-use stocks, and generation of old scrap. Rolled and extruded aluminum products in the form of doors, window frames, high-voltage transmissions wires, consumer durables, and aircraft fuselages constituted much of the initial growth in aluminum consumed and stored as in-use stocks within the U.S. economy. By the 1980s these markets were largely saturated and as growth stalled the next wave of aluminum took the form of cast components for the automotive market.

The variation in product mix over time has also changed the composition of in-use stocks. This, along with changes in the relative demand of primary and secondary aluminum, has implications for the recovery of old scrap from discarded products and the energy and environmental profile of aluminum consumption. Old scrap recovery is governed by the ability of the existing recycling infrastructure to economically recover scrap. Products that yield high quality, highly concentrated scrap are a more attractive source of metal than products that yield contaminated, dilute scrap. Even with products that are theoretically excellent sources of old scrap, the characteristics of the recycling system play a large role in ultimate recovery. Aluminum beverage cans, which contain 95% aluminum, reached their peak recycling rate in the U.S. of 65% in 1992; this rate then proceeded to fall to 45% by 2004 (Container Recycling Institute 2010). Overall, from 1972 to 2007 it is estimated that the U.S. recycled 25% of its discarded aluminum products.

Although dynamic MFA models have been developed for aluminum in Germany (Melo 1999) and other metals (e.g., lead (Mao and Graedel 2009), iron (Müller et al. 2006), and copper (Zeltner et al. 1999)), most previous MFA models developed for aluminum in the U.S. have been constructed as single point estimates. Recalde et al. (2008) developed a model of aluminum stocks for the state of Connecticut in the year 2000. The authors utilized a bottom-up approach, gathering data on the aluminum composition of products consumed within the state. A summary of aluminum flows in the U.S. in 2000 was described by Plunkert (2006), and Sullivan (2005) estimated the amount of in-use aluminum stocks. While these estimates do provide an assessment of current conditions, they are unable to describe historical dynamics of growth or to forecast future scenarios. Additionally, while some (e.g., Iriarte-Goñi and Ayuda, 2008; Friedl and Getzner, 2003) have addressed the time series properties of data, discussion and testing of data stationarity remain absent in other studies of dematerialization and material intensity of use (e.g., Vehmas et al., 2007; and Canas et al., 2003).

Economy-wide changes to the aluminum product mix over time affect the relative demand of primary and secondary aluminum. This is the result of the requirement of different product applications for aluminum to be alloyed for specific physical properties. These distinct alloys differ in their tolerances for the presence and concentration of contaminants and alloying elements, which dictates the acceptability of primary and secondary aluminum as source materials. Most generally, the tolerances of wrought (i.e., rolled, extruded, or forged) alloys are incompatible with the composition of cast alloys

(Das 2006). Additionally, whereas cast alloys contain mostly old scrap with some primary aluminum for dilution (Das et al. 2007), old scrap that is even clean and sorted is only a minor input for wrought alloy production (Kevorkijan 2002).

In addition to representing a change in the physical flows of aluminum, a new relative demand of primary and secondary aluminum also has important implications for energy and environmental emissions. Compared to primary production, secondary aluminum production consumes only 5% of the energy (International Aluminum Institute 2000), emits only a fraction of the GHGs, and does not contribute to the loss of a nonrenewable resource.

Products that utilize secondary aluminum thus have environmental advantages compared to products that rely on primary aluminum, yet the LCA community continues to debate the appropriate way to account for, or "allocate", these advantages. The metals industry has issued a declaration describing the characteristics of metals recycling that supports a single method of allocation for metals (Atherton 2007). Other allocation approaches have been developed based on physical relationships (Fava et al. 1991), economic relationships (Werner and Richter 2000; Vogtländer et al. 2001), and systems expansion principles (Ekvall 2000). What the metals industry's declaration and most of these proposed approaches lack, however, are quantitative evaluations of their assumptions and results. Work that has provided such an evaluation (e.g., Schmidt 2010; Nicholson et al. 2009; Thomassen et al. 2008; Ekvall and Andrae 2005) has not addressed how these approaches account for the timing of material flows and emissions.

Dynamic processes drive the functional flows of material, energy, and emissions in the aluminum life cycle. As such, these flows are analyzed best using methods and tools that capture their temporal nature. Introducing dynamic, temporal elements to both LCA and MFA provides a more robust analysis of aluminum, from its extraction to its recycling and ultimate disposal.

1.3 Research Objectives

This dissertation endeavors to answer the following research questions, which are organized by research topic area:

U.S. Aluminum Dynamic MFA Model

1. How has aluminum consumption between 1900 and 2007 affected the accumulation of aluminum in-use stocks and the recovery of aluminum old scrap?
2. How can quantitative time series analysis be used with MFA results to test for a statistically significant relationship between in-use stocks and economic output?
3. How might global sensitivity analysis be used to quantify model sensitivity?

Dynamic Primary Aluminum Life Cycle GHG Emissions

4. What are the absolute and relative life cycle GHG emissions associated with primary ingot production, trade, and consumption across the six world regions from 1990-2005?
5. How have the largest contributing factors to GHG intensity changed from year-to-year across regions?

Synthesis of Metals Recycling Characteristics

6. How is metals recycling in practice for aluminum, iron, and copper consistent or inconsistent with the position of the metals industry presented in Atherton (2007)?
7. Are the conditions of the U.S. primary and secondary aluminum markets from the early-1990s to 2007 consistent with secondary production offsetting primary production?

LCA Allocation Approaches for Aluminum Recycling

8. Under what conditions do the market-based (Ekvall 2000), value-corrected substitution (Werner 2005; Werner and Richter 2000), and end-of-life recycling (Atherton 2007) allocation approaches succeed and fail in accurately describing the temporal nature of metals recycling and GHG emissions?

1.4 Research Contributions

This work provides the following research contributions to existing knowledge:

1. U.S. Aluminum Dynamic MFA Model: The model estimates the annual accumulation of U.S. in-use stocks and recovery of aluminum for seven end-use categories from 1900 to 2007. The model also estimates several variations of total aluminum recycling rates. Model calculations are based on annual apparent consumption data, estimated product lifetime distributions, and estimated recycling and recovery rate for each end-use category. Data calculated by this

model are utilized in the U.S. GDP and Aluminum Net Addition to In-Use Stock Model and in the Synthesis of Metals Recycling Characteristics.

2. U.S. GDP and Aluminum Net Additions to In-Use Stock Model: This model quantifies a model of annual U.S. GDP and net additions to in-use stock (NAS) using statistical regression. Additional variations quantify the relationship between NAS and GDP disaggregated to the construction and transportation sectors. The development of the model includes unit root testing of NAS data, which is an infrequent, but necessary consideration when regressing time series MFA data.
3. Dynamic Regional Primary Aluminum GHG Model: Life cycle inventory data for aluminum is typically published every three to five years and only provides a single, world average value. The Dynamic Regional Primary Aluminum GHG Model estimates annual GHG emissions and emissions intensities of primary aluminum ingot production for six world regions from 1990 to 2005. Model results also include consumption-based estimates based on the calculated emissions embodied in exports and imports of primary ingot. Emissions intensities from this model are utilized in the Quantitative Analysis of LCA Allocation Approaches Applied to Aluminum.
4. Synthesis of Metals Recycling Characteristics: Existing literature on uses and recycling systems of aluminum, copper, and iron and steel are synthesized in a response to the metals industry's position paper on metals recycling (Atherton 2007). This synthesis is used to build an argument that the position of the metals industry oversimplifies the determinants of metals recycling and that metals are not necessarily recycled at high rates, recycled over and over again, and constrained in recycling beyond material availability.
5. Evaluation of Primary Metal Production Displacement by Secondary Metal Production: Trends in U.S. production, international trade, supply, and consumption of primary and secondary aluminum are analyzed to find evidence of whether or not a large increase in consumption of secondary aluminum was associated with a change in primary aluminum consumption. This evaluation aims to investigate the claim of the metals industry (Atherton 2007), which is not

accompanied by supporting evidence, that secondary metal production displaces primary metal production

6. Quantitative Analysis of LCA Allocation Approaches for Aluminum: Several LCA allocation approaches have been specifically proposed and advocated for application to aluminum and metals. The assumptions of the market-based and value-corrected substitution approach are evaluated in terms of their ability to capture the temporal nature of aluminum markets and aluminum prices. The end-of-life and recycled content approaches are compared using case studies of the U.S. aluminum beverage can and aluminum engine blocks in a hypothetical fleet of automobiles.

1.5 Organization

This body of research is presented in a multiple manuscript format. Chapters 2, 3, 4, and 5 are individual manuscripts complete with abstract and references. Chapter 2 presents the U.S. aluminum market dynamic MFA model and the U.S. GDP and NAS model. This manuscript is published in *Ecological Economics* (McMillan et al. 2010).

Chapter 3 presents the dynamic model of primary aluminum GHG emissions. This manuscript was published in *Environmental Science & Technology* (McMillan and Keoleian 2009a) and serves as the foundation of a conference proceedings abstract (McMillan and Keoleian 2009b). Additionally, emission factors from the model are utilized in a conference paper published by IEEE (Kim et al. 2008).

Chapter 4 consists of the synthesis of existing literature on the characteristics of aluminum, copper, and iron and steel recycling and an evaluation of the ability of secondary aluminum production to displace primary aluminum production. Chapter 4 has been accepted to the *Journal of Industrial Ecology* pending revisions.

Chapter 5 presents the quantitative analysis of LCA allocation approaches as applied to aluminum, which includes an evaluation of the assumptions of the market-based and value-corrected substitution approaches. This manuscript has not yet been submitted to an academic journal for review.

Figure 1.1 depicts the interrelationships of the four research topic areas and their location in the dissertation. The figure also serves as a basic material flow diagram and shows the typical flows of the aluminum life cycle.

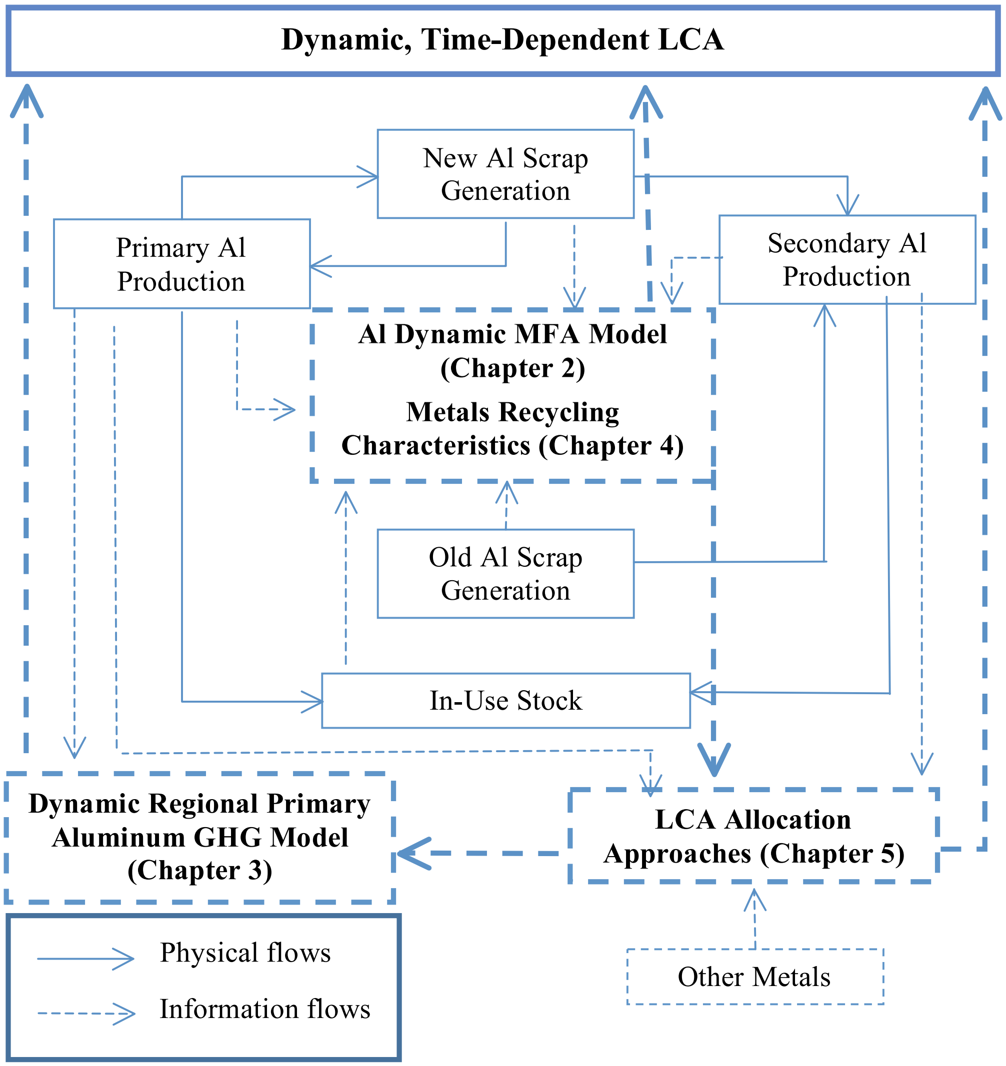

Figure 1.1 Research Framework

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CHAPTER 2

QUANTIFYING U.S. ALUMINUM IN-USE STOCKS AND THEIR RELATIONSHIP WITH ECONOMIC OUTPUT

ABSTRACT

A dynamic material flow analysis model is developed to quantify aluminum in-use stocks and old scrap recycling and recovery in the United States for the period of 1900 to 2007. The total in-use aluminum stock in 2007 is estimated as 93 million metric tons, which represents approximately 34% of the cumulative apparent consumption since 1900. Alternately, since 1900 nearly 40% of the cumulative discarded aluminum has not been recycled for domestic use in the U.S. or for export to foreign consumers. Statistical time series analysis is used to explore the relationship between model results of in-use stocks and gross domestic product (GDP). Unlike most previous studies of material consumption and economic activity, which ignore the statistical properties of time series data to the detriment of model estimation and inference, data stationarity is explicitly evaluated through unit root testing and model specification is adjusted accordingly. The annual percentage change in GDP is found to have a large and significant association with the annual percentage change in net additions to in-use stocks. Model sensitivity and uncertainty are quantified through the application of the Fourier Amplitude Sensitivity Test and alternate specifications of product lifetime probability density functions.

2.1 Introduction

The demand for aluminum has grown tremendously since the mid-1800s and its worldwide use now is exceeded only by steel (IAI, 2009). The significance of aluminum as an industrial metal and climate change concerns have focused attention on the environmental impacts of aluminum production. While producing aluminum from mineral bauxite (i.e. primary production) is recognized for its large energy intensity and greenhouse gas (GHG) emissions, aluminum produced from recycled metal (i.e. secondary aluminum) is notable for its much lower environmental impact. Because of aluminum's nature as a lightweight metal and the large difference between primary and secondary aluminum, two potential emissions mitigation strategies are to use aluminum in reducing the mass of appropriate products such as automobiles and to substitute secondary aluminum for primary aluminum. However, these strategies would require changes in the consumption of primary and secondary aluminum. The assessment of their feasibility should include the analysis of where potentially-recoverable aluminum resides in the U.S. economy and what drives its accumulation.

Material flow analysis (MFA) is a method of quantifying the mass of a material or product of interest as it moves throughout specified temporal and economic or geographic boundaries. A MFA is essentially a mass balance whose results are used to estimate intensity of use, in-use stocks, material recovery rates, and other aspects of the flows and stocks of materials within the chosen boundary (Bringezu and Moriguchi, 2002). When applied to an entire economy, MFA can provide information on the structure and dynamics of physical metabolism and resource productivity (Giljum et al., 2009).

MFA models can be developed for a single year, providing a static snapshot, or over multiple years, creating a dynamic analysis. Numerous static MFA models have been developed for a variety of materials and products. Other MFA studies have utilized dynamic models to calculate the changes of flows and stocks over time. These include an analysis of lead (Mao and Graedel, 2009), cement (Kapur et al., 2008), iron (Müller et al. 2006), and copper in the U.S. (Zeltner et al, 1999), iron and steel in the U.K. (Davis et al., 2007), and furniture in private households in Colombia (Binder et al., 2001). Assessment of the global industrial metabolism of metallic ores is included in work by Krausmann et al. (2009). Research has also been undertaken to forecast material flows in applications such as concrete in Dutch housing (Müller 2006) and global production of silicon (Williams, 2003).

Most previous MFA models developed for aluminum in the U.S. have been constructed as single point estimates. Recalde et. al (2008) developed a model of aluminum stocks for the state of Connecticut in the year 2000. The authors utilized a bottom-up approach, gathering data on the aluminum composition of products consumed within the state. A summary of aluminum flows in the U.S. in 2000 was described by Plunkert (2006), and Sullivan (2005) estimated the amount of in-use aluminum stocks. While these estimates do provide an assessment of current conditions, they are unable to describe historical dynamics of growth or to forecast future scenarios.

A dynamic MFA approach was recently applied to the U.S., Japan, Europe, and China by Hatayama et al. (2009) to analyze aluminum recycling potential. The authors estimate the possible reduction in primary aluminum consumption in each country/region by forecasting stocks and flows and by accounting for the alloy composition of the aluminum consumed and scrapped. Forecasts of per capita in-use stock for each country/region are made by curve-fitting a logistic function to an assumed relationship between per capita in-use stocks and per capita GDP. This method explicitly assumes that there will be no future product breakthroughs that push per capita in-use stocks above their prior saturation level, an assumption that we show to be in contradiction to historical behavior of aluminum in the U.S.

One of the significant contributions of our work is the linking of MFA and statistical time series analysis. Dynamic MFA models estimate flows and stocks over time, making time series analysis a natural choice for additional study of model results. In particular, we use this approach to quantify the relationship between in-use stocks and economic output as measured by gross domestic product (GDP). This work also analyzes the time series properties of the material flow and economic data. These issues have frequently been ignored in previous studies of the relationships between economic output and material consumption. If appropriate corrections are not made to time series data, common regression techniques can yield results with serious weaknesses related to estimation and inference.

These modeling efforts provide a novel analysis of the behavior of in-use stocks and lay the foundation for future work in forecasting potentially-recoverable aluminum. Overall, we aim to improve the management of aluminum as both a non-renewable resource and as a potential means of reducing GHG emissions by increasing the understanding of the drivers and dynamics of U.S. aluminum in-use stocks.

2.2 Methods

This research first develops a dynamic MFA model of U.S. aluminum and then applies quantitative time series analysis to describe the relationship between in-use stocks and GDP. The MFA model utilizes a top-down approach to estimate the U.S. in-use stock and old scrap recycling and recovery of aluminum beginning in the year 1900 and ending in 2007. Discarded aluminum is collected in the form of new and old scrap. Old scrap is generated once a product reaches the end of its useful lifetime and is retired and disposed. New scrap is generated during the production of semi-fabricated and finished products. New scrap recovery is not explicitly estimated by the model, but the consumption of new scrap is implicitly included in the apparent consumption data. Apparent consumption serves as a metric of total metal demand and is calculated as domestic primary and secondary production plus imports minus exports and adjusted for inventory change. Because it is generally of a known and homogenous quality, nearly all new scrap is recycled and recovered soon after its generation. Data on the apparent consumption of aluminum by major end use category (USGS, 2009) are used to calculate model results for the seven major end-use categories of construction, consumer durables, containers and packaging, electrical, machinery and equipment, transportation, and other. Model equations and detailed discussion of the model calculations are provided in Appendix A.

Aluminum products are added to the existing U.S. in-use stock of aluminum when they are consumed in the economy. As these new products enter their use phase, others are retired and discarded when they reach the end of their useful lives. The cumulative in-use stock accounts for the flows of new and retired products. There are instances when products, such as buildings, reach retirement and are not immediately discarded. These are referred to as "hibernating stocks" (Bergbäck and Lohm, 1997) and their effect has not been included in this model due to a lack of data. Additionally, based on the major areas of consumption it is a reasonable assumption that most of the aluminum products enter the waste stream after they reach the end of their useful lives.

Annual product retirement flows are calculated for each end use category using a probability density function estimated to be representative of each category's average product lifetime. Product lifetime probability density functions based on the normal, beta, and Weibull distributions were selected from Melo (1999) and are identified in Appendix A. These product lifetime distributions were developed by first identifying lifetime intervals for sub-categories of products. The lifetime interval for each average end-use sector was then calculated by taking the consumption weighted average of lifetime interval of the appropriate sub-categories. Although product lifetimes evolve over time, the subjective nature of estimating a lifetime range for even a current product makes this parameter uncertain. In order to address this uncertainty, we first quantify the sensitivity of this model parameter and utilize alternate estimates of product lifetimes as an uncertainty analysis. These alternate estimates are also included in Appendix A.

The nature of the model's top-down approach and use of apparent consumption data means that imports and exports of finished products containing aluminum are not included as input data. These indirect flows may represent significant sources of aluminum for the U.S. economy, as the U.S. is a net importer of many finished goods. Although no analysis has been published on mass of aluminum contained in the net trade of finished products for the U.S., Johnson and Graedel (2008) found that metal in traded products accounted for between 13% and 57% of total metal trade flows for copper, lead, zinc, chromium, and silver.

In order to increase the model's capture of U.S. aluminum consumption, the existing apparent consumption data are augmented with the data that are available on the net trade of aluminum products (i.e. doors and windows, household items, and motor cars and other motor vehicles) for the period of 1989 to 2007 (USITC, 2009). Although these data do not capture all of the aluminum contained in traded finished products, they do represent products that are part of the major end-use categories of aluminum consumption. Including these net trade data increases the model's capture of consumption by an average of 13% over the period, compared to USGS apparent consumption data, and provides a lower bound of estimated aluminum use. Details are included in Appendix A.

Statistical time series analysis is used to investigate and quantify the relationship between in-use stocks and GDP. Unit root testing is performed to determine covariance stationarity for each data series. Non-stationarity refers to the condition where the probability distributions of data are time dependent. When data exhibit this property, the ordinary leas squares (OLS) method results in a spurious regression, where the regression estimators are biased and inefficient and have biased standard errors. Under these conditions, the inference of statistical significance of the estimators is invalid (Granger and Newbold, 1974). Following unit root testing, non-stationary data are subjected to first-differencing or trend removal. The relationship between in-use stocks estimated by the MFA model and GDP is then statistically estimated using linear estimation methods.

MFA model parameter sensitivity is quantified using the Fourier Amplitude Sensitivity Test (FAST) method (Cukier et al., 1978). The FAST method provides a quantitative measure of input sensitivity expressed as the fraction of total model variance. It is capable of accounting for nonlinear and interaction effects of input parameters, unlike a sensitivity analysis technique such as perturbation analysis (Saltelli et al., 1999).

2.3 Results

2.3.1 In-use Stocks

Two distinct periods of logistic growth in aluminum in-use stocks are seen in Figure 2.1. The first corresponds to the period between 1946 and 1986, when aluminum consumption was rapidly increasing in the construction and electrical sectors. Although there was growth in each of the end-use sectors as a result of overall economic expansion, consumption was largely driven by new product development and product substitution (Brubaker, 1967). The second period of logistic in-use stock growth occurs from 1986-2006 and unlike the first expansion, this was driven by consumption and substitution in the transportation sector. In particular, substitution occurred for many cast iron components of automobiles (Sheridan, 1996).

The U.S. in-use aluminum stock in 2007 is estimated as 91.1 million metric tons (Mt) assuming a beta distribution for product lifetimes, 97.6 Mt assuming a normal distribution, and 92.2 Mt assuming a Weibull distribution. On average, approximately 34% of the cumulative apparent consumption of aluminum is contained in in-use stocks. The construction and transportation sectors represent the largest components of in-use stock, a result of their large fraction of apparent consumption and the length of their average product lifetimes. The container and packaging sector is another large consumer of aluminum, but the short lifetimes of its products result in little accumulation. The model estimates that the total in-use stock decreased for the first time in the post-war period in 2007. A loss in the total in-use stock indicates that the mass of aluminum products retired exceeds the mass of aluminum products consumed. The average net stock loss across the three lifetime distributions in 2007 was 546,000 metric tons.

Sullivan (2005) estimates the in-use stocks in 2002 as 142 Mt, a figure that is 56% larger than the average of our model results of 91.1 Mt for the same year. Unfortunately, essential model details are not published there and it is not possible to determine the reasons for the large difference between estimates. Hatayama et al. (2009) estimate U.S. in-use stocks in 2003 at 120 Mt, compared to our model estimate of 92.3 Mt. Although the two models rely on different data sources, the difference in in-use stock estimates is most likely due to the product lifetime and recovery assumptions made by each model. When the same product lifetimes are used, which is discussed in Section 4, our model estimates 2003 in-use stocks as 119 Mt.

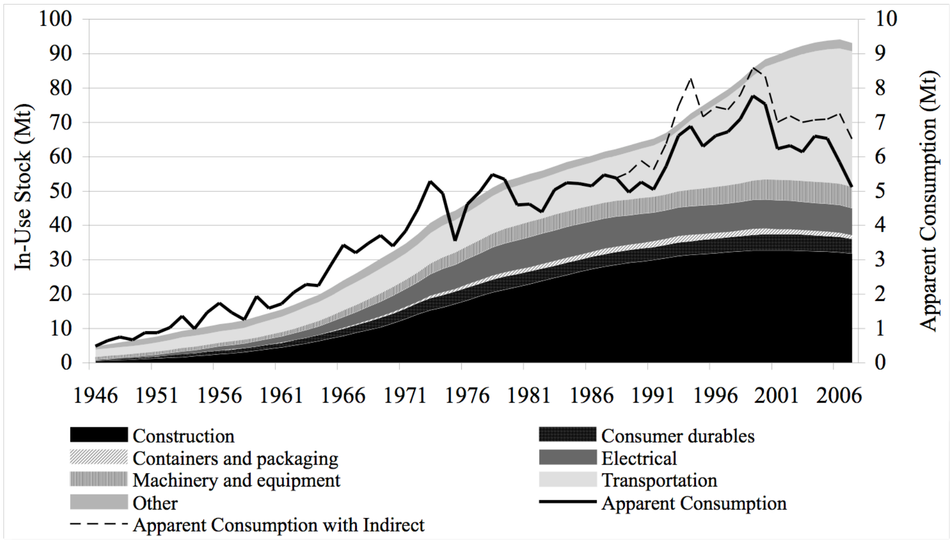

Figure 2.1 U.S. Estimated In-Use Aluminum Stocks (Average Across Distributions), Apparent Consumption (USGS, 2009), and Apparent Consumption Including Indirect Flows.

2.3.2 Aluminum Recycling and Recovery

The model estimates the annual mass of aluminum collected from product retirement (i.e., aluminum recycling) and aluminum metal obtained from scrap remelting (i.e., aluminum recovery). Unless specifically stated, the estimates of aluminum recycling and recovery do not include scrap trade flows. Figure 2.2 illustrates that the transportation and containers and packaging sectors contribute the vast majority of aluminum recovered from old scrap. These data are consistent with the fact that the sectors represent a large fraction of apparent aluminum consumption and have high recycling rates relative to the other end-use sectors. The model estimates that the construction sector contributes a much smaller fraction of the aluminum recovered from old scrap even though the sector represents the largest portion of in-use stock. This can be explained by the assumed low recycling rate and the long product lifetimes of the sector. The accumulated unrecovered aluminum in the U.S. is estimated to be 107 Mt in 2007, which is equal to approximately 39% of the cumulative apparent consumption since 1900. This mass represents the material that was not collected for recycling in the U.S. and was therefore not made available for domestic consumption or for export to foreign markets.

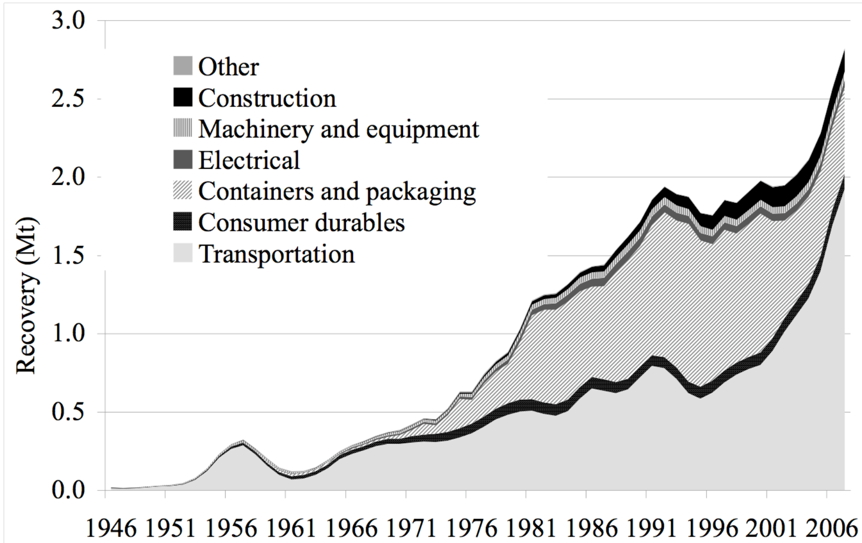

Figure 2.2 Domestic Recovery of Aluminum from Old Scrap by End Use Sector (Average Across Distributions)

Additional information on U.S. aluminum is revealed by estimating the annual percentage of total aluminum collected for recycling by the domestic economy. This metric is calculated by dividing the mass of old scrap recycled domestically by the total mass of aluminum retired for that year based on model estimates of the annual amount of aluminum entering the waste stream[1]. To provide such a measure, it is necessary to first estimate the annual amount of old scrap that is collected domestically. USGS data on old scrap consumption do not represent domestic collection of old scrap because they include net scrap trade. As a result, data on scrap imports are subtracted and data on scrap exports are added, which leaves the mass of scrap recycled by the U.S. for consumption domestically or abroad.

Estimation of an overall recycling rate can be further improved by correcting for the consumption of aluminum beverage cans. Since the widespread adoption of aluminum beverage cans in the mid-1970s, the total recycling and recovery of aluminum has been largely driven by the collection of used beverage cans (UBCs). Yet, UBCs are part of the closed loop system of aluminum beverage cans, whereby UBCs are collected for

[1] Model estimates of old scrap recycling rely on assumptions of constant recycling rate, with the exception of the containers and packaging sector. Results of this calculation would reflect changes in the fraction of aluminum consumed by each sector and not changes in the recycling rate.

remelting into new cans. Due to this closed-loop system, the mass of UBCs collected does not provide the best indication of the amount of scrap available for producers of products other than beverage cans. Removing data on the consumption, disposal, and collection of aluminum beverage cans develops a more appropriate metric of old scrap recycling rate. Additional discussion is provided in Appendix A.

Without including UBCs, the highest recycling rate during 1972-2007 was approximately 37% in 2007. Preceding this peak was a period of gradually decreasing recycling rate, which concluded with a value of 13% in 2004. An earlier peak in recycling occurred in 1990 when approximately 29% of the aluminum from waste streams was recycled.

Even if aluminum is collected for recycling in the U.S., it is not necessarily consumed within the domestic economy. Because of its large endowment of in-use stocks, the U.S. has become a significant exporter of old scrap to the rest of the world. Using data available for UBCs beginning in 1989 (USITC 2009), it is possible to estimate the percentage of U.S. recycled non-UBC scrap that is consumed domestically. Adjusting for UBC scrap lowers the rate of domestic old scrap consumption by as much as 50 percentage points; it is estimated that in 2004 only 2% of non-UBC old scrap that was recycled from U.S. waste streams was consumed domestically. Data used for this analysis, as well as an accompanying figure, are included in Appendix A.

2.3.3 Net Additions to In-Use Stock

Net additions to in-use stock (NAS) are calculated as the difference between annual aluminum consumption and retirement; this is equivalent to the annual net consumption of aluminum. The basic underlying thought is that periods of economic growth will lead to positive NAS. NAS peaked in 1973 at an average of 3.5Mt then fell, likely due to a combination of the economic disruptions of the same year (e.g., stock market crash, first oil crisis) and a saturation of aluminum products in the construction market, as shown in Figure 2.3. Although the NAS of the subsequent years remained positive, the growth trend experienced from 1946-1973 was not matched until the large expansion of the early 1990s when consumption in the transportation sector increased. The decreasing NAS since 2001 appears to have been first precipitated by a recession and then maintained by flattening consumption and growing product retirement in the transportation market.

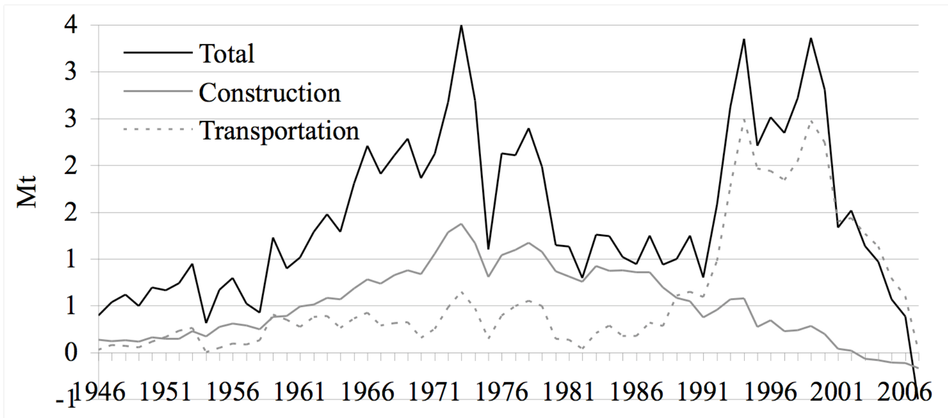

Figure 2.3 Net Additions to In-Use Stock, Average across End-of-Life (EOL) Distributions

2.3.4 Quantitative Analysis of Aluminum Stocks and GDP

One popular framework for analyzing metals use in an economy is the intensity of use hypothesis (International Iron and Steel Institute, 1972; Malenbaum, 1975). A type of environmental Kuznets curve (EKC) (Selden and Song, 1994; Grossman & Krueger, 1995), the intensity of use hypothesis asserts that metal consumption expressed on the basis of a per capita measure of gross economic output follows an inverted U-shape. As an economy first develops and expands its industrial base and infrastructure, it experiences an increasing intensity of use. The increase then slows and finally decreases as the economy matures and transitions from manufacturing to less resource-intensive activities. A comprehensive review of intensity of use and dematerialization studies is provided by Cleveland and Ruth (1999).

One purpose of this paper is to illustrate a number of potential metrics for evaluating the economy's in-use stocks. The focus of our analysis is the relationship between in-use stocks, population, and GDP. Figure 2.4a presents indices of in-use aluminum stock per GDP, per capita, and per GDP/capita for 1946-2007. In-use stocks on a per GDP basis reveal a distinct plateau between 1975 and 1984, followed by a decrease of 18% from 1984 to 2007. Unless aluminum consumption increases on a large scale relative to GDP growth, it appears that the U.S. aluminum in-use stock per GDP peaked at 10.6 metric tons aluminum per million US$ GDP in 1982.

Instead of in-use stocks, a more appropriate approach under the EKC framework is to analyze GDP with NAS, which, like GDP, is a flow variable. Figure 2.4b depicts indices of NAS per GDP, per capita, and per GDP/capita for 1947 – 2007. With the exception of the early 1990s, which experienced a positive surge in NAS in the transportation sector, there is a distinct downward trend since 1973 for all three indices. This trend likely reflects the service sector's increasing share of GDP over the same period. In fact, the service sector share of GDP grew at an annual rate from 1973 to 2007 that was nearly twice as fast as its annual rate from 1947 to 1973 (BEA, 2010a).

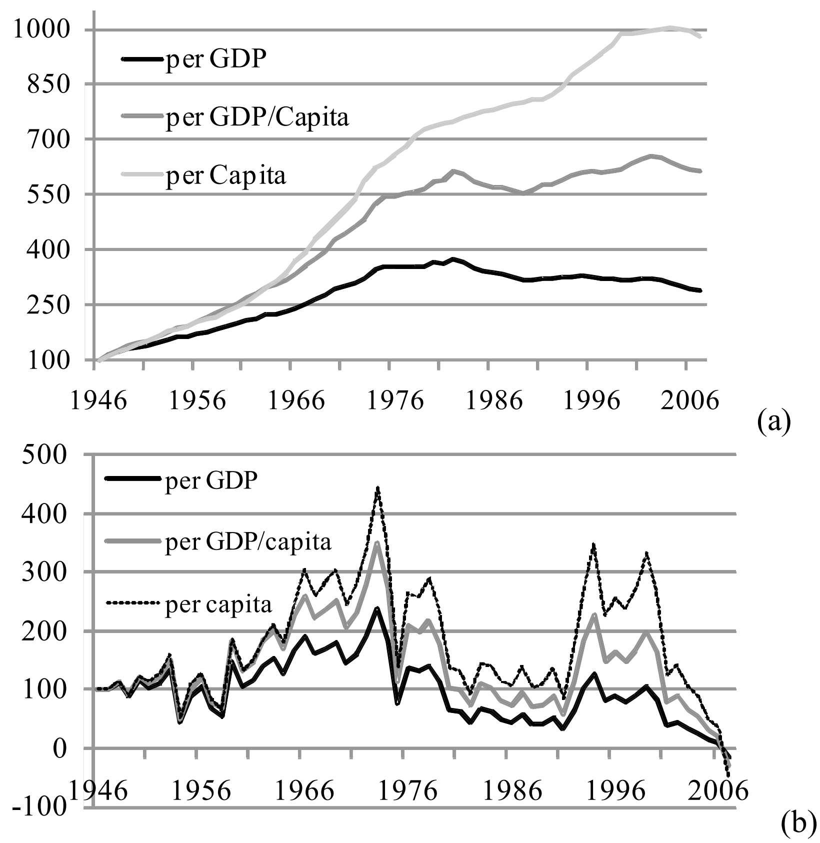

Figure 2.4 Indices of (a) In-Use Stock (rebased, 1946=100) and (b) Net Additions to In-Use Stock (rebased, 1947=100)

We further analyze NAS per GDP and per GDP/capita by disaggregating the underlying NAS data by end-use sector. Data for the construction and transportation sectors, which represent the two largest components of total in-use stock, are presented as Figure 2.5. The disaggregated data reveal that the NAS of nearly all the end-use sectors peaked by 1980, with the transportation sector the only exception. As was identified previously, the sector has experienced a surge in consumption since the early 1990s when large-scale substitution for cast iron components intensified in automobiles. NAS in the transportation sector have grown so quickly relative to GDP that they have largely offset the declines seen in the remaining end-use sectors. Figure 2.5, together with Figure 2.3, hint at an impending saturation in this market, an observation supported by the technical and economic difficulties associated with moving beyond cast components and producing vehicles with aluminum body panels and structural elements (Schatzberg, 2003).

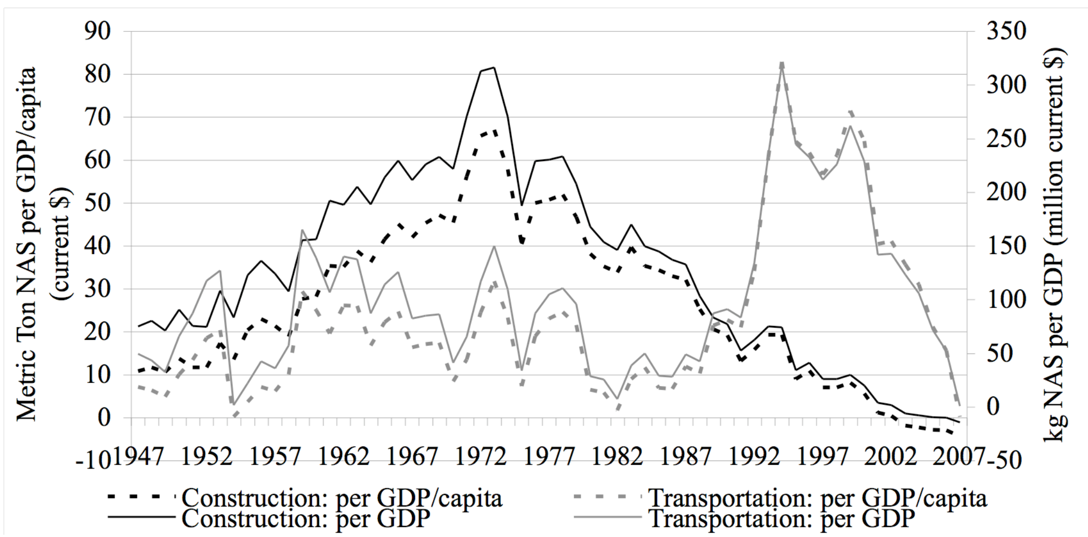

Figure 2.5 Disaggregated Construction and Transportation Net Additions to Stock per GDP

We also analyze NAS on the basis of first difference of natural logs (Δln), which approximates the annual percentage change for small changes in data. Visual inspection of these data, shown in Figure 2.6, indicates that there is some correlation between the Δln GDP and Δln NAS. This potential relationship is explored in detail in Section 2.3.4.1.

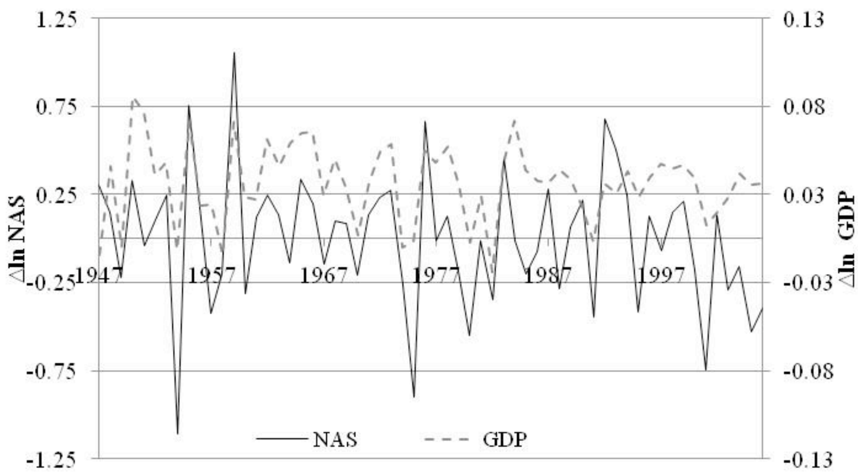

Figure 2.6 First Differenced Natural Log of NAS and GDP

2.3.4.1 Stationarity Testing of Net Additions to In-Use Stock and GDP

The graphical analysis discussed in the beginning of Section 3.4 lends support to the existence of a systematic relationship between NAS and GDP. As a result, a statistical analysis was undertaken to develop a quantitative model for the period from 1948 to 2006[2]. Model parameters are chosen based on quantitative measures rather than on assumptions of their behavior, providing a more statistically rigorous approach than what is utilized by Hatayama et al. (2009). By analyzing time series data of NAS and GDP, we take a different approach than what has been used previously for cross-sectional studies of copper and zinc flows and stocks (Binder et al., 2006; Reck et al., 2006) and the largely cross sectional analysis of European material and waste flows (Andersen et al., 2007).

Non-stationarity of time series data, defined as data having a time-dependent probability distribution, is a common condition and the appropriate testing and adjustments to model specification must be undertaken to obtain valid regression results. Previous econometric models utilizing OLS regressions of metal stocks data have neither acknowledged nor accounted for the possibility of non-stationary data. Consequently,

[2] The time period was chosen to correspond with the years subsequent to the end of the Second World War. Based on the use of first differenced and lag values, 1948 is the starting year.

these may represent instances of spurious regression. The most recent examples include models of Western European secondary aluminum production (Blomberg and Söderholm, 2009; Blomberg and Hellmer, 2000) and of the influence of old scrap flows and stocks on secondary copper production (Gómez et al., 2007). EKC analysis is also subject to the problems of non-stationary data**,** as discussed by Perman and Stern (2003). While some (e.g., Iriarte-Goñi and Ayuda, 2008; Friedl and Getzner, 2003) have addressed the time series properties of data, discussion and testing of data stationarity remain absent in other studies of dematerialization and material intensity of use (e.g.**,** Vehmas et al., 2007; and Canas et al., 2003).

In this research stationarity testing of all data series were performed using the one- and two-break minimum Lagrange multiplier (LM) unit root tests of Amsler and Lee (1995) and Lee and Strazicich (2003), respectively. Initial testing was performed on the data in levels, which concluded that the GDP series contains a unit root. In order to have balanced equations where the data series are integrated of the same order, the data were then transformed by natural log and first-differenced to remove this unit root. Unit root testing results for all data are presented in Table 2.1. The two-break test was used first to identify the number of structural breaks in intercept and slope for each data series. In the instances where the LM unit root tests did not reveal a structural break significant at the 5% level, such as the disaggregate GDP data for durable manufacturing-motor vehicles, the Phillips-Perron (1988) and Kwiatkowski et al. (1992) tests were used. Data series were then detrended based on the identified structural break.

Table 2.1 Unit Root Testing Results for Data as First-Differenced Natural Log and in Levels (where noted)

Data Series	Period	Break Point(s)	Critical Value at 5%	Test Statistic	Unit Root?
Aggregate Data					
NAS	1948-2006	1973	-4.5	-9.58	No
NAS (level)	1948-2006	1972, 1990	-5.7	-6.57	No
NAS per capita	1948-2006	1973	-4.5	-9.57	No
NAS per capita (level)	1948-2006	1966, 1990	-5.7	-5.79	No
NAS alternative	1948-2006	1973	-4.5	-9.84	No
NAS alternative (level)	1948-2006	1977, 1990	-5.7	-5.80	No
NAS per capita alternative	1948-2006	1973	-4.5	-9.81	No
NAS per capita alternative (level)	1948-2006	1966, 1990	-5.7	-6.03	No
GDP	1948-2006	1970	-4.5	-7.32	No
GDP (level)	1948-2006	1980	-4.5	-3.37	**Yes**
GDP per capita	1948-2006	1970	-4.5	-7.38	No
GDP per capita (level)	1948-2006	1970	-4.5	-3.84	**Yes**
Data Disaggregated by Sector					
NAS (construction)	1948-2002	1993	-4.5	-6.75	No
NAS per capita (construction)	1948-2002	1994	-4.5	-6.79	No
NAS (transportation)	1948-2006	2000	-4.5	-9.56	No
NAS per capita (transportation)	1948-2006	2000	-4.5	-9.56	No
GDP (construction)	1948-2006	1957	-4.5	-5.75	No
GDP per capita (construction)	1948-2006	1957	-4.5	-5.70	No
GDP (motor vehicles)	1978-2006	na	PP: -2.97 KPSS: 0.463	PP: -4.99 KPSS: 0.227	No
GDP per capita (motor vehicles	1978-2006	na	PP: -2.97 KPSS: 0.463	PP: -4.97 KPSS: 0.229	No

Note: "Alternative" refers to NAS calculated using alternative product lifetime assumptions.

2.3.4.2 Model Estimation of Net Additions to Stock and Gross Domestic Product

The relationship between NAS and GDP is estimated for aggregate and disaggregate data series. The aggregate data includes estimation of the NAS calculated by the alternative lifetime distribution assumptions and NAS and GDP per capita. OLS is used to estimate the aggregate model for the period of 1948-2006 based on the specification

$$y_t = \alpha + \beta_1 y_{t-1} + \beta_2 x_t + \varepsilon_t \tag{2.1}$$

where α is the intercept term, y_{t-1} is the one-year lag Δln NAS, x_t is Δln GDP, and ε_t is the random disturbance term. Regression results are presented in Table 2.2, with t-statistics shown in parenthesis. The Breusch-Pagan and Breusch-Godfrey tests were used to test for the presence of heteroskedascitity and serially-correlated errors of the first order, respectively. Results of these tests indicate that their null hypotheses of no heteroskedasticity and no serial correlation cannot be rejected below the 16% level.

The parameter coefficients in each model are interpreted as a percentage point change in Δln NAS that is associated with a one percentage point change in a regressor. For example, the regression estimates for Δln NAS indicate that each one percentage point increase in last year's Δln NAS and current Δln GDP is associated, *ceteris paribus*, with a change in current Δln NAS of -0.196 percentage points and 10.6 percentage points, respectively. Overall, the model results indicate that large, statistically significant changes in NAS are associated with changes in economic output as measured by GDP. These results are nearly the same using data measured on a per capita basis.

Table 2.2 Regression Results for First-Differenced Total Net Additions to Stock (Δln NAS)

Regressors	**Δln NAS**	**Δln NAS per Capita**	**Δln NAS (alternative)**	**Δln NAS per Capita (alternative)**
Intercept	-0.361*** (-5.23)	-0.241*** (-4.51)	-0.241*** (-5.74)	-0.154*** (-4.74)
L.Δln NAS	-0.196* (-1.94)	-	-	-
L.Δln NAS per capita	-	-0.184* (-1.81)	-	-
L.Δln NAS (alternative)	-	-	-0.251*** (-2.86)	-
L.Δln NAS per capita (alternative)	-	-	-	-0.239*** (-2.69)
Δln GDP	10.6*** (6.22)	-	8.19*** (7.90)	-
Δln GDP per capita	-	10.4*** (6.04)	-	8.11*** (7.73)
R^2	0.437	0.424	0.562	0.552
Breusch-Pagan	0.910 p-value = 0.635	0.829 p-value = 0.661	1.82 p-value = 0.404	1.65 p-value = 0.439
Breusch-Godfrey (order 1)	1.83 p-value = 0.176	1.95 p-value = 0.163	0.0271 p-value = 0.869	0.0026 p-value = 0.959

Notes: "Alternative" refers to NAS calculated using alternative product lifetime assumptions. "L" refers to the 1-year lag of the variable. T-stats of regression estimates in parenthesis. * denotes significance at the 10% level; ** denotes significance at the 5% level; *** denotes significance at the 1% level.

Model estimation for construction and transportation NAS included the effect of changes in GDP value added in the construction and durable goods— motor vehicles categories (BEA, 2010b), in addition to aggregate GDP. These were chosen because they represent a large fraction of apparent aluminum consumption and have significant economic importance. NAS data from the alternate product lifetime distribution assumptions were not included in the model estimation efforts. Initial regression using OLS revealed non-normally distributed residuals and as a result the models were estimated using maximum likelihood MM-regression estimators (Yohai, 1987).

Results of the model estimation reveal that changes in GDP by industry are associated with much smaller and mostly insignificant changes in $\Delta \ln$ NAS for both the construction and transportation sectors than aggregate GDP. Additionally, changes in aggregate GDP were associated with much larger changes in $\Delta \ln$ NAS for transportation than for construction. A detailed summary of model estimation and results is provided in Appendix A.

2.4 Sensitivity and Uncertainty Analysis

The product lifetime probability distribution, recycling rate, and recovery rate were subjected to the FAST method and results are presented in Table 3. Separate results are shown for the containers and packaging category because product retirement is assumed to be a simple one-year lag, which does not follow any statistical distribution. Results show the largest sensitivity in the lifetime distribution and recycling percentage parameters for all end-use categories but containers and packaging. The recycling rate has the largest sensitivity for the containers and packaging category.

A successful application of the FAST method results in the summation of input sensitivity equal to unity. The results shown in Table 2.3 sum to approximately 0.83, indicating that 17% of the total model variance is not captured by the three selected parameters.

Table 2.3 FAST Results for Recovery Model: Contribution to Model Variance

Parameter	Containers & Packaging	All other End-Use Categories
Lifetime Distribution	0.01	0.4
Recycling Rate	0.8	0.4
Metallic Recovery	0.03	0.02

The uncertainty analysis of the model focuses on the lifetime distributions assumed for each product category. Additional normal and Weibull lifetime distributions were calculated based on product lifetimes provided in Müller et al. (2006) and are provided in Appendix A. Although these product lifetimes were originally applied to ferrous products in the U.S. market, aluminum is similarly used in many markets and it can be assumed that the aluminum products share the same product lifetime characteristics.

The largest difference between the product lifetimes provided by the studies of Müller et al. and Melo occurs in the construction end-use category. Mello assumes an average construction product lifetime of 31.5 years under a normal distribution. Müller et al. utilize a more comprehensive analytical methodology and develop a normally distributed average product lifetime of 75 years. Because the sector consumes a significant fraction of aluminum in the U.S., such a large disparity of when products are retired has major implications for the results of the model. In addition to longer product lifetimes, the alternate distributions have slightly different shapes than developed by Melo, which also affects the estimates of product retirement.

The general effect of increased estimates of product lifetimes is to increase current in-use stocks and shift old scrap availability into the future. Due to the timing of consumption growth, differences in model results for in-use stocks emerge toward the end of the period. On average, the product lifetimes from Müller et al. result in in-use stock estimates that are 14% higher and recovery estimates that are 24% lower than when using Melo. Additionally, the average domestic recycling rate during 1972-2007 inclusive of UBCs is on average seven percentage points higher using lifetime estimates from Müller et al.

2.5 Summary and Conclusions

U.S. aluminum consumption and in-use stocks have grown enormously since the beginning of the 20th century and by 2007 in-use stocks represented 34% of the cumulative aluminum consumption since 1900. Aluminum recovery has also dramatically increased, although the average recycling rate from 1972 – 2007 including UBCs is estimated as 25%. As a result, nearly 40% of cumulative apparent consumption was not removed from the waste stream for recycling. Additional significant losses of aluminum by the domestic economy have recently occurred due to scrap exports. These conditions represent significant opportunities for the U.S. domestic market to increase its recycling and recovery of aluminum from old scrap and indicate the need for more aggressive recycling policies. One option would be to explore the use of extended producer responsibility (EPR), or take-back, programs such as Europe's Waste Electrical and Electronic Equipment (WEEE), End-of-Life Vehicle, and Packaging and Packaging Waste Directives (Tojo and Hansson, 2004).

The exponential increases in aluminum in-use stocks have historically been the result of a combination of new product development and substitution and economic growth; however, most of the aluminum end-use sectors have become saturated as measured by their mass of in-use stock per GDP and per GDP/capita. Our graphical and quantitative analyses of in-use stocks provide an increased understanding of where and why potentially-recoverable aluminum accumulates in the U.S. economy.

We have demonstrated the potential of time series analysis and other econometric techniques in building quantitative, explanatory models of MFA data. This work also highlights the importance of stationarity testing of MFA data, a consideration that has largely been ignored by the MFA community. The quantification of a relationship between the annual percentage changes in NAS and GDP leads to a better understanding of the extent to which economic output drives U.S. aluminum use. Due to the success of this methodology in analyzing a complex system like aluminum, we expect that the approach could be widely applied to other metals and material commodities.

One avenue for future research involves additional time series modeling. Testing for a cointegrating relationship between NAS and GDP data and then developing error correction models based on the cointegration results could provide further enhancements to the analysis. A second avenue is to investigate further the influence of economic activity on aluminum use. Variables could be constructed for the disaggregated components of GDP – final consumption, investment, government purchases, and net exports – to develop a richer analysis of the relationship between economic output and NAS of aluminum. As demonstrated in the paper, such an analysis would be complicated by the need for unit root testing of the individual variables as a precursor to estimating a regression model.

Although it was not in the scope of this research, it would be possible to use results of the model to forecast old scrap availability based on existing GDP forecasts. Estimating the future changes to in-use stocks and old scrap availability would aid both the evaluation of potential GHG mitigation strategies involving aluminum substitution and the management of aluminum as a non-renewable resource. For example, forecasts could provide planning agencies with metrics to help match recycling infrastructure capacity with anticipated flows of discarded aluminum products. More effective aluminum management could be a relatively inexpensive approach to GHG mitigation.

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CHAPTER 3

NOT ALL PRIMARY ALUMINUM IS CREATED EQUAL: LIFE CYCLE GREENHOUSE GAS EMISSIONS FROM 1990 TO 2005

ABSTRACT

Primary aluminum ingot is a globally traded commodity, and large regional differences in technology and electricity fuel mixes exist among the industry's smelters. A life cycle assessment model is developed to calculate absolute emissions and emissions intensities of greenhouse gases (GHGs) from the production, trade, and consumption of primary ingot in six world regions. Global production emissions in 1990 are estimated at 283 (±18) Mt CO2-eq, or 14.7 (±0.95) kg CO2-eq/kg primary ingot on an intensity basis. In 2005 global emissions are estimated at 468 (±26) Mt CO2-eq, or 14.7 (±0.80) kg CO2-eq/kg primary ingot on an intensity basis. In total, the production of primary aluminum accounts for 0.78 and 0.93% of world GHG emissions in 1990 and 2004, respectively. Regional production GHG intensities in 2005 range from 7.07 (±0.69) kg CO2-eq/kg primary ingot in Latin America to 21.9 (±3.0) kg CO2-eq/kg primary ingot in Asia. The GHG implications of expanding global trade of primary ingot are examined in terms of the emissions embodied in the imports and exports and the consumption-weighted emissions intensities of each region.

3.1 Introduction

The latest wave of globalization has aided the separation of the world's centers of production and consumption. As this separation has increased and economies have grown more open, the consumption of many goods and services has become disconnected from the environmental burdens associated with production. Efforts have consequently turned to quantifying resource depletion and environmental impacts embodied in international trade. A review of the most common modeling approach, input-output (I-O) analysis, is provided by Wiedmann et al (2007). I-O modeling is typically performed at the household, country, or regional level and because these models rely on aggregated sector data, the analysis of a single commodity (e.g. primary aluminum ingot) or product is difficult.

Calculating consumption-based greenhouse gas (GHG) emissions is part of the larger movement of analyzing environmental impacts embodied in international trade flows. Consumption-based GHG inventories have been proposed as a means of addressing the allocation of emissions from international activities, such as shipping, and the effectiveness of the Kyoto Protocol (Peters and Hertwich 2008a). To date, analyses of GHG emissions embodied in trade have relied on aggregated trade data in the form of I-O tables (Ahmad and Wyckoff 2003; Munksgaard and Peterson 2001; Peters and Hertwich 2008b; Weber and Matthews 2007). These analyses have not been applied at the level of an individual commodity and lack consideration of year-to-year changes in GHG emissions intensity. Observed changes in the geographic distribution of production and consumption of primary aluminum ingot, together with efforts in the industry to reduce electricity consumption and perfluorocarbon (PFC) emissions, indicate the importance of explicitly accounting for the annual changes and regional differences in emissions.

Primary aluminum ingot provides unique characteristics for quantifying GHG emissions embodied in commodity trade. As ingot demand has increased across the globe, capital in the primary aluminum industry has flowed to regions with large, inexpensive and secure sources of electricity and has created regions that are large net exporters or net importers of ingot. The resulting electricity fuel mix among regions, along with other technological differences described herein, has created a wide range of regional GHG intensities of primary ingot production. In turn, the GHG emissions embodied in a region's consumption of primary ingot have come to depend not only on the amount of ingot consumed, but also the origin of the ingot.

Our objective is to quantify the temporal and geographic variation in life cycle GHG emissions of production, trade, and consumption of primary aluminum ingot. To do so, we have constructed a dynamic life cycle model that utilizes data on smelter technological performance and bilateral ingot trade flows to calculate a time series of emissions estimates. Our model captures the dynamic nature of intensities of smelter electricity consumption, electricity fuel mix, carbon intensity of fossil fuel electricity generation, and PFC emissions.

Time series life cycle emissions data have not been provided by existing primary aluminum ingot life cycle inventory (LCI) reports or life cycle inventory databases, which instead offer point estimates of emissions every few years. The time series approach to emissions is also not used in the previous I-O studies on GHGs embodied in trade. Not considering annual changes associated with smelter technology and electricity generation may completely overlook annual variation in emissions intensity of primary ingot production and consumption.

With this dynamic approach, it is possible to show how emissions associated with primary aluminum ingot are evolving over time and across regions. Calculating emissions embodied in trade and consumption of ingot reveals potential distortions that are caused by focusing only on production emissions. If one region is a net importer of ingot, accounting solely for domestic production emissions will underestimate the GHG contribution from the region's economic activity. As a result, quantifying the emissions embodied in trade has implications for GHG mitigation policies that are unilaterally implemented, which potentially creates carbon leakage (i.e., an increase in GHG emissions in countries without GHG regulation due to the adoption of regulation by other countries).

Results of the model can be used to provide a more detailed, life cycle perspective to existing estimates on carbon leakage from the broader non-ferrous metals sector (Paltsev 2001), to inform future policy development on carbon regulation, and to assess future applications of aluminum, such as vehicle lightweighting. The model and its results may also be of particular interest to life cycle assessment (LCA) practitioners who are concerned with the progression of life cycle emissions for the primary aluminum industry and may encourage the greater LCA community to look beyond static modeling assumptions.

3.2 Background

Primary aluminum production begins with mining bauxite ore. The ore is then crushed to size and refined into aluminum oxide (alumina) through the Bayer process. Alumina is formed through precipitation, which involves first dissolving crushed bauxite in heated sodium hydroxide and then clarifying the solution. Alumina is then reduced by electricity in the Hall-Héroult process to produce primary aluminum metal. Reduction in the Hall-Héroult process occurs in steel "pots", which are fitted with a cathode and an anode and lined with carbon and refractory for electric and thermal insulation. Alumina is fed into the pots and dissolved in a bath of molten sodium aluminum fluoride known as cryolite. The cryolite is necessary as an electrical conductor and also acts to lower the melting point of alumina (Choate and Green 2003). As electric current is passed through the alumina, molten aluminum forms at the bottom of the pot. The aluminum is periodically drained and then cast into ingots. Most primary aluminum is alloyed for use in rolled and extruded products for the building and construction, beverage can, and transportation sectors (Aluminum Association 2006).

There are two basic anode technologies used in the Hall-Héroult process. The first technology used was the Söderberg anode. Here the anodes are fixed in the pots and carbon paste is continuously added to the anode as it is consumed by the reduction reaction with alumina. Söderberg anodes are being replaced with the newer anode technology, prebake anodes (Bergsdal et al. 2004). Prebake anodes are produced by first combining carbon paste with pitch and other binders and then baking in natural gas-fired ovens for a number of weeks. Unlike Söderberg anodes, prebaked anodes are not permanently fixed in the pot and are replaced once they are consumed during electrolysis.

Both types of anodes produce CO_2 when they are consumed during the process of electrolysis. Process GHG emissions also occur as a result of the "anode effect", which is the release of high global warming potential (GWP) PFCs from the molten cryolite bath. If the concentration of alumina in the cryolite bath falls below a critical level during electrolysis, an oxidation-reduction reaction between the carbon anode and the cryolite occurs (Choate and Green 2003). This reaction produces carbon tetrafluoride (CF_4) and carbon hexafluoride (C_2F_6), two extremely potent greenhouse gases with 100-year time horizon global warming potentials (GWPs) of 7,390 and 12,200, respectively (Forster et al. 2007). Overall, PFCs, hydrofluorocarbons, and sulfur hexafluoride contributed approximately 1.1% to global anthropogenic GHG emissions in 2004 (Bernstein et al. 2007).

3.3 Methodology

Annual life cycle GHG emissions and emissions intensities of primary aluminum ingot production are calculated for a global average and for six regions during the period of 1990 to 2005. This analysis considers only the life cycle stages of bauxite mining through production and consumption of cast primary ingot. Model equations are presented in Appendix B. Following the International Aluminum Institute's (IAI) classification, these six regions are Africa, North America, Latin America, Europe, Asia, and Oceania (Australia and New Zealand). The GHGs and emissions sources under consideration are CO_2, methane (CH_4), and nitrous oxide (N_2O) from fossil fuel combustion and electricity consumption in bauxite mining, alumina refining, electrolysis, and ingot casting; and process emissions of CO_2 and PFCs from electrolysis. Emissions from fossil fuels include upstream emissions associated with extracting, transporting, and otherwise producing the fuels (NREL 2007). The outputs of each stage of primary ingot production (i.e., bauxite, alumina, aluminum ingot) can be considered high volume commodities and the equipment used to produce these outputs have relatively long average useful lifetimes. Emissions from the production of this equipment are not included in the scope of this research because they would be allocated over many metric tons of primary ingot, representing only a small addition to the GHG total of primary ingot production (Vigon 1993; Frischknecht et al. 2007). The model utilized GWP values from the Intergovernmental Panel on Climate Change's *Third Assessment Report* (2001).

Primary aluminum smelters in the six world regions have technology differences among them that significantly influence the GHG intensity of production. The differences accounted for in the GHG emissions model include primary aluminum production, electricity consumption per metric ton of aluminum, electricity transmission and distribution losses, fuel mix of electricity consumed, CO_2 emissions per kilowatt hour (kWh) of natural gas, coal, and oil-fired electricity generation, and PFC emissions intensity of primary aluminum production.

Primary aluminum production data were obtained from the United States Geological Survey (2006). Because these data are reported on the country level, it was necessary to aggregate them based on the IAI's regional definitions. Bilateral trade data of primary aluminum were obtained from the United Nations ComTrade database (2007). In total, import and export data under the Standard International Trade Classification Second Revision were obtained from this database and data were aggregated according to the six regions. Trade data were adjusted to exclude data on re-export (i.e., imported goods that are subsequently exported) of ingot and were reconciled following Gehlhar (1996). Inventory changes of each region were obtained from the IAI (2007a).

This research is concerned only with the commodity of primary aluminum ingot and not primary aluminum in semi-finished or finished product form. As a result, apparent consumption was used to capture the material flows of primary aluminum at the commodity level. Apparent consumption in each region was calculated region using the relationship

$$\mathrm{C}_i = \mathrm{P}_i + \mathrm{M}_i - \mathrm{X}_i + \mathrm{I}_i \tag{3.1}$$

where C_i is the apparent consumption, P_i is the domestic production, M_i is the import, X_i is the export, and I_i is the inventory change of primary aluminum ingot for region *i*. Using the definition of apparent consumption, the consumption based GHG intensity of a region was calculated by weighting imports by the GHG intensity of the region of their production. Net domestic production (i.e., production minus exports) and additions and subtractions of inventory were weighted by the GHG intensity of production of the home region.

Due to differences in the reporting practices between countries, data on the production and trade of ingot may or may not include the masses of added alloying elements and aluminum scrap. The differences in reporting may affect estimates of apparent aluminum consumption and regional consumption-weighted emissions intensities. This research acknowledges this potential mass balance inequality, but it was not corrected due to a lack of data on the composition of ingot imports and exports. A sensitivity analysis was performed and its results show that GHG emission estimates change by an average of -0.2% and by no more than 2% with a corrected mass balance. An explanation of this correction is provided in Appendix B. GHG emissions associated with the transport of primary ingot between regions were included. Transport was assumed to be provided by residual oil fueled ocean freighters (NREL 2007) and backhaul and transport from the smelter to the shipping port were not accounted for. The Appendix B includes details on the data used for the remaining model inputs: smelter electricity intensity (i.e., the amount of electricity consumed by smelters per unit mass aluminum produced) and electricity fuel mix, CO_2 intensities of coal, oil, and natural gas electricity generation, PFC emission factors, and GHG emission factors for bauxite mining and alumina refining.

The production and consumption of secondary aluminum (i.e., aluminum produced from scrap) was not included in the scope of the model. Although secondary aluminum has resource and environmental advantages over primary aluminum (e.g., production consumes approximately 5% of the energy required for primary production), primary aluminum continues to dominate global production and consumption, satisfying 67% of total aluminum demand in 2006 (IAI 2008a). The growth of secondary aluminum production and consumption is constrained by technical and market factors, including its substitutability for primary aluminum and the amount of economically-recoverable aluminum scrap that is available, both of which are highly dependent on recovery infrastructure.

An uncertainty analysis was performed for the production-weighted and consumption-weighted emissions and emissions intensities. The model parameters of smelter electricity intensity, carbon intensities of fossil fuel electricity generation, and PFC emissions were the focus of the analysis. Standard errors of these parameters were calculated from published sources (IAI 2007b) and were then used to analyze uncertainty propagation through the life cycle model. Confidence intervals of 95% are reported in parentheses in the results. Additionally, a sensitivity analysis was performed to explore the effects of using updated GWPs for PFCs. Confidence intervals and descriptions of the uncertainty and sensitivity analyses are provided in Appendix B.

3.4 Primary Aluminum Production and Consumption Trends

3.4.1 Aluminum Production, Trade, and Apparent Consumption

Global production of primary aluminum increased annually from 1990 through 2005 with the exception of three years. The largest growth occurred after 2001, where production increased by an average of 7% per year. Figure 3.1 summarizes the regional and global changes in production. North America is the only region to have produced less aluminum in 2005 than 1990, as falling production in the United States was not completely offset by gaining production in Canada. In contrast, Asia saw nearly a five-fold increase in production. Production has been rising at an especially brisk pace in recent years for the region, with an average annual increase of nearly 19% from 2002 to 2005. Complete data on ingot import, export, production, inventory change, and apparent consumption are provided in Appendix B. This appendix also contains discussion of the trends in smelter capacity, relative fraction of purchased and self-generated electricity, and primary ingot import reliance.

Global trade of primary ingot increased during the period of 1990 through 2005 in terms of both absolute amount and as expressed as a fraction of global apparent consumption. Beginning in 1990, imports accounted for approximately 19% (3.6 million metric tons) of global apparent consumption. In 2005, global imports totaled 20% of apparent consumption (6.4 million metric tons), down from the period high of 26% (6.4 million metric tons) in 2000. In 1990 the global trade of primary ingot was characterized by two main importing regions and three main exporting regions. Asia and Europe were responsible for over 97% of global imports and Oceania, Latin America, and North America were responsible for 90% of global exports. By 2005, Asia remained the largest importer, while North America surpassed Europe to become the second largest importer. At the same time, exports from North America receded dramatically, replaced by export growth from Europe, Oceania, and Africa.

The global apparent consumption of primary aluminum ingot increased dramatically from 1990 to 2005 as shown in Figure 3.1. Driving this increase was consumption in

Asia, particularly in China. In 1990, 21% of the world's consumption of primary ingot occurred in Asia and by 2005 this figure increased to 49%. This represents an average annual increase of approximately 7.8% from 1990 to 2001 and 15.7% from 2002 to 2005.

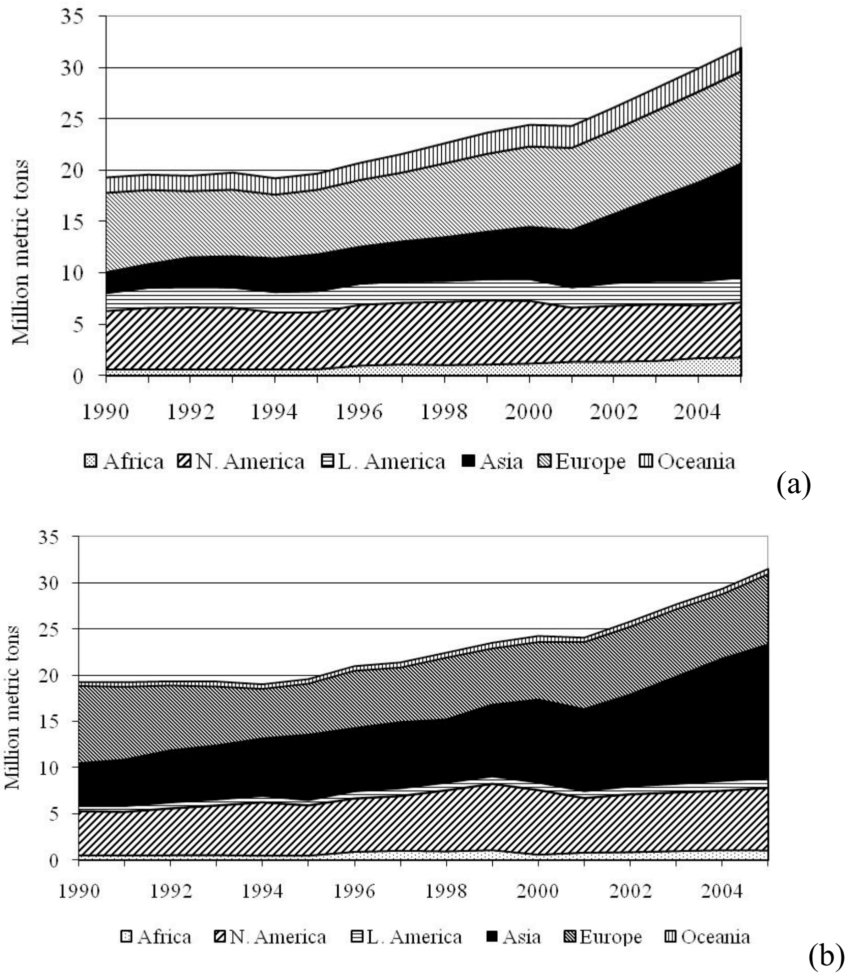

Figure 3.1 Primary Ingot Production (USGS 2007) (a) and Apparent Consumption (b) by Region

3.4.2 Electricity Fuel Mix

Primary aluminum smelters in the six world regions generally receive a majority of their electricity from either coal-fired generation or from hydroelectricity. The coal intensive regions include Africa, Asia, and Oceania, while the hydro intensive regions are North America, Latin America, and Europe. In particular, Africa and Oceania rely on coal-fired generation for greater than 60% and 70%, respectively, of their electricity consumption.

The regional fuel mixes reported by smelters remained relatively constant between 1990 and 2005. The largest changes in fuel mix occurred in Africa, as shown in Figure 3.2. In Africa coal-fired generation has replaced hydro as the dominant source of electricity for smelters. The decrease in hydro was especially precipitous in 1998, likely due to extensive drought experienced in Ghana and, to a lesser extent, southern Africa (Peel 1998; BBC 1998).

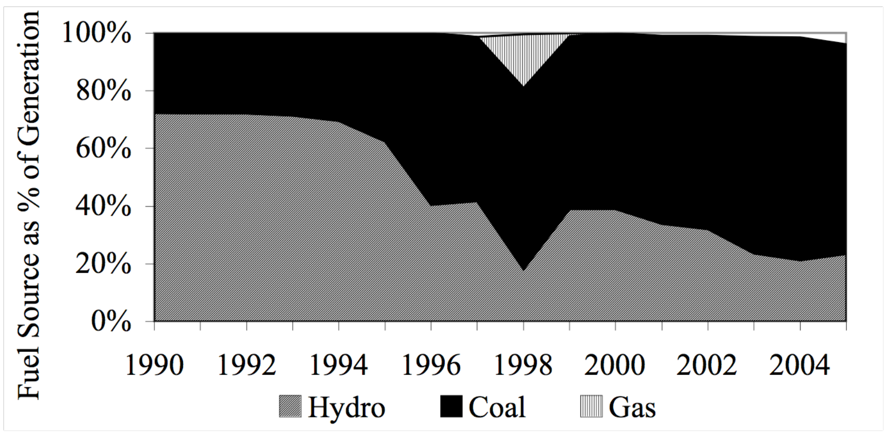

Figure 3.2 Smelter Fuel Mix of Africa (IAI 2007)

Figures depicting the fuel mixes of smelters in the remaining five regions are provided in Appendix B. Additional figures have been included to demonstrate the striking differences that exist between the electricity fuel mix of smelters and the overall grid for each region.

3.4.3 Smelter Electricity Intensity

Between 1990 and 2005, the global average intensity of electricity consumed by primary aluminum smelters decreased by an estimated 6%, from 16,521 kWh/kg to 15,594 kWh/kg. As a reference point to the improvements of the industry, the theoretical minimum energy required for producing aluminum via the Hall-Heroult process is 5,990 kWh/metric ton (Choate and Green 2003). Approximately 85% of this value is energy for driving the reduction reaction and the remainder is thermal energy associated with molten aluminum and maintaining reaction equilibrium. An intensity target of 11,000 kWh/metric ton has been established for the year 2020, which is expected to be reached through continued improvements in cell design, process controls, and other incremental improvements in technology (Choate and Green 2003).

Due to inefficient smelters and large production volumes of China and the former Soviet Union, Asia and Europe were consistently estimated to be the most electricity intensive regions throughout 1990 - 2005. In 1990 smelters in Oceania exhibited the lowest electricity consumption of primary production. Concurrently, these smelters experienced the smallest change in electricity intensity of all regions. Between 1990 and 2005, Africa experienced the largest reduction in electricity intensity of electrolysis.

3.4.4 PFC Emission Intensity

Clear differences in the trends of PFC emissions exist for regions that have chosen to adopt emissions reduction programs, such as North America and Oceania. The most PFC emissions intensive region in 1990 was North America, which emitted 4.36 kg CO_2-eq/kg of primary aluminum. By 2005, emissions of PFCs had decreased by 75% to 1.12 kg CO_2-eq/kg primary aluminum. Asia emitted 3.73 kg CO_2-eq/kg primary aluminum in 1990 and after an initial decrease, the emissions intensity steadily increased due to China's emergence as the region's dominant producer and the country's high emissions intensity. Asia's PFC emissions intensity in 2005 was the highest among all regions at 3.25 kg CO_2-eq/kg primary aluminum, a decrease of only 13% from 1990.

3.5 Results and Discussion

3.5.1 Total GHG emissions

It is estimated that the global production of primary aluminum emitted 283 (±18) Mt CO_2-eq in 1990 and 468 (±26) Mt CO_2-eq in 2005. As a percentage of global GHG emissions, the primary aluminum industry is estimated to have accounted for 0.78% in 1990 and 0.93% in 2004. Total emissions are shown as Figure 3.3. Complete time series data are provided in Appendix B.

Europe, North America, and Asia accounted for 82% and 80% of the total global GHG emissions from the production of primary aluminum in 1990 and 2005, respectively. Asia's estimated emissions total in 2005, 243 (±33) Mt CO_2-eq, represented an increase of 452% over the region's 1990 emissions of 44.1 (±13) Mt CO_2-eq.

Conversely, Europe and North America experienced a decline in total emissions from 1990, with emissions from Europe decreasing by 24% and emissions from North America decreasing by 36%. Unlike Europe, however, North America produced less primary aluminum in 2005 than in 1990.

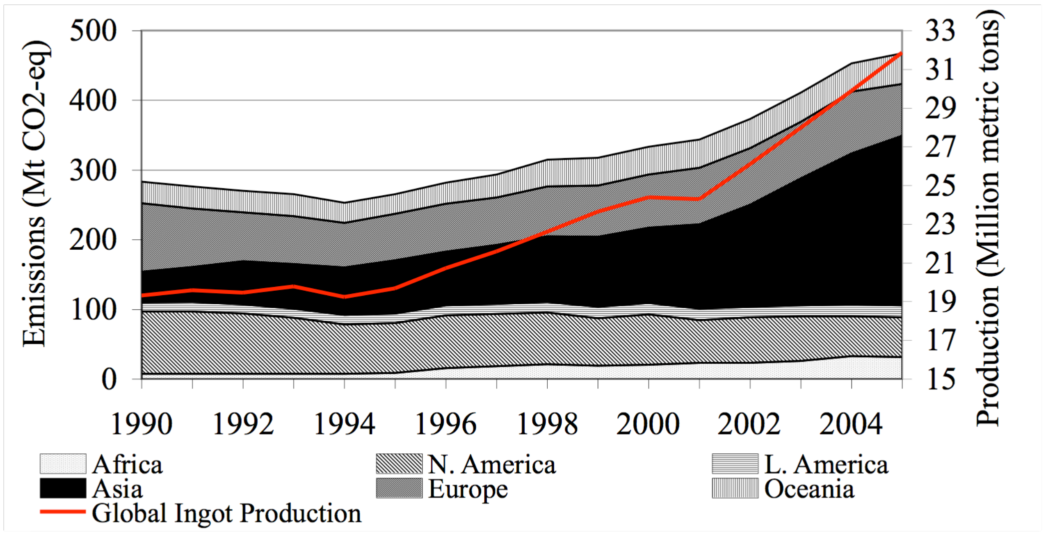

Figure 3.3 Total Life Cycle GHG Emissions by Region and Global Ingot Production (USGS 2006)

3.5.2 Production-Weighted Emission Intensity

The global production-weighted average GHG emission intensity of primary ingot was 14.7 (±0.94) kg CO_2-eq in 1990. Approximately 56% of this value was due to emissions from the generation of electricity consumed during smelting and 24% was due to PFC emissions. Detailed breakdowns of the process contributions to GHG intensities and the time series intensities are provided in Appendix B. From 1990 and 1994, the global average GHG emissions intensity declined at an average annual rate of 3%. After 1994, however, the downward trend in emissions intensity reversed and rose by an average of 1% per year to reach 14.7 (±0.78) kg CO_2-eq/kg primary aluminum in 2005. The portion of GHG intensity attributable to smelter electricity consumption increased further to 65%, while PFC emissions decreased to 15% of the total intensity.

The IAI published a global average primary ingot emission intensity of 9.812 kg CO_2-eq/kg primary ingot for 2005 (IAI 2007b). This value is 33% lower than the estimate developed using our model. Much of the difference between the two estimates is due to the fact that the IAI does not include data from Chinese smelters, which rely heavily on coal generated electricity, have high PFC emissions intensities, and have low smelter electricity efficiencies.

Production-weighted emission intensities are summarized in Figure 3.4. Ocean transportation of ingot between regions accounted for approximately 0.807 Mt CO_2-eq (0.28%) and 1.29 Mt CO_2-eq (0.28%) of global emissions in 1990 and 2005, respectively. Emissions from the export of ingot by Latin America exhibited the largest fraction of total production emissions. In 1990 and 2005 transport emissions accounted for 2.7% (0.348 Mt CO_2-eq) and 1.9% (0.225 Mt CO_2-eq) of Latin America's production emissions.

Asia consistently exhibited the largest GHG emission intensity of primary aluminum production from 1990 through 2005, a result of the region's intensive use of coal fired electricity generation, low electrical efficiency of electrolysis, and high PFC emissions intensity. Values ranged from a minimum of 20.8 (±3.0) kg CO_2-eq/kg in 1994 to a maximum of 22.5 (±2.6) kg CO_2-eq/kg in 2004. By being the most GHG-intense region, Asia also presents the greatest potential for reducing its GHG intensity. Improvements in process technologies and controls, such as upgrading to the latest pot design, would decrease the intensity of both smelter electricity consumption and PFC emissions.

Africa distinguished itself among regions by being the sole region to exhibit a higher intensity in 2005 than in 1990. GHG intensity increased 46% over the 1990 value of 12.3 (±1.9) kg CO_2-eq/kg to 18.0 (±1.6) kg CO_2-eq/kg. Although the region experienced the greatest reduction in electricity consumed during electrolysis, the fuel mix of African smelters changed dramatically over the same period with a shift from hydroelectricity to coal-fired generation. Without the large shift in fuel mix the GHG intensity in 2005 would have been 10.8 kg CO_2-eq/kg. This reflects a decrease of 6% in emissions from electricity consumption and a 39% decrease in PFC emissions.

In contrast to Africa, both North America and Europe experience an overall decrease in GHG intensity of production from 1990 to 2005. North America saw the largest absolute decrease in emissions intensity, from 15.8 (±1.9) kg CO_2-eq/kg primary aluminum in 1990 to 10.7 (±0.67) kg CO_2-eq/kg primary aluminum in 2005. Much of this reduction was a result of a decrease in PFC emissions intensity of 74%, or approximately

3.24 kg CO_2-eq/kg primary aluminum. Europe exhibited a similar trend in total emissions intensity as North America, decreasing by 34% from 12.6 (±2.2) kg CO_2-eq/kg primary aluminum to 8.31 (±0.74) kg CO_2/kg in 1990 and 2005, respectively.

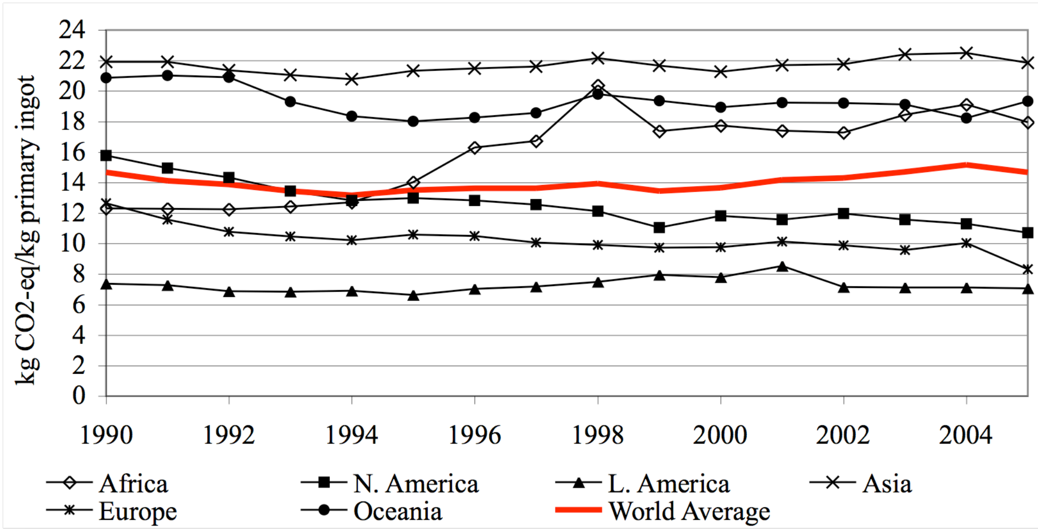

Figure 3.4 Production Based GHG Intensity of Primary Aluminum Production by Region

3.5.3 Consumption-Based Emissions and Emissions Embodied in Trade

The results of accounting for GHG emissions embodied in primary ingot trade are shown in Table 3.1. Compared against production emissions, the measures of consumption emissions and emissions embodied in imports and exports demonstrate the influence of ingot trade and the presence of potential carbon leakage. Regions whose consumption emissions largely differ from their production emissions are either net importers or net exporters of ingot. These regions are, in effect, also net importers or net exporters of GHG emissions. If only production emissions are quantified, the result is either an underestimation or an overestimation of the GHG contributions of economic activities associated with primary ingot. The production emissions for Africa, Latin America, and Oceania exceed their consumption emissions. These regions are net exporters of ingot and emissions and their production driven by outside demand. Conversely, the consumption emissions of Asia and, more recently, North America indicate that the regions are net importers of both ingot and emissions.

The measures of emissions embodied in imports and exports provide additional detail regarding potential carbon leakage. Import and export embodied emissions for each region are reported as a percentage of production emissions in Table 3.1. For example, of the emissions from production of ingot in Oceania in 2005, 75% were associated with ingot that was exported for consumption in other regions. The remaining 25% of the production emissions that year were associated with ingot produced and consumed in Oceania. The emissions embodied in imports portray the converse. In Asia in 1990, the emissions embodied in imports were equal to approximately 97% of the emissions from the region's domestic production. Although the region has dramatically increased its domestic production, by 2005 emissions embodied in Asian imports were 22% of production emissions. In sum, production emissions underestimate the GHG contribution from ingot consumption by 21% (22% - 1%), or nearly 54 Mt CO_2-eq for Asia.

In addition to their GHG emissions totals, emissions intensities were calculated for ingot imports and consumption. The most dramatic disparity between the emissions intensities of production and consumption was exhibited by Asia. This is due to the region's reliance on imported aluminum and the high GHG intensity of domestic production relative to the GHG intensities of imported ingot. Recently, Asia has met an increasing fraction of its apparent consumption with domestic production. This has put upwards pressure on the region’s consumption-based GHG intensity of primary aluminum, but the effect has been largely offset by the falling GHG intensity of imports.

The consumption weighted GHG emission intensity revealed only modest differences when calculated for the remaining five regions. In most cases, the combination of low imports and a close agreement between the GHG intensity of imports and domestic production resulted in consumption-weighted intensities that differed little from the production weighted intensities. An analysis of conditions like these may not add any additional insight to the GHG emissions associated with primary aluminum consumed in a region, but the approach gains importance as trends change. For instance, in 1990 North America produced nearly 98% of the primary aluminum it consumed, but by 2005 this figure had decreased to approximately 75%. Couple this trend with more GHG-intensive imports and only modest reductions in GHG intensity of domestic production and the GHG intensity of aluminum consumed in North America will begin to increase.

Table 3.1 Primary Aluminum Life Cycle GHG Emissions and Emissions Intensities by Region (kg CO_2-eq/kg primary ingot)[a]

		Emissions (Mt CO_2-eq)[b]		Embodied Emissions		Emission Factor (kg CO_2-eq/kg)		
		Production	Consumption	Exports	Imports	Production	Imports	Consumption
Africa	1990	7.45(±1.1)	6.07(±0.64)	24%	2.3%	12.3(±1.9)	12.1(±0.81)	12.3(±1.3)
	2005	31.5(±2.9)	18.2(±1.2)	42%	1%	18.0(±1.7)	11.9(±0.46)	17.8(±1.1)
% change		323%	201%	72%	-75%	46%	-2%	45%
N. America	1990	89.8(±11)	73.9(±6.1)	19%	1%	15.8(±1.9)	8.26(±1.1)	15.5(±1.3)
	2005	57.6(±3.7)	71.0(±2.3)	7%	29%	10.7(±0.67)	9.87(±0.30)	10.5(±0.33)
% change		-36%	-4%	-66%	2040%	-32%	19%	-32%
L. America	1990	12.7(±3.1)	4.18(±0.73)	69%	0.2%	7.36 (±1.8)	14.8(±0.96)	7.18(±1.3)
	2005	16.8(±1.7)	7.47(±0.46)	56%	1%	7.07(±0.69)	9.22(±0.37)	6.96(±0.43)
% change		34%	79%	-19%	196%	-3%	-37%	-3%
Asia	1990	44.1(±11)	84.0(±8.2)	0.4%	97%	21.9(±5.7)	15.8(±0.87)	18.3(±1.8)
	2005	244(±29)	285(±20)	1%	22%	21.9(±2.6)	13.9(±0.58)	19.7(±1.4)
% change		452%	239%	196%	-77%	-0.2%	-12%	8%
Europe	1990	98.3(±17)	105(±10)	2%	8%	12.6(±2.2)	10.9(±0.79)	12.5(±1.2)
	2005	74.6(±6.5)	66.6(±3.0)	25%	13%	8.31(±0.73)	12.2(±0.42)	8.66(±0.39)
% change		-24%	-36%	1307%	55%	-34%	12%	-31%
Oceania	1990	31.1(±3.8)	8.9(±0.74)	72%	0.1%	21.0(±2.5)	13.5(±1.0)	20.7(±1.7)
	2005	43.6(±4.3)	10.9(±0.77)	75%	0.2%	19.3(±1.9)	20.1(±1.6)	19.2(±1.4)
% change		40%	23%	4%	95%	-7%	50%	-7%

[a]The percentage of embodied emissions in exports is calculated as the emissions associated with ingot exports divided by production emissions. The percentage of embodied emissions in imports is calculated as the emissions associated with ingot imports divided by production emissions.
[b] Note that by including inventory changes in calculation of apparent consumption, the annual total production and consumption emissions do not sum to equal amounts.

The results from the model demonstrate the large temporal and regional variation in the life cycle GHG emissions of primary ingot production and consumption. To compare the regional time series consumption and production-based GHG emissions of primary aluminum ingot provided by the model results and values from four common LCA databases, please refer to Appendix B. These databases provide a production-weighted GHG intensity for a single year in a single geographic region. In general, these databases report European data for the mid-1990s and an intensity of around 12.5 kg CO_2-eq/kg.

Large differences in the outcomes of LCA studies may result from using the production- and consumption-weighted GHG emission intensities from our model versus using primary aluminum data found in existing LCA databases and reports. The largest differences will occur in GHG payback analyses of aluminum lightweighting and in aluminum products with minimal use phase (e.g., containers and packaging and certain consumer durables).

In addition to informing policies on carbon regulation, recognizing these disparities may play an increasingly important role in the sourcing decisions of aluminum ingot consumers. Although supply contracts and purchases on the spot market are unlikely to include information on the ingot's production origin, it would be possible to use our model results and obtain a rough estimate of the life cycle GHG emissions intensities of the major aluminum producers. This would be done by identifying their regions of operation (data are readily available from corporations' annual reports and websites) and applying the applicable GHG emissions factor for each region. The companies that have their operations concentrated in GHG-intensive regions (i.e., Africa, Asia, and Oceania) would be expected to have a higher GHG intensity of production than companies concentrated in Latin America, North America, and Europe. This rough method does not account for the many differences in factors affecting GHG emissions that likely exist across the smelters of different companies, but it does serve as a general indicator of the carbon intensity of production and exposure to potential carbon taxes. The model could be used on a finer resolution of emissions calculations on a country or company scale, but it would require data on the aluminum industry that is not readily available to the public.

The model results provide a more accurate specification of emission factors, which can strongly influence LCA results involving primary ingot. For example, promoting GHG reductions from personal transportation through vehicle lightweighting with aluminum will have very different results if ingot is sourced from Asia as opposed to Latin America. LCA practitioners are encouraged to use the model results to explore how temporal and regional variation and consumption-based emissions intensities affect analyses of GHG emissions associated with aluminum products.

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CHAPTER 4

AN EVALUATION OF THE METAL INDUSTRY'S POSITION ON RECYCLING

ABSTRACT

A healthy debate on the treatment of metals recycling in the life cycle assessment (LCA) community has persisted for over a decade. While no clear consensus has emerged, the metals industry has endorsed a set of recycling "facts" that support a single approach, end-of-life recycling, for evaluating the environmental benefits of metals recycling. In this article we draw from research conducted in several disciplines and find that three key tenants of the metals industry capture the theoretical potential of metals recycling from a metallurgical standpoint, rather than reflect observed economic behavior. We then discuss the implications of these conclusions on environmental emissions from metal production and recycling. Evidence is provided that, contrary to the position of the metals industry, metals are: not necessarily recycled at high rates, recycled only a small number of times before final disposal, and are limited in recycling potential by quality constraints. The analysis concludes that metal recycled from old scrap largely serves as an imperfect substitute for primary metal. As a result, large-scale displacement of primary production and its associated environmental emissions is limited to a few specific instances.

4.1 Introduction

Recycling has evolved from what was first a solely economic response to material scarcity to become synonymous with environmental considerations. Although metals have long been a significant component of recycling, a debate surrounding the appropriate method to account for the environmental benefits in life cycle assessment (LCA) has emerged only relatively recently. The metals industry has formalized its position on the nature and evaluation of recycling in Atherton (2007), entitled "Declaration by the Metals Industry on Recycling Principles". The publication identifies how the characteristics of metals and their recycling practices lead to a single appropriate method for calculating the emissions credits that should be associated with recycling in LCA studies. According to this method, called the end-of-life recycling approach, the environmental performance of metals is to be analyzed based on how much metal is recovered at the end of a product's useful life; the fraction that is not recycled must be replaced by primary production. This is in contrast to materials such as paper, plastics, and glass, which may be characterized by their recycled content as materials "that would otherwise be incinerated or landfilled as waste" (Atherton 2007, 59). In the end-of-life recycling allocation method, any metal that is not recycled at the end of life is made up for by consuming primary metal (i.e., metal produced from its mineral form). The ultimate position is that "metal recycling offsets primary production processes – and their associated environmental impacts and energy consumption" [original emphasis] (Atherton 2007, 59).

In our view the end-of-life recycling allocation method neglects to acknowledge the body of work that directly contradicts its assumptions. Some of this work explicitly describes the frequency and extent of recycling, while other work provides information on the characteristics of metals that determine why and how they are recycled. For instance, Peck (2003) provides a critical discussion of conventional recycling assumptions in an analysis of material cycle closures. The author discusses aluminum recycling in Western Europe and finds that many system complexities lead to significant "myths, simplifications, and complications" (3-48) in the commonly held views of recycling.

This article considers the world's three most widely consumed metals/alloys: iron and steel, aluminum, and copper. By synthesizing existing information on the characteristics of these metals and their recycling systems, the validity of end-of-life recycling approach advocated by the metals industry is called in to question. The analysis is specifically organized around three major assumptions of the metals industry as identified by Atherton. These three assumptions provide a foundation for the end-of-life recycling allocation approach, where the function of primary production is to replace metal that is unrecovered from end-of-life products or is lost during remelting. We address the following points as they apply to iron and steel, aluminum, and copper:

1. Metals from end-of-life products are widely recycled at high rates (Section 4.2);
2. Metals can be, and are, recycled over and over again (Section 4.3); and
3. The constraint to metals recycling is the availability of feedstock material (Section 4.4) (Atherton 2007, 59).

These three assumptions support the concept of a pool of material existing for each metal, which is manifest in the end-of-life recycling allocation approach. As long as material is present in the pool, it will be recycled; high end-of-life recycling rates assure that material is not lost from the pool; and the ability of metals to be recycled over and over again keep the material pool from degrading. The final section of our analysis deals with the assertion that metals recycling offsets primary production. Although the question of offsetting is a complicated issue, sufficient data are available for the U.S. aluminum market to evaluate how well the assertion might hold. We analyze the rise in U.S. secondary aluminum production that began in the late 1980's and conclude that increased consumption of secondary aluminum (i.e., aluminum produced from scrap) was driven by demand from an expanding market for automotive components and had little impact on the production of primary aluminum.

While the focus of this work is to provide a critique of the metals industry's position as stated in Atherton (2007), it is equally important to reinforce the theoretical potential of metals to be recycled in the manner stated by the industry. There is a sound metallurgical argument in Atherton (2007) that supports the belief that metals recycling offsets primary production. The inherent properties of metals do support indefinite recycling and contaminants can be removed from scrap to yield high-purity metal. However, the realization of this potential has been hindered by the economic limitations of current recycling systems. Improvements to recycling infrastructure and the adoption of more recycling-friendly alloys would help move metals recycling much closer to the desired vision of the metals industry.

This work identifies the differences that can exist between the metallurgical, theoretical view of metals recycling and the observed, economics-driven practice. Ultimately, metals recycling is framed by metallurgy and determined by economics. The economic considerations associated with supplying and consuming scrap metal are implicit in our qualitative arguments when they are not stated outright; however, quantitative economic analysis must be called upon to provide a more robust answer to the long-sought question of recycling displacing primary production.

4.2 "Metals from End-of-Life Products are Widely Recycled at High Rates"

In general terms, secondary metals are produced from new and old metal scrap and primary metals are produced from mined mineral ore. New scrap is generated during the production of semi-fabricated (i.e., metals that are in an intermediate, not fully finished, form) and finished products and its supply is therefore a function of the overall demand for metal, the product mix, and production technology. New scrap is generally of a known origin, a uniform consistency, and relatively free of contaminants. Due to its relatively homogenous nature and high quality, nearly all new scrap is considered to be collected for recycling nearly immediately after generation. Old scrap is generated once a product reaches the end of its useful lifetime. Product lifetimes range from less than a year for aluminum beverage cans to decades for building and construction products. Old scrap generally has a more mixed composition and contains more contaminants than new scrap. The economics of old scrap recycling, unlike the majority of new scrap grades, is tied more closely to the ability to remove these contaminants and to recover high-quality metal.

One of the defining characteristics of metals according to Atherton is the maturity of their old scrap recycling markets. In the view of the metals industry, materials other than metals, such as paper products and plastics, lack economically-justified recycling markets and would otherwise be landfilled or incinerated after their useful lifetime. Conversely, the strong economic incentive to recycle metals has created mature recycling markets and conditions where “metals from end-of-life products are widely recycled at high rates” (Atherton 2007, 59).

Our first question to investigate therefore is: “how do the old scrap recycling rates of iron and steel, aluminum, and copper compare with a material said to have less economical recycling markets such as paper?” We define the old scrap recycling rate as the fraction of metal collected from products that are retired and disposed in a given year t,

$$\text{old scrap recycling rate}_t = \frac{\text{metal collected}_t}{\text{old scrap metal retired and disposed}_t} \,. \tag{4.1}$$

Interestingly, while the average recycling rate of municipal paper and paperboard for the U.S. in 2000 – 2007 was 48% (U.S.EPA 2008), aluminum and copper are recycled at rates between 30% and approximately 40% (McMillan et al. 2010; Spatari et al. 2005; Zeltner et al. 1999; and Sibley et al. 1995). Iron and steel recycling rates range from 50% to 73% (Müller et al. 2006; Fenton 2004), which is only slightly better than municipal paper and paperboard. The rate of copper recycling has been found to be slightly higher in Europe than in the U.S. and estimates range widely from 48% (Bertram et al. 2002) to 67% (Ruhrberg 2006).

The caveat to the recycling statistics referenced above is that they are highly sensitive to the assumed average lifetime for each average product category. For instance, Zeltner et al. utilize scenarios of product mean residence times to estimate that the U.S. recovered between 31% and 74% of copper from retired and disposed products in 1990; the most realistic scenario yields a rate of 42%. Nonetheless, these studies represent an established methodology for estimating the mass of a specified material that is retired and disposed in a given year. By and large the statistics indicate that metals are not always recycled at high rates. The implication for environmental emissions is that without recycling there is no secondary metal production and no associated environmental benefit, no matter the method of treating recycling in LCA.

4.3 "Metals Can be, and are, Recycled Over and Over Again"

In principle metal atoms can stay in use indefinitely, passing from one product to another, and to another, etc. due to their infinite recyclability. However, Atherton's statement that metals are recycled over and over again is not consistent with analyses of recycling practices. For instance, Markov chain analyses have estimated that copper is used 1.9 times on a global scale (Eckelman and Daigo 2008) and steel is used 2.67 times within the Japanese economy (Matsuno et al. 2007) before ultimate disposal in a landfill or loss to the environment. Wood pulp is estimated to be recycled 2.2 to 3.0 times in the Japanese economy (Hiroyuki, Y. et al. 2006)

Although a Markov chain analysis for aluminum has yet to be published, it is likely that the estimated number of uses would be similar to that of steel and copper. The key parameters in the analysis (i.e., new and old scrap recovery rates, product lifetimes, and fraction of consumption by end-use market) are much the same for the three metals. A sampling of these parameters is included in the supplementary material (SM).

These examples of the Markov chain approach do provide informative characterizations of metals use. However the examples are subject to certain limitations, most notably their lack of accounting for changes in consumption by end-use market. For instance, Eckelman and Diago (2008) rely on consumption characteristics from the year 2000, which have likely evolved to some extent as the end uses of copper have changed. The number of times a given unit of copper is recycled could rise if there were increases in the recycling rate and demand for secondary copper products, but evidence presented later the following section indicates that this has not yet been the case. Taken as a whole these results provide evidence that current recycling systems fall far short of exploiting the infinite recyclability of metal atoms.

The ramification of this gap is clear for allocating production emissions off-sets to primary material production. If metals are recycled only a small number of times before being disposed and ultimately becoming unavailable for future use, there remains a need for primary material if market demand for the metal is constant or increasing. This requirement for replacement material limits the degree to which metal recycling displaces the environmental emissions of primary production. As described in the subsequent section, if separate markets exist for products whose material property requirements will allow the use of old scrap and those that will not (e.g., aluminum sheet), the number of times a metal is recycled will have little bearing on whether primary production is displaced or not.

4.4 "The Constraint to Metals Recycling is the Availability of Feedstock Material"

Atherton (2007) identifies material availability as the constraint to recycling. While it is also mentioned that material may not be economically recovered at the end of life, the discussion of metals recycling would benefit greatly from a more detailed review of other considerations, particularly old scrap quality, that play a critical role in determining if and how metals are recycled. In the following sections we further explore how scrap quality constrains the recycling of aluminum, copper, and steel.

4.4.1 Quality Considerations in the Use of Secondary Aluminum

There are two general forms of finished aluminum products: wrought (i.e., rolled, extruded, or forged) and cast. The requirements of the product system impose constraints on the physical properties of the alloy and, as a consequence, on their chemical composition (metal grade). This can limit the type and amount of scrap that can be utilized for each alloy. However, there is no physical constraint to alloying primary aluminum for both wrought and cast alloys.

Casting alloys generally contain mostly secondary aluminum, though they require some addition of primary aluminum to dillute contaminants to an acceptable level (Das et al. 2007). Conversely, wrought applications require a different alloy that is designed for higher strength and ductility. Since cast aluminum alloys tend to have three or more percent silicon, they are unsuitable for use in wrought alloys after recycling. Even old scrap that is clean and sorted serves only as a minor input for the production of wrought alloys due to the sensitivity of wrought alloys to impurities (Kevorkijan 2002). The exception to this the recycling infrastructure that has evolved for the aluminum used beverage can (UBC). Here, UBCs are collected and remelted in a closed-loop system, which directly returns UBCs to make new aluminum cans. The keys to this system are that the UBCs are segregated from contaminting metals and that the wrought alloys used in for can are designed to accommodate direct use of the remelted UBCs.

The division between the markets for wrought products (made mostly of primary aluminum) and cast products (made mostly of secondary aluminum) is also mentioned in a number of economic analyses of the aluminum industry. Deadman and Grace (1979) note the separation of the wrought and cast markets and describe the contaminants encountered during recycling as the reason why primary aluminum and secondary alloys are not substitutes. Bloomberg and Söderholm (2009) state that secondary alloys are used mainly for cast products in the Western European market. In the analysis of primary and scrap price ratios, Xiarchos (2006) found that neither new nor old aluminum scrap prices share a long-term relationship with primary aluminum price over the period of 1985-2000. This finding lends support to the view that separate markets exist for products made of mostly primary aluminum and those that are made with mostly scrap.

4.4.2 Quality Considerations in the Use of Secondary Copper

In addition to the categories of "primary" and "secondary" other basic forms of copper can be distinguished. These include unrefined copper, refined copper, and coppery alloys. Unrefiend copper refers to intermediate forms that have not undergone electrolytic refining, such as black copper and blister copper; refined copper contains at least either 99.85% copper by weight, or 97.5% copper by weight (International Copper Study Group (ICSG) 2010); and copper alloys include brass, which is copper alloyed with up to 45% zinc, and bronze, which is copper alloyed with 12-16% tin.

It is important to note that refined copper can contain both primary and secondary metal that has been re-refined. Old scrap can be re-refined into high purity, high-conductivity copper for electrical and electronic applications, as long as the levels of impurities are not prohibitively high. Evidence of a long-run relationship between the prices of primary copper and unalloyed old copper scrap is provided by Xiarchos (2006). Old scrap that cannot be re-refined to high purity copper is relegated to use in copper alloys. This is a reflection of the economics of copper recycling as low purity scrap requires complete smelting and converting (Richardson 2000). Ayers et al. (2002) state that the castings market is driven by "the supply of secondary copper that cannot be purified sufficiently for use as wire" (35). This separation is born out in data on reported scrap consumption by end user type (e.g., ingot makers, refineries, and brass and wire-rod mills). In 2006 and 2007, U.S. brass and wire-rod mills were responsible for 90% of the new scrap consumption and less than 10% of old scrap consumption, while ingot makers consumed nearly 60% of all old scrap (USGS 2010).

Electrical and electronic applications, which require pure refined copper, have grown to become the predominant end-use market and are responsible for more than 70% of copper consumption (Henstock 1996). Although old scrap can theoretically be re-refined to meet the requirements of electrical applications, the fraction of total world consumption that is secondary production has fallen from 18% in 1966 to 13% in 2005 (Gómez et al. 2007). While true that this trend reflects growth in total copper consumption outstripping secondary production growth, the authors estimate that increased availability of old scrap has mostly kept pace with total consumption. These results support the observations of old scrap recycling rates, which indicate that the copper industry recovers roughly half of all old scrap that is generated. Gómez et al. find that secondary production is much more closely related to old scrap flows than old scrap stocks, which constitute sources of copper that are relatively more costly to recover.

Similar to the global industry, the U.S. secondary copper industry has also experienced a decrease in reliance on old scrap: the annual mass of copper in old scrap consumed decreased by 71% between 1990 and 2008, from 536,000 metric tons to 155,000 metric tons (USGS 2010). While the mass of old scrap consumed has fallen, the mass of new scrap consumed has remained relatively stable. The two trends result in the ratio of old scrap to new scrap consumption plummeting from 80% in 1990-1991 to 45% in 1998 to 22% in 2008 (USGS 2010). Jolly (2000) indicates that this trend is related to increased new scrap collection from rising manufacturing, and shuttered processing capacity, increased exports, and decreased collection of old scrap. Indeed, the export of old copper scrap increased nearly three-fold between 1990 and 2008, from 324,000 metric tons to 908,000 metric tons (USITC 2010).

Although it is possible for copper old scrap to be re-refined for use in high-purity applications, contaminants can still render scrap suitable only for copper alloys like brass for valves and bronze for stautary. The large gap between total world stocks and consumption of old copper scrap indicates that recycling this material may be cost prohibitive at current copper prices (Gómez et al. 2007). This economic barrier is likely due to a combination of factors, such as products containing low concentrations of copper, or a recycling infrastructure that is insufficiently developed to recover high quality copper. Unlike aluminum, where old scrap recycling is largely constrained by the differences between wrought and cast alloys, the limits to old copper recycling appear to be the difficulties of economically recovering copper from existing stocks.

4.4.3 Quality Considerations in the Use of Secondary Iron and Steel

The use of secondary metal in the production of steel is also subject to quality considerations, although it is less of a constraint as in the case of copper and much less of a constraint in the case of aluminum. The current basic oxygen furnace (BOF) process produces primary steel and is limited to a scrap input of 30%. On the other hand secondary steel produced by the electric arc furnace (EAF) route for can be sourced from 100% scrap (Fenton 2004), as well as direct reduced iron (DRI) and pig iron. EAF steel can be tailored to many applications, but BOF steel is mostly used for products requiring rolled steel. The use of scrap iron and steel will not be universal as long as high scrap content EAF steel is unable to meet material property requirements for certain products, such as automotive body panels and packaging.

Due to the different scrap tolerances for BOF and EAF steel, it is not appropriate to follow Atherton (2007) and assume that the recycling of any given mass of steel by default offsets emissions from the BOF process. A more appropriate approach would be to first note what type of steel is used in a product (e.g., BOF cold rolled sheet) and use an emissions intensity that is calculated based on measured life cycle emissions for that type of steel. This would avoid assumptions of how iron and steel scrap is recovered and utilized decades into the future when the product is eventually retired and disposed.

4.5 “Metal Recycling Offsets Primary Production Processes”

Here we evaluate the ability of secondary aluminum to offset primary production. Since it was first commercially produced, aluminum has experienced its greatest increases in consumption when new product applications have emerged. McMillan et al. (2010) identify a distinct period from 1986 to 2006 where aluminum consumption rapidly increased. The article also estimates in-use stocks of aluminum following a logistic growth trend. During this period non-UBC old scrap consumption also experienced significant growth. Figure 4.1 depicts this growth in old scrap consumption relative to the growth in U.S. aluminum producer net shipments less shipments of aluminum for cans. Producer net shipments data provide a measure of industry output to markets and are calculated as the U.S. gross shipments minus the sum of domestic producers' receipts (Aluminum Association (AA) 2008a). Unless identified otherwise, the measures include imports and exports. Note that producer net shipments data starting in 2001 include Canada, while old scrap data are for the U.S. only.

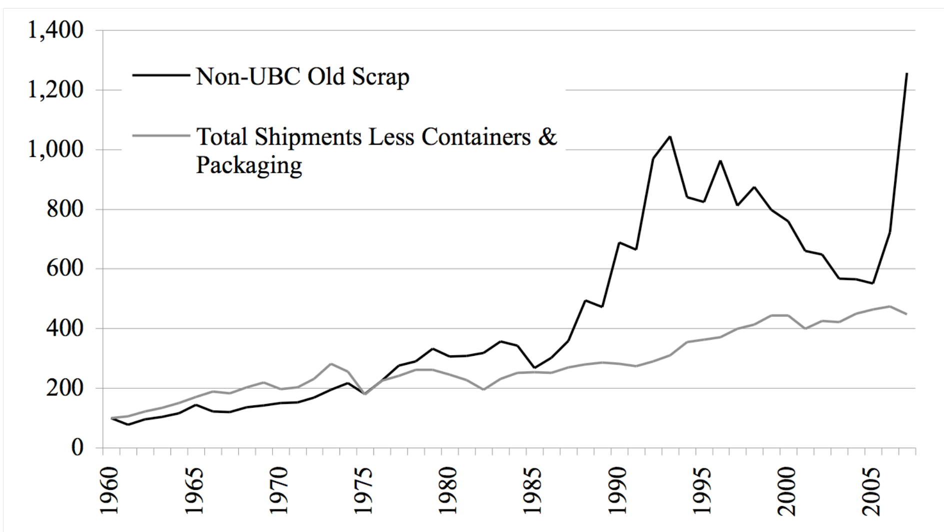


Figure 4.1 Growth of Non-Used Beverage Container Old Scrap Consumption (USGS 2009) and Total Shipments Less Containers and Packaging (Aluminum Association (AA) 2008a), Rebased (1960 = 100)

We contend that increased non-UBC old scrap and secondary aluminum consumption experienced during the period from 1986 to 2006 was the result of large-scale adoption of cast aluminum components in cars and light trucks and did little to affect the production of primary aluminum. Additionally, the decrease in old scrap consumption experienced after 1993 was compensated by increased new scrap recovery and increased imports of alloyed aluminum and not additional production of primary aluminum. These observations and conclusions are consistent with the limited substitutability of aluminum produced for castings, which is predominantly sourced from secondary metal, and aluminum produced for wrought products, which are predominately sourced from primary metal.

Analyzing the trends of the U.S. aluminum market provides the foundation for an applied economic analysis of the interactions between the primary and secondary aluminum markets. This is the type of analysis that is necessary to quantify the extent to which aluminum recycling displaces primary production. Even though the examination of market trends is qualitative in nature, it nonetheless draws important conclusions about how the economics of aluminum recycling determine how secondary material is supplied and consumed. The market trends illustrate how non-UBC old scrap supply and demand bear little relation to the consumption of primary aluminum. This adds support to the manuscript's thesis that metals recycling does not necessarily displace primary production. The recommendation that a quantitative economic analysis be developed in the future is discussed in the Summary.

Our evaluation does encompass the U.S. market system, including international trade flows. The inclusion of international trade data captures some of the influence of aluminum markets in foreign countries and is a sufficient addition to the analysis given its largely qualitative nature. Extending the system boundaries of the analysis to include the fate of aluminum after it is exported to a foreign country is greatly constrained by the availability of detailed, public aluminum production and consumption data.

UBC scrap (approximately 60% of total old scrap consumption during the same period (USGS 2009)) is excluded from the analysis due to the closed-loop recycling system of aluminum beverage cans. This closed-loop system represents an instance where recycling metal does offset primary production.

4.5.1 Old Scrap Demand Derived from Secondary Ingot Consumption

Because old scrap is an input for the production of secondary alloys, old scrap demand is derived from the consumption for aluminum products manufactured from secondary alloys, namely cast products such automotive engine blocks and transmission housings. An indication of this association is the short-run elasticity for secondary alloy demand with respect to automotive production, which was 0.52 as calculated by Blomberg and Hellmer (2000).

Cast aluminum was first widely used in the transportation sector in the years leading up to the Great Depression and in the 1920's the automotive industry consumed over half of primary and secondary production (Wallace 1937). This trend was short-lived and use in automobiles plummeted during the 1930's. Renewed interest in automotive applications did not reemerge until the oil shocks of the 1970's (Schatzberg 2003) and it took another decade before significant and wide-spread use of aluminum components began.

Collection and consumption of old scrap stagnates without demand from markets that can cost-effectively utilize old scrap to meet product material property requirements. It is difficult to imagine with today's sophisticated automobile recycling infrastructure that not long ago the U.S. suffered from what was called the "junk automobile problem" (Adams 1973). Automotive hulks in the mid-1960's began accumulating in auto wreckers' yards due to the contemporaneous factors of surging vehicle sales and retirement, and the transition from open-hearth to BOF steelmaking (Adams 1973). The BOF process utilizes less scrap than the open-hearth process and, as an additional impediment to recycling, #2 bundles formed from vehicles were low-quality scrap due to contamination by non-metallic materials and nonferrous metals. Recycling the large stock of vehicle hulks did not occur until later in the decade with widespread use of the hammermill auto shredder, which enabled separation of metallic and nonmetallic fractions, and the emergence of EAF steelmaking, which can utilize 100% scrap (Adams 1973).

Returning to U.S. aluminum market, figure 4.2 depicts the growth in ingot domestic producer net shipments to the transportation sector, producer net shipments of all ingot as a fraction of producer net shipments of aluminum (ingot and mill products exclusive of cans), and total scrap consumption (less UBCs) by secondary smelters. These data include imports, but not exports. For the period of 1970 to 1990, ingot shipments maintained a stable fraction of domestic aluminum shipments at 25%. As the design of cars and light trucks once again incorporated cast aluminum components, the demand for secondary aluminum increased markedly.

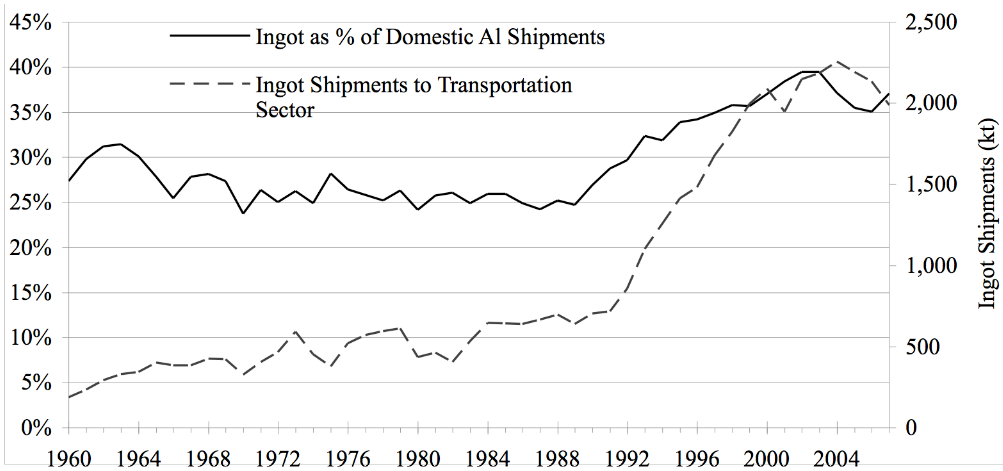
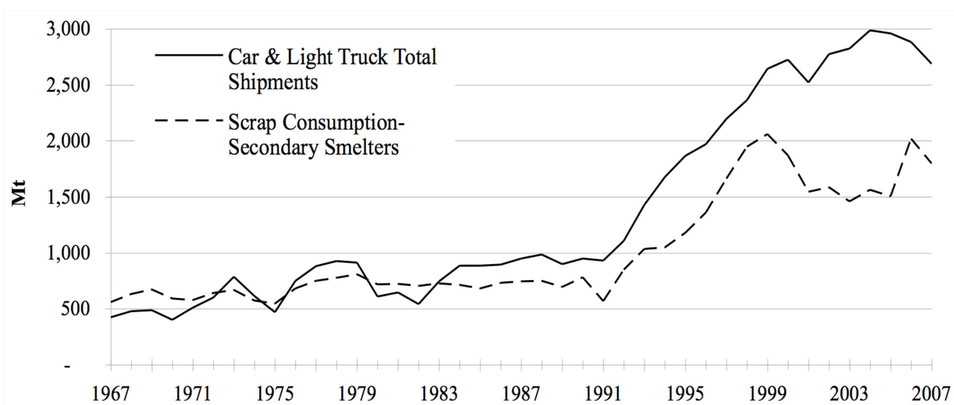


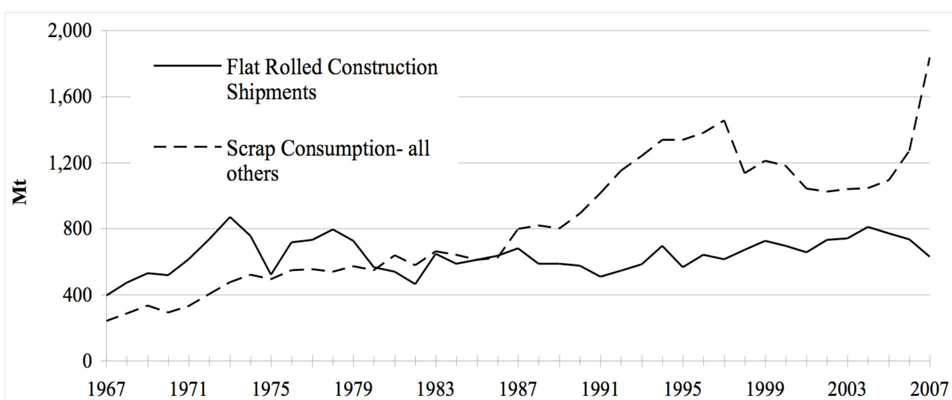
Figure 4.2 Ingot Shipments to Transportation Sector and Total Ingot Shipments as a Percentage of Domestic Shipments (exclusive of containers and packaging) (AA 2008a)

Although ingot shipments for cars and light trucks likely account for a large portion of non-UBC old scrap consumption, the Aluminum Association (AA) states that the total recycled content of flat rolled products (i.e., sheet and plate) in the building and construction sector is 85%, of which 60% is old scrap (AA 2008b). Building and construction flat rolled products accounted for an average of 7% of aluminum producer net shipments during 1990 – 2007, compared to 19% for producer net shipments of ingot for automobiles and light trucks. Even with a smaller share of aluminum producer net shipments, it would be expected that changes in demand for flat rolled construction products would be related to changes in demand for scrap. The correlation between flat rolled construction products and old scrap consumption is explored in the proceeding paragraph.

Figure 4.3 presents the producer net shipments of flat rolled products (i.e., sheet and plate) for the building and construction sector and producer net shipments of ingot and mill products for passenger cars and light trucks (AA 2008a). Total scrap consumption exclusive of UBCs by facility type is also included in the figure: scrap consumption by secondary smelters is depicted in figure 4.3a, while scrap consumption by integrated aluminum companies, foundries, independent mill fabricators, and other consumers (i.e., "all others") is depicted in figure 4.3b. The typical view of scrap consumers is that secondary smelters (also known as "refiners") consume mostly old scrap to produce castings and integrated producers, refiners, and fabricators consume new scrap to produce wrought products (Blomberg and Hellmer 2000). What is readily apparent from the figure is that the total aluminum net shipments for cars and light trucks are more closely correlated to scrap consumption by either facility group than producer net shipments of flat rolled construction products. Quantifying the Pearson correlations (ρ) for the period shows a value of 0.95 (95% confidence interval of $0.94 \leq \rho \leq 1.0$) for car and light truck net shipments and scrap consumption by secondary smelters and 0.21 (95% confidence interval of $-0.10 \leq \rho \leq 0.49$) for producer net shipments of flat rolled products for construction and scrap consumption by integrated aluminum producers, refiners, and other facilities. This simplistic analysis indicates that demand for non-UBC scrap is not necessarily associated with demand outside of the car and light truck market, which runs contrary to the statement of the AA



(a)



(b)

Figure 4.3 Comparison Between Shipments (AA 2008b) and Scrap Consumption (USGS 2009) for Cars and Light Trucks (a) and Flat-Rolled Construction Products (b)

4.5.2 Increased Recovery and Consumption of New Aluminum Scrap

In the mid-1990's secondary smelters were faced with a shrinking recovery of non-UBC old scrap, as was shown in figure 4.1. Evaluating data on new scrap generation and consumption reveal that these consumers turned to new scrap as an alternate source of material. The consumption of new scrap and non-UBC old scrap by secondary smelters is shown in figure 4.4, which indicates that new scrap grew to become an important source of input material during the time that ingot shipments for transportation were rapidly increasing. Secondary smelters' share of total new scrap consumption was in a general decline from 1960 to 1991, falling from 76% to 25%. However, the share increased to 59% by 1999 and remained around 55% through 2007. The reversal of this trend corresponds to the same period when ingot net shipments to the transportation sector rapidly increased.

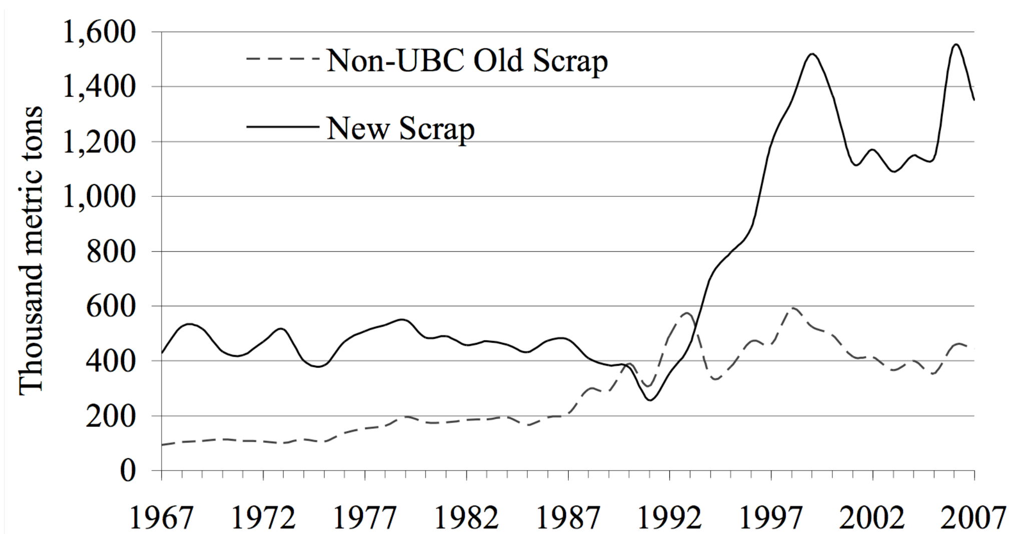


Figure 4.4 Secondary Smelter Consumption of New Scrap and Non-UBC Old Scrap (USGS 2009)

The large increase in new scrap consumption by secondary smelters that began in 1992 was accompanied by an increase in the total amount of new scrap consumed in the U.S. The general assumption regarding the generation and consumption of new scrap is that it closely follows the total amount of aluminum used in an economy. Until the early

1990's secondary recovery from new scrap did indeed follow apparent consumption. From 1946-1993, the Pearson correlation coefficient of the two series is 0.99 (95% confidence interval of $0.98 \leq \rho \leq 0.99$). In the mid-1990's this relationship changed and growth in new scrap consumption vastly outpaced that of net shipments and apparent consumption. From 1994-2007 the correlation coefficient is -0.19 (95% confidence interval of $-0.65 \leq \rho \leq 0.38$).

Much of increase in new scrap is the result of improved recovery from dross and skimmings. In 1990 approximately 65,000 metric tons (6% of total new scrap consumed) of aluminum was recovered from dross and skimmings; by 1995 recovery had increased to 224,000 metric tons (12% of total new scrap consumed) (USGS, 2009). Based on their increasing share of new scrap consumption it appears that secondary smelters were largely the recipients of the increased new scrap supply.

4.5.3 Evaluating the Offset of Primary Aluminum

Estimates of the total primary and secondary aluminum consumed in the U.S. can be generated by incorporating trade data on unwrought alloyed and unalloyed aluminum with existing data on scrap consumption and primary production. This section essentially takes a mass balance approach in its estimation of total consumption; however, the goal is not to develop a detailed accounting of all aluminum mass flows. Instead, the goal is to explore the underlying trends of secondary and primary consumption in light of the contemporaneous trends for old scrap consumption and new scrap consumption examined in the previous two sections.

The analysis of these data shows that even with the increased recovery of new scrap, U.S. consumption of new scrap and non-UBC old scrap decreased from 1999 to 2003. However producer net shipments of aluminum to the automotive market continued to increase. These producer net shipments likely came from imports of alloyed ingot from abroad, although it is not possible to disaggregate trade data in a way to quantify or classify these imports. This analysis supports the position that the demand for products that predominantly utilize secondary aluminum has little effect on the production and consumption of primary aluminum.

Although it is not possible to know the fraction of primary and secondary alloys in the net trade of unwrought aluminum alloys, we assume that 100% are consumed for castings. This assumption is made to evaluate whether or not a large increase consumption of aluminum for castings is associated with any change in the consumption of primary material for wrought products. Estimates of total primary consumption are then obtained by adding primary production, net trade of unalloyed aluminum, and imports of semi-fabricated products. Total secondary consumption is estimated by the sum of net trade of unwrought alloys and total old scrap consumption.

Even with using the upper bound estimate that 100% of the net trade of unwrought alloyed ingots is consumed for castings, the estimated total consumption of primary and secondary aluminum follows the general movement of producer net shipments of mill products and ingots respectively. Figure 4.5 does not reveal a response in primary consumption from the large increase in secondary consumption in the early 1990's. Estimated production of secondary aluminum more than tripled between 1991 and 1999, yet there is little evidence from a graphical analysis that this rapid increase had much, if any, effect on primary consumption. More likely both primary and secondary consumption were responding to demand in their respective markets and little, if any, displacement of primary production occurred.

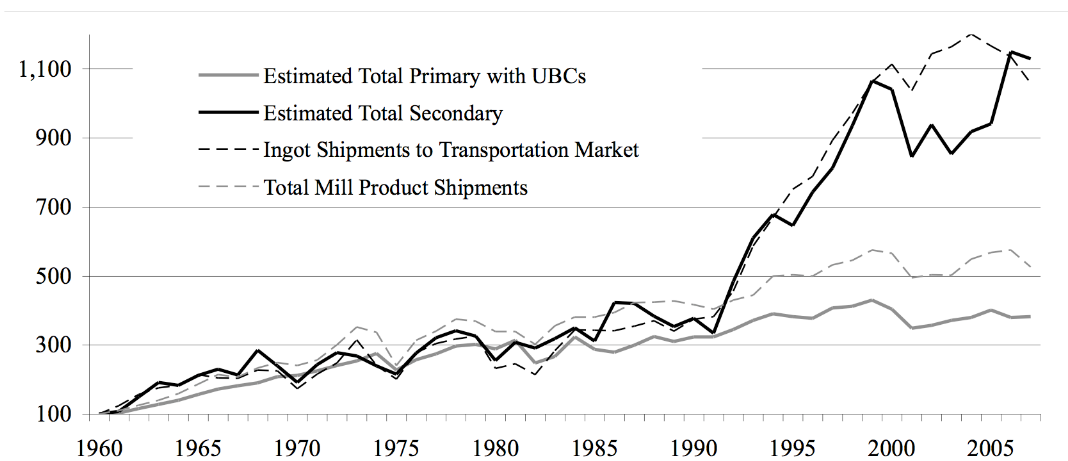


Figure 4.5 Index of Estimated Consumption of Primary and Secondary Aluminum, Rebased (1960=100

4.6 Summary

The purpose of this analysis is to establish that the basic tenets articulated by Atherton (2007) reflect the metallurgical potential of metals recycling and not the economic realities of current practice. The claims that: 1) metals are widely recycled at high rates, 2) metals are recycled over again, and 3) the constraint to metals recycling is strictly the availability of feedstock material serve as the foundation for the metal industry's position that recycling directly displaces primary production, and therefore reduces the net effect of environmental emissions from metals production processes. The complex and dynamic behavior of metals recycling precludes its distillation to a set of universal "facts" and the blanket statement that recycling displaces primary production ignores the effects of the economics of removing contamination accumulated during recycling and the different tolerances for contamination across alloys. Rather than generalizing the behavior of metals, we recommend that the metals industry revise its stance to acknowledge the complexities of recycling and take a more nuanced view of the unique characteristics of individual metals and their recycling systems.

This paper has identified evidence that iron and steel, aluminum, and copper are not necessarily recycled at high rates and significant constraints on use of scrap exist. The paper has also provided evidence that supports the hypothesis that the consumption of secondary aluminum, while beneficial in terms of nonrenewable resource conservation and reduced emissions, is currently limited in its ability to offset primary production and its environmental emissions outside of the aluminum beverage can system. The combination of modest recycling rates (similar to municipal paper and paperboard products), the sustained demand for metal containing products, and the likelihood that metals are recycled only a small number of times before ultimate disposal indicate the inevitability of primary metal production and associated emissions. Old scrap recycling has a particularly limited ability to offset emissions from primary aluminum due to the contamination limits of wrought products.

The graphical analysis of aluminum market trends was performed only for the U.S., although interactions with foreign markets were captured to some extent by the inclusion of international trade data. Performing the same analysis for other markets may reveal different trends in aluminum production and consumption, but similar conclusions are likely to be reached regarding primary production displacement as long as the economics governing old scrap consumption are comparable. Discussion in Blomberg and Söderholm (2009), Blomberg and Hellmer (2000) and Peck (2003) indicates that the Western Europe shares similarities with the U.S. market.

The question of to what extent metals recycling displaces primary production is best answered by a quantitative economic analysis. In economic terms this question becomes an exercise in quantifying the substitution between old scrap and primary metal. This type of analysis is beyond the scope of this paper, but it would involve first determining the appropriate production function for each type of metal/alloy (e.g., cast aluminum, wrought aluminum, refined copper, copper alloy, BOF steel, and EAF steel) and then econometrically estimating the associated substitution between old scrap and primary metal. An alternate approach would be to develop an econometric model of the market for each type of metal/alloy and estimate cross-price elasticity of demand (i.e., the percentage change in demand of good x associated with a 1% change in the price of good y).

While we find it necessary to identify the disparities between the views of the metals industry and the current economic realities of recycling systems, we fully recognize the theoretical potential of metals recycling from a metallurgical standpoint. Indefinite recycling of secondary metal could occur if detrimental contaminants were not accumulated through successive recycling, or could be cost effectively removed or diluted, or if alloys were without tolerances for contaminants. Scrap contamination and variation in alloy tolerances have implications for the economics of scrap recycling. No matter the metallurgical possibilities of metals recycling, there will be little, if any, demand for scrap sources that are uneconomic to recycle.

Many opportunities have already been identified for increasing metals recycling. The ability to minimize contamination is especially important for aluminum, but less so for copper. For aluminum, studies have identified means of negating the impact of scrap contaminants and increasing the use of secondary aluminum, such as the development of recycling friendly alloys (Gaustad et al. 2010; Das et al. 2007; Das 2006; Gesing and Wolanski 2001) and improved scrap sorting technology (Gesing 2004; Gesing and Wolanski 2001). For copper, large technological improvements are possible over existing methods of recycling electronic waste that increase metal recovery as well as reduce process emissions (Hagelüken 2006). Ilgin and Gupta (2010) have provided a review of many product design and manufacturing strategies for increasing and improving recycling. Improvements in infrastructure systems for segregation of metals and greater participation rates in recycling programs would certainly move the current state of recycling much closer to what the metals industry has envisaged in Atherton (2007).

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CHAPTER 5

TEMPORAL CONSIDERATIONS OF ALLOCATION APPROACHES APPLIED TO ALUMINUM RECYCLING

ABSTRACT

Allocating the environmental burdens of metals recycling is a contentious issue in life cycle assessment (LCA). LCA practitioners have developed numerous approaches, yet few studies have quantitatively compared their results. Consequential approaches, which aim to quantify the indirect effects of recycling, in particular have received very little scrutiny of their suitability for metals. These approaches most often rely on assumptions of market behavior and introduce layers of complexity not encountered in traditional LCA. We analyze the performance of four allocation approaches for calculating greenhouse gas (GHG) emissions for aluminum. Two of these approaches, value corrected substitution (VCS) and end-of-life recycling (EOLR) have been specifically advocated for application to aluminum. We find that the VCS and market-based approaches fail to capture the temporal nature of aluminum recycling. The EOLR and recycled content (RC) approaches are analyzed using two case studies: the U.S. aluminum beverage can market and a hypothetical fleet of vehicle engine blocks. The EOLR approach is found to distort the timing of material flows and emissions relative to the RC approach in both case studies. In the case of the engine blocks, emissions associated with initial production account for over 99% of total GHG emissions using the RC approach and 36% to 50% using the EOLR approach. Additionally, estimated total GHG emissions are 18% and 79% larger using the EOLR approach than the RC approach. The distortion of the timing of emissions has particular implications for climate change, as well as the time-dependent impacts of ozone, nitrogen oxides and other emissions.

5.1 Introduction

Certain metals and their alloys have been exploited by humans for millennia in the production of durable, long-lived products. These products are used for many years, even decades, before they are retired. Upon retirement these products are either recycled for secondary production or disposed to the environment. Because secondary metals (i.e., recycled metals) often offer economic and environmental advantages over primary metals (i.e., virgin, produced from mineral ore), recycling has long been associated with metals production. Recycling and remelting typically consume less energy than extracting the metal from its mineral form. This is particularly true for aluminum; it is generally accepted that producing secondary aluminum consumes 5% of the energy as producing aluminum from bauxite (International Aluminum Institute 2000).

While the environmental and energy advantages of secondary metals are relatively well established, how to quantitatively account for them in a life cycle assessment (LCA) framework is not. Metals are often recycled in an open loop, where recovered material from one product system is consumed in an entirely different product system. Also, metals have the theoretical ability to be infinitely recycled without losing their inherent atomic properties. The question that arises for a LCA is how to account for, or "allocate", the emissions associated with a mass of metal that can potentially be used across multiple product systems an infinite number of times.

The LCA community has formulated two general approaches to the open-loop recycling allocation problem. The first general approach is to apportion the environmental burdens of primary production between the initial mass of metal and the subsequent use(s) of the recycled metal. These approaches follow the recommendations of ISO 14041 (Anon. 1998) for multi-function processes: If it is not possible to avoid allocation in the first place, the approaches should be based primarily on physical relationships, or secondarily, on other relationships between environmental burdens and system functions. Additionally, ISO 14041 establishes that an open-loop recycling system may be treated as a closed-loop system and allocation avoided if the inherent properties of the material in question are not changed.

Examples of allocation approaches for metals that are based on physical relationships include cut-off/recycled content (RC) (Fava et al. 1991) and 50:50 (Fava et al. 1991), which equally distributes emissions from primary production and waste treatment to the first and last uses; these, along with other similar approaches, have been summarized in Werner (2005), Ekvall and Tillman (1997), and Klöpffer (1996). Other allocation approaches have been based on economic relationships, such as value-corrected substitution (VCS) (Werner 2005; Werner and Richter 2000a; Werner and Richter 2000b) and eco costs/value ratio (Vogtländer et al. 2001).

The second general approach for the open-loop recycling allocation problem follows the ISO 14041 recommendation that allocation should be avoided where possible, through either subdivision or system expansion (Anon. 1998). For metals this has typically followed the recommendation of system expansion. Examples include the end-of-life recycling (EOLR) approach (Atherton 2007; European Aluminum Association 2007), market-based (MB) approach (Ekvall 2000), and the parametric approach developed by Geyer (2008).

Recycling allocation can also be examined within the larger discussion of attributional and consequential LCA. Attributional LCA, which can be thought of as LCA in its traditional form, describes the relevant material and energy flows of the life cycle of a system and its subsystems (Finnveden et al., 2009). Attributional LCA can also be defined as describing the physical and energy flows of a system as they are at a specified point in time (Curran et al., 2005). These definitions are consistent with open-loop recycling allocation approaches that apportion environmental burdens based on physical or economic relationships, or number of uses. We will refer to this group of allocation approaches as attributional allocation approaches.

Consequential LCA expands the definition of attributional LCA by including the possible direct and indirect effects associated with changes in the life cycle of a system brought about by a decision (Curran et al. 2003; Weidema 2000). Open-loop allocation approaches that expand system boundaries in an effort to capture the indirect effects of the decision to recycle metal are consistent with the definition of consequential LCA. For example, the MB approach attempts to capture the additional, indirect consequences of a marginal change in the amount of secondary metal produced and consumed by the system in question. The EOLR approach treats the consequence of recycling as the reduction in primary production.

A small number of quantitative analyses have been performed that compare the results of attributional and consequential LCA (e.g., Schmidt 2010; Thomassen et al. 2008; Gamage et al. 2008; Merrild et al. 2008; Lesage et al. 2006). These studies generally indicate substantial differences in results based on the system boundaries drawn by an attributional or consequential approach. Less dramatic differences between the two general approaches are found in two studies performed for metals (PE Americas 2010; Ekvall and Andrae 2005). A study by Nicholson et al. (2009) finds that the choice of allocation approaches affects material selection decisions when the materials in question have similar primary energy burdens.

The research presented here aims to inform the metals recycling LCA allocation discussion by evaluating the market-based (MB), value-corrected substitution (VCS), end-of-life recycling (EOLR), and recycled content (RC) approaches as they are applied to aluminum. The MB and VCS approaches are specifically advocated for application to aluminum by Frees (2007), Werner (2005), the European Aluminum Association (EAA) (2007), and Werner and Richter (2000). The EOLR approach has also been advocated for analysis of aluminum (EEA 2005), in addition to being the acceptable approach for all metals irrespective of closed-loop or open-loop recycling (Atherton 2007).

What these four approaches and the aforementioned work on evaluating allocation in practice share is a lack of discussion and analysis of whether or not the approaches accurately account for the timing of emissions and other material flows. This is problematic in regards to the approaches advocated for aluminum and other metals, as time plays a significant role in economic and recycling behavior. Additionally, there are emissions with time dependent impacts (e.g., greenhouse gases (GHGs), sulfur oxides, nitrogen oxides, and ozone) that are accounted for in potentially different ways based on how each approach treats the temporal nature of metals recycling.

We examine the MB and VCS approaches and find that their underlying assumptions are not valid for aluminum. The MB approach fails to acknowledge the importance of stockholding behavior in storable commodities markets. The VCS approach was originally proposed with an incomplete understanding of the statistical properties of aluminum prices and its assumption of data stationarity is found to be incorrect for aluminum. We then analyze the temporal implications of the EOLR and RC approaches using two case studies—the U.S. aluminum beverage can market and a hypothetical fleet of automotive engine castings—and find that the EOLR distorts the timing of aluminum flows and emissions relative to the RC approach.

5.2 Value-Corrected Substitution (VCS) Approach

The VCS approach was developed as means of explicitly accounting for the degradation of material quality that occurs from recycling (Werner 2005; Werner and Richter 2000a). This approach assumes that primary metal is the highest quality, which is reflected in its market price and its price relative to the prices of old scrap and secondary alloy. Price ratios of primary and secondary metal are used to credit the amount of recycled material consumed in product manufacturing and generated at product end-of-life. This method was subsequently endorsed by the EAA for use when the market prices for primary and recycled aluminum products differ (EAA 2007).

The VCS procedure utilizes material price ratios to develop allocation factors α and β. For aluminum applications, α is the ratio of alloyed price to primary ingot price and β is the ratio of old scrap price to primary ingot price. The devaluation caused by recycling is represented by the difference $\alpha - \beta$. The value-corrected emissions from primary material production are calculated as the product of the difference of the price ratios and the unallocated primary material production emissions. The value-corrected environmental burdens of primary production (Epp) of product n (Pn) are then calculated as

(5.1)

where Epp are the unallocated environmental burdens.

The VCS approach assumes that the relative prices α and β reflect the extent to which alloyed ingot, primary ingot, and scrap may be used in aluminum products. In this assumption, a form of aluminum that can be used in more products will command a higher price relative to a form with a more limited use. Werner (2005) provides the example of the presence of iron and zinc contaminants limiting the use of casting alloys, which is captured in their low price relative to unalloyed primary aluminum. However, this assumption is violated in the case of the contracts for U.S. secondary alloy and London Metals Exchange (LME) primary aluminum. As figure 5.1 shows, the monthly average price ratio of American Metal Market (AMM 2010) secondary alloy to LME primary (i.e., "Pa:Pp", the ratio α) remained above 1.0 for much of the period from January 1985 to March 2010. Following the reasoning behind the VCS approach this indicates that secondary alloy, which is by and large used for castings, has a higher functionality than primary aluminum, which can be alloyed for any aluminum product. Furthermore, a ratio above 1.0 means that the production of secondary alloy would be ascribed environmental burdens above those of primary aluminum production.

The VCS approach was developed with the requirement that an allocation approach "contain no 'jumps' of the allocation factor" (Werner 2005, p. 180). This translates to the condition that the price ratio α is stable over time. The stability of the price ratios is also taken as an indication of their ability to reflect the relative material value of aluminum and of the competitive nature of aluminum markets.

Figure 5.1 shows the price ratio of old scrap (AMM 2010) to LME primary (i.e., "Pos:Pp" the ratio β) and the difference α - β. Figure 5.1 indicates that α and β do in fact vary; their difference reaches a maximum of 0.9 and a minimum of 0.29. This is much larger than the fluctuation of 4.5% identified by Werner for the α ratio. Such large fluctuations are not consistent with the assumption that the price ratios do not "jump" and are stable over time. For instance, if the annual changes in price ratios were reflected by the VC approach the allocation factor would be 10% higher for aluminum produced in 2009 than aluminum produced in 2008.

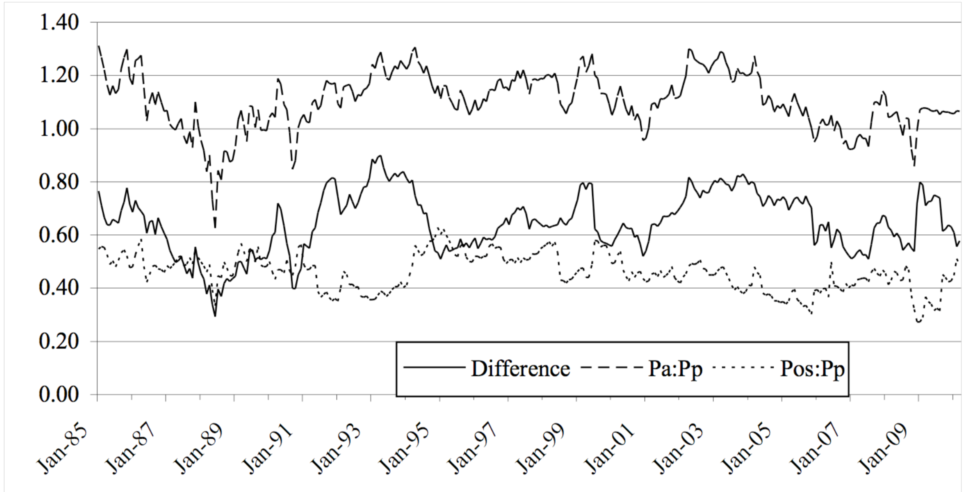


Figure 5.1 Price Ratios of Secondary Alloy to Primary (Pa:Pp), LME Secondary Alloy to Primary (Pa(LME):Pp), and Old Scrap to Primary (Pos:Pp)

Visual inspection of the aluminum price ratios may indicate that they contradict the VCS assumption of being stable over time; however, a more robust analysis is needed. The original support for the assumption of stable price ratios relied on rudimentary calculation of descriptive statistics, such as mean, median, variance, and correlation (Werner 2005). A more appropriate approach is to use statistical time series analysis and unit root testing to determine whether or not the price ratios are weakly stationary. A weakly stationary data series that does not have a unit root has a mean and variance that do not depend on time (Wooldridge 2006). If the price ratios are found to be nonstationary, their mean and variance are time-dependent and the assumption of the VCS method is no longer valid.

Several existing studies have tested the stationarity of aluminum prices. Watkins and McAleer (2006) conduct unit root testing on daily London Metal Exchange (LME) data for aluminum and aluminum contracts. Spot prices for both price series are found to contain unit roots over their selected periods. Lee et al. (2006) find evidence of a unit root in primary aluminum price under several structural break specifications. In addition to primary aluminum price, Xiarchos (2006) finds that prices of new clippings, used beverage containers (UBCs), and old sheet and cast scrap contain a unit root. Xiarchos goes further to find that primary and scrap prices do not share a long-run, stable relationship (i.e., they are not cointegrated) and that the ratios of primary price to new scrap price and primary price to old scrap price also contain a unit root.

The two price ratios shown in figure 5.1 are utilized to test the VCS approach's underlying assumption of data stationarity, as well as the price ratio of LME secondary alloy contract to LME primary aluminum. Prior to conducting stationarity testing, the sample autocorrelation functions (ACF) were calculated for each ratio series. The ACFs depict a high degree of persistence for each price series, which is potentially indicative of nonstationary data. The Phillips and Perron (PP) (Phillips and Perron 1988), the augmented Dickey-Fuller (ADF) (Dickey and Fuller 1979), and the Elliott, Rothenberg and Stock Dickey-Fuller Generalized Least Squares (ERS DF-GLS) (Elliott et al. 1996) root tests were selected as part of stationarity testing. The inability to reject the null hypothesis at a chosen confidence value in each of these tests indicates the presence of a unit root and nonstationarity. Additionally, the price ratios were subjected to stationarity testing with the KPSS test (Kwiatkowski et al. 1992). Unlike the three other unit root tests, the KPSS test defines the null hypothesis as stationary and rejection of the null implies nonstationary data.

The data series were also visually inspected for structural breaks. The presence of a structural break leads to a bias towards nonrejection of a unit root in the tests identified in the preceding paragraph. Based on inspection, Pa(LME):Pp and Pa:Pp potentially have single structural breaks in intercept and trend and Pos:Pp potentially has a single structural break in intercept. These data series were subjected to unit root testing using the Zivot-Andrews test, which endogenously estimates a single structural break (Zivot and Andrews 1992). Although testing did not indicate a structural break in Pa:Pp, breaks at April 2003 and at January 1994 were identified for Pa(LME):Pp and Pos:Pp respectively. The presence of a structural break and the aforementioned bias in the PP, ADF, and DF-GLS tests leaves Zivot-Andrews as the sole unit root test for Pa(LME):Pp and Pos:Pp.

Table 5.1 Unit Root Testing Results

Unit Root Test	Pa:Pp	Pa(LME):Pp	Pos:Pp
PP			
Test statistic	-3.821	na	na
Critical value @ 5%	-2.872	na	na
Unit root?	No	na	na
ADF			
Test statistic	-3.699	na	na
Critical value @ 5%	-2.87	na	na
Unit root?	No	na	na
ERS (DF-GLS)			
Test statistic	-1.107	na	na
Critical value @ 5%	-1.94	na	na
Unit root?	**Yes**	na	na
KPSS			
Test statistic	0.5548	na	na
Critical value @ 5%	0.463	na	na
Nonstationary?	**Yes**	na	na
Zivot-Andrews			
Break point	na	April 2003	January 1994
Test statistic	na	-4.139	-6.259
Critical value @ 5%	na	-5.08	-4.8
Unit root	na	**Yes**	No

Unit root testing results indicate a unit root in Pa(LME):Pp and reject a unit root in Pos:Pp even though its sample ACF exhibits high persistence. The Pa:Pp series shows mixed results. When considering the four unit root tests utilized for Pa:Pp, the ERS DF-GLS test is preferred to the ADF and PP tests when the deterministic component of the data series is unknown (Elliott et al. 1996). Together with the results of the KPSS, the conclusion is that the Pa:Pp series contains a unit root.

These results, in addition to the existing literature on the stationarity of aluminum prices, have serious implications for the assumptions of the VCS method as applied to aluminum. Unit root testing indicates that the price ratio β is nonstationary, which invalidates the critical assumption of a stable price ratio. Without a stationary β, the VCS method should no longer be considered an option for application to aluminum products.

5.3 Market-Based Approach

The MB approach (Ekvall 2000) assumes that consumption and generation of scrap from the system in question affects the consumption and generation of secondary scrap and consumption of primary material in the aggregate market. According to the approach, as long as the amount of scrap generated by the product system in question (ΔX) is small relative to the overall scrap market, the change in overall market supply (ΔS_X) and demand (ΔD_X) of scrap can be approximated by the price elasticity of supply (η_S) and demand (η_D). This is shown in (5.2) through (5.4):

$$\Delta X = \Delta D_X - \Delta S_X \approx \Delta D_X \left(1 - \frac{\eta_S}{\eta_D}\right) \tag{5.2}$$

$$\Delta D_X \approx \frac{\Delta X \eta_D}{\eta_D - \eta_S} \tag{5.3}$$

$$\Delta S_X \approx \frac{\Delta X \eta_S}{\eta_D - \eta_S} \tag{5.4}$$

The change in environmental burdens ΔB_X resulting from a change in the amount of material collected for recycling in the system in question is calculated as

$$\Delta B_X \approx \Delta D_X \left(R_{\mathrm{O}} - A V_{\mathrm{O}}\right) + \Delta S_X \left(C_{\mathrm{O}} - W_{\mathrm{O}}\right), \tag{5.5}$$

where R_{O}, V_{O}, C_{O}, and W_{O} are the environmental burden intensities of recycling, primary material production, recycling collection, and waste management, respectively, of other life cycles; and A is a constant that accounts for the condition that secondary material does not provide the same function as an equivalent mass of primary material. If, for instance, 2.0 kg of secondary material provides the equivalent function of 1.0 kg of primary material, scrap recovered from the product system would receive only 50% of the credit for offset emissions associated with decreased consumption of primary material. Nonetheless, the approach assumes that the generation of scrap offsets the consumption of primary material and an emissions credit is calculated based on the amount of scrap recovered in the system under study.

Proponents of the method acknowledge the difficulty associated with calculating the required price elasticities. A work-around proposed by Ekvall (2000) is to simply assume that a given mass of recycled material from the product system displaces 50% primary production and 50% of scrap collection from other sources. This is equivalent to assuming that supply and demand are equally elastic. Ekvall and Weidema (2004) propose an additional simplifying assumption that supply or demand is perfectly inelastic, which implies that scrap from the system in question only offsets either primary material or scrap from other systems. Additionally, Ekvall and Weidema propose the use of multiple scenarios to address the uncertainty associated with estimated elasticities.

The lack of suitable econometrically-derived elasticity estimates for old scrap greatly hinders the evaluation of the MB approach as it is applied to aluminum. The most recent econometric study of a secondary aluminum market estimates the price elasticity of supply and demand for secondary alloy (Blomberg and Söderholm 2009), not old scrap as is specified by the MB approach. Although Ekvall (2000) cites price elasticities for old scrap supply and demand, the estimates are not suitable for use in analysis because they are either estimated for a 70-year-old market (Suslow 1986) or for municipal aluminum (ICF 1979), which is largely UBCs.

Even if suitable old scrap elasticities made it possible to evaluate the approach in practice, the MB approach has a fundamental shortcoming in that it does not capture the temporal characteristics of aluminum and other commodities markets. The MB approach does not address the influence of stockholding, as well as other dynamic behavior of commodities. The consideration of stockholding behavior is important given that metal production is understood to be unresponsive to price (i.e., price inelastic) in the short run. Adjustments to stocks, which are more responsive to changes in price, help to equilibrate metals markets (Labys and Lord 1992). Acknowledging stockholding behavior is also important given the uncertainties that suppliers face in competitive markets. Producers use stockholding to reduce costs associated with stochastic shocks that affect production and delivery scheduling (Pindyck 2001; 1994). Weidema (2003) has identified temporal misrepresentation of market behavior in consequential LCA, but the proposed solution that scale and time horizon be extended is insufficient for characterizing the behavior of commodities markets. A more suitable approach to evaluating the market implications of changes to scrap availability is to introduce the concept of commodity storage. Williams and Wright (1991) develop a basic storage model for describing the interactions among commodity producers, consumers, and storers. Under a competitive market with storage, the storage industry will equate the expected price of the commodity in period $t + 1$ with the sum of storage costs k and the current spot price P_t. Equivalently, this central condition for competitive equilibrium with storage is

$$P_t + k - \frac{E_t\left[P_{t+1}\right]}{1+r} \geq 0 \quad S_t = 0, \tag{5.5}$$

where r is the interest rate and S_t is quantity stored at time t. When the expected price in period $t + 1$ is above the sum of current spot price and cost of storage, storers purchase the commodity. If expected price in period $t + 1$is below the sum of current spot price and cost of storage, no stocks are held. Equation (5) shows this latter relation.

The market supply in period t with stockholding is the sum of realized production h_t and the carryover of stocks from the previous period S_{t-1}. The total market consumption q_t is

$$q_t = h_t + S_{t-1} - S_t = A_t - S_t \tag{5.6}$$

where A_t is market availability (i.e., market supply) and S_t is storage in period t. Consumption is associated with price through the market demand curve; in the inverse form this relationship is

$$P_t = P\left[q_t\right], \quad \partial P/\partial q < 0. \tag{5.7}$$

Market equilibrium in period t can then be described using (5) through (7). In equilibrium, the sum of commodity price, a function of market availability and storage, and storage cost equal the expected commodity price in period $t + 1$. The expected price in period t is a function of planned production $\bar{h}_{t+1}$ and a random disturbance to production v_{t+1}; equivalently,

$$P_t\left[A_t - S_t\right] + k = E_t\left[P_{t+1}\left[\overline{h}_{t+1}\left(1+v_{t+1}\right) + S_t - S_{t-1}\right]\right] \Big/ \left(1+r\right). \tag{5.8}$$

In light of (5.8), it is clear that the equations governing the MB approach fail to capture much of the intertemporal behavior of commodity markets. A portion of the change in availability of scrap (ΔX) may be held as stock depending on the carryover of scrap from the previous period and the expected price of scrap in the subsequent period. The simple relationships between price elasticity and change in availability of scrap provided by the MB approach in (5.3) and (5.4) do not include terms related to storage. These equations are not sufficient to capture the indirect effects of recycling in their current form. As such, the MB approach in its current form should not be applied to any commodity subject to stockholding behavior.

By not including the effects of stockholding, the framework of the MB procedure has been misused to argue that recycling aluminum displaces primary production (Frees 2008). Frees uses the MB approach to argue that inelastic secondary alloy supply and an assumed linkage between primary and scrap prices results in aluminum recycling displacing 100% primary production. As indicated by Labys and Lord (1992), inelastic metal supply is compensated for by the use of stocks. Thus, the market response to inelastic supply can be explained by stockholding behavior, rather than making an assumption regarding the ability of recycling to displace primary production.

In addition to the lack of acknowledgement of temporal and stockholding considerations, the market-based approach does not address the spatial properties of scrap markets. This is particularly problematic for metals because of the large international flows of scrap. For instance, in 2005 U.S. scrap exports of iron and steel, aluminum, and copper represent 16.2%, 26.6%, and 40.6% of total supply (Lyons et al. 2009). This indicates that any analysis of the indirect effects of metal recycling should include the fraction of metal not consumed domestically and the emission factors for the foreign countries that ultimately consume the scrap.

5.4 Recycled Content (RC) and End-of-Life Recycling (EOLR) Approaches

The most methodologically straightforward of the allocation approaches is the RC approach (also known as the cut-off approach), which assigns environmental burdens to a system that are a direct result of only that system. This method is best thought of as describing “what is” because its calculations rely on observed, or otherwise estimated, data on the mass of primary and secondary metal consumed within the chosen system boundary. The approach does not attempt to describe “what ought to be”, does not credit future recycling activities, and does not assume the displacement of primary production by recycling.

The EOLR approach is the simplest of the consequential approaches. The approach is equivalent to a system expansion that creates a single, large pool of material where primary metal is only used to replace metal lost in unrecovered products and during melting. The consequence of recycling any amount of metal is the automatic offset of the equivalent amount of primary production and associated emissions. The metals industry has identified the EOLR approach as the single acceptable method of accounting for metals recycling in a LCA framework (Atherton 2007).

The ability of the EOLR approach to account for the temporal nature of recycling and GHG emissions is evaluated using case LCA studies of two distinct product systems. The first case study is a short-lived, closed-loop product system in the form of U.S. aluminum beverage can manufacturing from 1990-2000. The aluminum beverage can provides a unique opportunity to evaluate LCA recycling allocation procedures. Unlike many other aluminum products, data are available in sufficient detail to provide a relatively complete picture of the amount of primary and secondary aluminum consumed in production. Because the aluminum can has developed its own system of closed loop recycling, where UBCs and new scrap in the form of can stock clippings are recycled directly into new cans, whatever metal is not recovered from recycling scrap must be made up by primary aluminum. This is a case where recovered metal from UBCs does displace the consumption of primary aluminum and it would be expected that applying the recycled content and EOL recycling approaches to LCA allocation would yield essentially equivalent results. By including multiple years we are able to examine the effects of changes in UBC recycling rates, beverage can production, and aluminum GHG emission factors.

The second case study is a long-lived, open loop system of a hypothetical fleet of cast aluminum engine blocks. The focus in this case study is the temporal treatment of system mass flows. In addition to the engine block having a much longer life than the aluminum beverage can, taking a fleet-based approach allows examination of product retirement distributed across a number of years.

5.4.1 Case Study 1: U.S. Aluminum Beverage Can

5.4.1.1 System Overview

The most widely recognized example of a closed-loop metal recycling system is the aluminum beverage can. Cans have a short use phase and recovered UBCs are remelted with primary aluminum to produce new cans. More specifically, when UBCs are collected for recycling, they are first crushed and baled. The bales are then shredded and sorted to remove contaminants before being heated to remove paints and other coatings, a process known as delaquering. UBCs are remelted in combination with can manufacturing scrap and can stock home scrap, after which the can body and lid alloys are themomechanically separated. Each alloy is treated to specification, cast into ingots, and then rolled and trimmed as finished can stock.

Figure 5.2 summarizes the U.S. aluminum beverage can shipments and the collection, trade, and consumption of UBCs from 1990 to 2000. Later years are not included in the analysis due to the inclusion of Canadian data in the statistical reports of the Aluminum Association (AA) beginning in 2001. The mass of cans shipped is obtained from the AA (2008), the mass of net UBC trade is obtained from the U.S. International Trade Commission (USITC 2010), the mass of UBC inventory change is obtained from Plunkert (2002), and the mass of cans collected is calculated based on the UBC recycling rate provided by the Container Recycling Institute (CRI) (Container Recycling Institute). Unlike the AA, the CRI corrects for the number of imported UBCS, as well as the net trade of unfilled cans. The annual number of UBCs consumed is estimated as mass of UBCs collected plus net UBC imports and adjusted for inventory changes. Note that due to use of inventories and trade, the mass of UBCs consumed does not necessarily equal the mass of UBCs collected. Apparent consumption of UBCs is calculated as the mass of UBCs collected plus net UBC imports plus net UBC inventory release.

Beverage can demand was relatively stable over the period, with annual shipments averaging approximately 1.4 million metric tons (AA 2008). The UBC recycling rate experienced greater variation and was in general decline over the period. However, it was during this period that the U.S. experienced its all-time high UBC recycling rate of 65% in 1992 (CRI 2010).

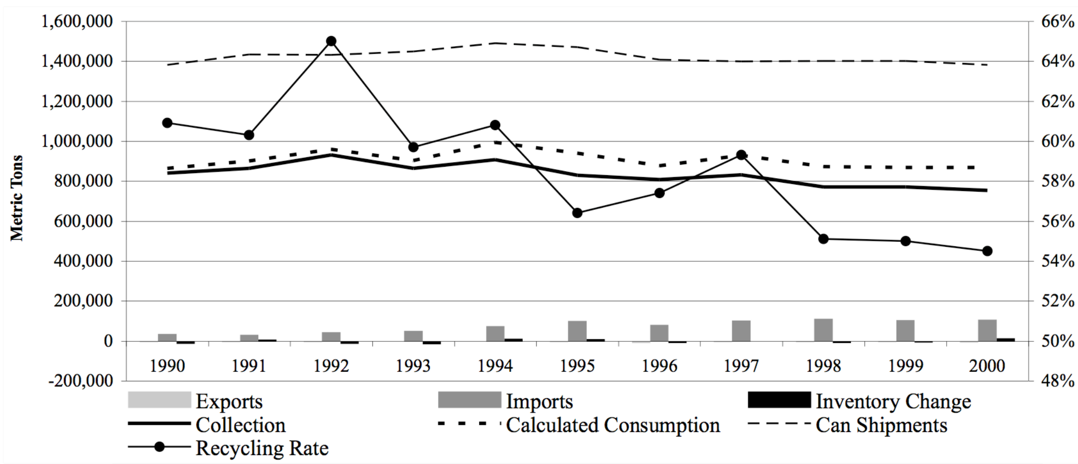


Figure 5.2 Aluminum Beverage Can Mass Flows and Recycling Rate (AA 2008; (Plunkert 2002); CRI 2010; USITC 2009)

5.4.1.1 Methodology

Sufficiently detailed data exist on aluminum beverage can manufacturing and UBC consumption and recovery to develop a system mass balance of aluminum flows and GHG emissions. Because our focus is on the impacts of choice of recycling allocation, we restrict our analysis to calculating the aluminum flows and emissions associated with the life cycle stages of primary aluminum production, UBC remelting, and secondary ingot production. Additionally, we calculate only GHG emissions due to our focus on the temporal inconsistencies of recycling allocation approaches. The emissions from the stages of sheet rolling, can manufacture, transport to consumer, and end-of-life disposal are equivalent using each allocation approach and are not included in the calculations as a result.

Table 5.2 presents the assumed loss factors used in the mass balance calculations. The model assumes that all can body and lid stock is produced from the UBC and can manufacturing stock consumed domestically. The case study focuses only on the mass of aluminum and does not include the mass of alloying elements. The 3104 body alloy and 5182 lid alloy contain roughly 96% and 94% aluminum, respectively. It is assumed that the body constitutes 75% of a can's mass (Hosford and Duncan 1994), resulting in an average aluminum content of 95%.

Table 5.2 U.S. Aluminum Beverage Can Mass Balance Parameters

Parameter	Value	Source
Sheet rolling loss	17%	BCS (2007)
Can manufacturing scrap rate	5%	Katok et al. (1999)
UBC melt loss	7%	Boin and Bertram (2005)
General melt loss	5%	Assumed

The model first estimates the annual total amount of aluminum required to produce the mass of aluminum cans shipped. The can manufacturing scrap rate is used to back out the required mass of can stock from the mass of aluminum cans shipped. The mass of required alloy is then calculated using the sheet rolling loss factor. Detailed model equations are included in Appendix D.

The recycled content approach calculates the mass of primary and secondary aluminum consumed annually to produce the reported mass of shipped beverage cans. The mass of secondary material consists of UBCs from domestic collection, net imports, and inventory adjustments, as well as new scrap generated during beverage can manufacture. The annual mass of primary aluminum consumed is assumed to equal the difference between the mass of beverage can sheet and the mass of UBCs and beverage can new scrap consumed, adjusted for melt losses.

Unlike the recycled content approach, which utilizes an estimate of the actual mass of aluminum consumed in a given year, the EOL recycling allocation approach requires an estimate of the annual amount of material lost from the system. In the case of aluminum beverage can, system losses are defined as uncollected UBCs and scrap melting losses. Any metal in the product life cycle that is not recovered for recycling is assumed to be made up by primary aluminum. The amount of secondary material is calculated as the difference between the required amount of ingot and the system losses.

As with the aluminum flows, GHG emissions are calculated on an annual basis. The source of primary aluminum is assumed to be North America and the annual greenhouse gas emission factor is obtained from McMillan and Keoleian (2009). The GHG emission factor for secondary aluminum ingot production is assumed to be 0.506 kg CO2-e/kg (European Aluminum Association 2008). Although the emission factor for secondary alloy production provided by IAI (2000) is more appropriate based on the time frame, the EAA emission factor was chosen based on its explicit accounting of the mass of aluminum metal contained in remelted scrap and mass of alloying elements in the cast ingot (6% of total mass input to refining stage; substituted by an equal mass of primary aluminum). The choice of secondary alloy emission factor affects model GHG estimates by no more than 1.6%; results using the IAI emission factor are provided in Appendix D.

5.4.1.2 Results

The EOL approach consistently results in larger estimates of required primary aluminum and total GHG emissions, as shown in table 5.3. The annual amount of primary aluminum required for the system is between 4% and 19% larger in the case of the EOL approach. In terms of GHG estimates, which also reflect the annual variation in GHG emission intensity of primary production, the EOL approach is between 4% and 18% larger over the period than the recycled content approach. The differences in the estimated masses of primary aluminum and GHGs are due to the EOLR assumption that primary input to the system simply equals the sum of aluminum lost from the system. This discrepancy is largely driven by the fact that due to trade and stockholding in the form of UBC inventories, the annual mass of UBCs consumed is not necessarily equal to the mass of UBCs collected. The fact that these two amounts are not equal is due to international UBC trade and inventory changes.

High and low GHG emission intensities for primary aluminum from McMillan and Keoleian (2009) were used for the purpose of additional comparisons. The percentage difference of emissions estimates between the two approaches using the high and low values were nearly the same as the mean value shown in table 5.3. It is concluded that uncertainty in the GHG emissions intensity of primary production may increase or decrease the absolute difference, but has little effect on the relative difference of total GHG emissions estimates of the U.S. aluminum beverage can system.

Table 5.3 Comparison of Material Flow and GHG Results by Allocation Approach

Year	Primary Aluminum (metric tons)			Total GHG Emissions (million metric tons CO_2-eq)		
	EOLR	RC	Difference (%)	EOLR	RC	Difference (%)
1990	595,000	573,000	3.7%	9.78	9.45	3.6%
1991	626,000	590,000	6.0%	9.76	9.23	5.7%
1992	565,000	537,000	5.3%	8.55	8.14	5.0%
1993	640,000	604,000	6.0%	9.02	8.53	5.7%
1994	648,000	565,000	14.8%	8.78	7.71	13.9%
1995	698,000	594,000	17.5%	9.49	8.15	16.5%
1996	653,000	588,000	11.1%	8.79	7.95	10.5%
1997	628,000	533,000	17.8%	8.33	7.14	16.7%
1998	681,000	584,000	16.5%	8.67	7.50	15.6%
1999	682,000	588,000	15.9%	7.93	6.90	15.0%
2000	680,000	570,000	19.3%	8.44	7.15	18.2%

The differences in model results are due to the way each approach accounts for trade and inventory adjustments of UBCs. These flows result in disparate estimates when the masses of UBCs consumed and recovered are not equal, which has become commonplace in the U.S. For instance, in 1999 approximately 770,000 metric tons of UBCs were recycled and approximately 868,000 metric tons of UBCs were consumed. During the same year net imports of UBCs were approximately 104,000 metric tons and 6,100 metric tons of UBCs were added to total inventory. So, the annual mass of uncollected

UBCs is not indicative of the mass of primary aluminum required for can production. Only when the amount of UBCs consumed equals the amount of UBCs recycled are the two allocation approaches equivalent.

A sensitivity analysis was utilized in order to further explore the effects of international trade and stockholding on GHG emissions estimates. The parameter developed for this exercise is defined as

$$\eta_t = \frac{\text{UBC Consumption}_t - \text{UBC Collection}_t}{\text{UBC Shipments}_t - \text{UBC Collection}_t} \tag{5.9}$$

This parameter effectively measures the percentage of uncollected UBCs that are replaced by international trade and stock changes, and is closely correlated to the percentage difference in primary aluminum flows calculated by each allocation approach. Positive values reflect consumption of UBCs above what is collected domestically, through net imports and/or releases from inventory; negative values reflect consumption below domestic consumption, through net exports and/or additions to inventory. From 1990 to 2000, replacement UBC consumption accounted for as much as 18% of uncollected UBCs. Figure 5.3 depicts the effects of varying this parameter, along with the UBC recycling rate, on the total GHG emissions estimated by allocation approach. Total GHG emissions are represented by the figure's contours. Emissions are shown as decreasing in the direction of the arrows. Emissions decrease moving from bottom to top and right to left for the RC approach (solid lines), and moving from top to bottom and right to left for the EOLR approach (dashed lines). These results are calculated using data for the year 2000.

Figure 5.3 shows that the two allocation approaches are equivalent when the same amount of UBCs is recycled domestically as are consumed. Also, the RC approach is shown to be much more sensitive to the relative mass of UBCs consumed. This occurs at all UBC recycling rates but is particularly prominent at low rates. It is interesting to note that unlike the RC approach the EOLR approach calculates an increase in GHG emissions as a greater fraction of uncollected UBCs are replaced via international trade (represented by an increase in ηt). With the year 2000 recycling rate of 54.5% a one percentage point increase in ηt results in an emissions reduction of 138,000 metric tons CO_2-eq using the RC approach, but an emissions increase of 17,700 metric tons CO_2-eq using the EOLR approach. This is an artifact of the EOLR approach calculating required primary aluminum based on system losses. In the EOLR approach additional consumption of UBCs beyond what is collected domestically results in additional metal losses from remelting UBCs, which must be made up for by additional primary production. Conversely, the RC approach treats additional UBC consumption beyond domestic collection as a reduction in the need for primary aluminum.

Both allocation approaches respond similarly to changes in the UBC recycling rate. At $\eta 2000 = 18.4\%$ a one percentage point increase in the UBC recycling rate reduces estimated GHG emissions by 112,000 metric tons CO_2-eq using the RC approach and by 141,000 metric tons CO_2-eq using the EOLR approach. At $\eta 2000 < 0\%$ (i.e., consuming fewer UBCs than are collected domestically, through net exports and/or stock additions), each one percentage point increase in the recycling rate results in a larger emissions reduction using the RC approach. At $\eta t = 0$ (i.e., there is no net trade or stockholding of UBCs) both allocation approaches react the same to changes in the recycling rate and at $\eta 2000 > 0\%$ (i.e., consuming more UBCs than are collected domestically, through net imports and/or stock drawdowns) an increase in the recycling rate results in a larger emissions reduction using the EOLR approach.

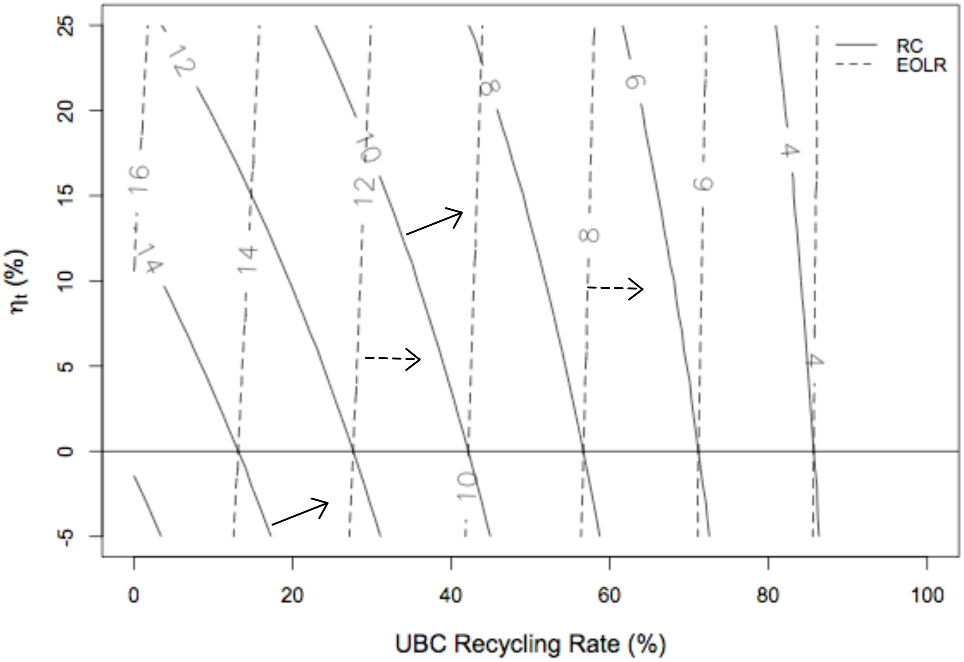


Figure 5.3 Sensitivity Analysis of GHG Emissions in the year 2000 by Allocation Procedure (Milion Metric Tons CO_2-eq)

5.4.2 Case Study 2: Aluminum Engine Block in a Hypothetical Automotive Fleet

5.4.2.1 System Overview

Whereas an aluminum beverage can has a useful life of less than a year, an aluminum engine block may be in use for over a decade until its retirement. Also, an aluminum engine block is manufactured from cast aluminum, which is predominantly composed of secondary aluminum. By evaluating a relatively long-lived product it becomes possible to further analyze how the RC and EOLR approaches treat the temporal nature of metal products and metal recycling.

The system boundary encompasses primary and secondary aluminum production, engine block fabrication, and end-of-life processes of collection, dismantling, and shredding. After a mass of scrapped engine block is shredded, it is no longer considered part of the system. Vehicle use phase is not included in the analysis. Unlike the beverage can example, which relied on reported supply and demand data, we assume a hypothetical scenario of a fleet of 100,000 vehicles manufactured in 1990. The calculations extend 30 years, to the year 2020.

5.4.2.2 Methodology

In this analysis it is assumed that the mass of the engine block is 18.1 kg, including alloying elements. Only aluminum and its associated emissions are included in the analysis. The engine block is comprised of three alloys: 319 (75% by mass), 356 (16%), and 380 (9%), which yield an aluminum mass of approximately 15.8 kg. Two vehicle lifetime distributions (Schmoyer 2001; Lu 2006) were used to estimate the annual engine block retirement.

Model parameters are provided in table 5.4. The model calculations for the EOLR approach include the mass of aluminum lost due to dross formation during engine block casting. The value is estimated using a mass-weighted average of aluminum loss during die casting using molten, ingot, and in-house remelt charges provided by Grotke (2004). For the RC approach, similar aluminum losses are assumed to be included in the existing data sets. Among the fundamental parameters required for a mass balance calculation of GHG emissions is the amount of primary and secondary metal consumed during production. Without direct measurements at the facility, this parameter must be assumed. Detailed model equations are provided in Appendix D.

Table 5.4 Cast Aluminum Engine Block Model Parameters

Life Cycle Stage	Parameter	Value	Source
Production	Recycling melt loss (δ_l)	0.05	EAA (2005)
Production	Fraction of secondary aluminum (χ)	0.85	Assumed
Production	Sand casting yield (θ_c)	0.65	Schifo and Radia (2004)
Production	Sand casting scrap rate (δ_c)	0.04	Schifo and Radia (2004)
End of Life	ELV recycling (θ_{ELV})	0.94	Staudinger and Keoleian (2001)

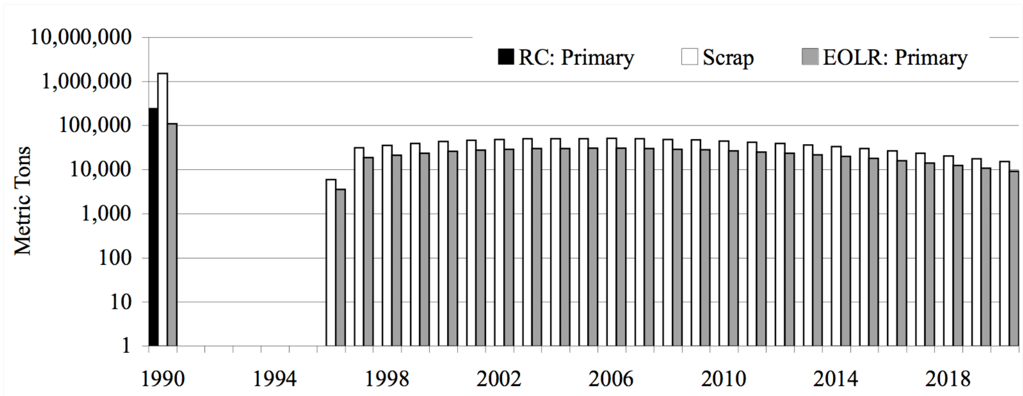
Life Cycle Stage	Parameter	Value	Source
End of Life	Shredding recovery (θ_s)	0.70	Das and Curlee (1999)
End of Life	Shredding & separation recovery (θ_n)	0.90	Das and Curlee (1999)

Like the in the beverage can case study, GHG emissions estimates are calculated using a dynamic approach and rely on North American primary production emission factors estimated by McMillan and Keoleian (2009) for 1990 – 2005. Primary production emissions beyond 2005 are assumed to equal the 2005 value. The secondary alloy production emission factor from EAA (2008) is used. The choice of emission factor affects model GHG estimates by no more than 2.8%; a detailed comparison is provided in Appendix D.

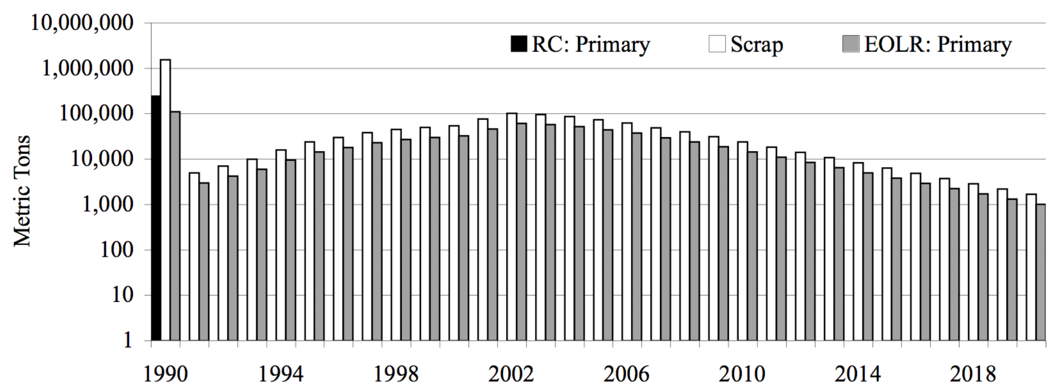
5.4.3 Results

The RC and EOLR approaches calculate equivalent flows of secondary aluminum, but they result in substantial differences in the magnitude and timing of primary aluminum flows. Based on the assumed distribution of vehicle scrappage, the first amount of engine block old scrap does not appear until either one year or six years after vehicle production. Peak scrap collection occurs roughly at the same time in either distribution, twelve to thirteen years after the vehicles are produced.

As depicted in figure 5.4 the RC approach calculates primary aluminum solely as the mass that is consumed in production in 1990. Conversely, the EOLR approach equates primary production to the mass of aluminum lost to the system and losses occur not only during the production of casting, but also during end-of-life management. All told, the RC approach calculates a primary aluminum requirement of 254,700 metric tons, while the EOLR approach calculates 663,000 metric tons using the Schmoyer distribution and 699,000 metric tons using the Lu distribution. Nearly 110,000 metric tons of primary aluminum are assumed to be used in the initial engine casting production phase by the EOLR approach and the remainder – approximately 84% – is the result of metal losses during the end-of-life management processes of collection, shredding, and nonferrous separation.



(a)



(b)

Figure 5.4 Temporal Distribution of Primary and Secondary Aluminum Mass Flows using Vehicle Retirement from (a) Schmoyer (2001) and from (b) Lu (2006)

The large discrepancies in the mass of primary aluminum utilized by each allocation approach are also evident in the mass of GHG emissions and their temporal distribution, as shown in figure 5.5. A 30-year time correction factor (TCF) (Kendall et al. 2009) for CO_2 was used to more fully capture the time-dependency of emissions. Some imprecision is associated with applying this CO_2-based TCF to total GHG emissions, but its use still serves the purpose of highlighting the difference in how the two allocation approaches treat emissions timing. Our results also reflect the uncertainty associated with the GHG emissions factors for primary production. Base case, high, and low emissions intensities (McMillan and Keoleian, 2009) are applied in each allocation approach.

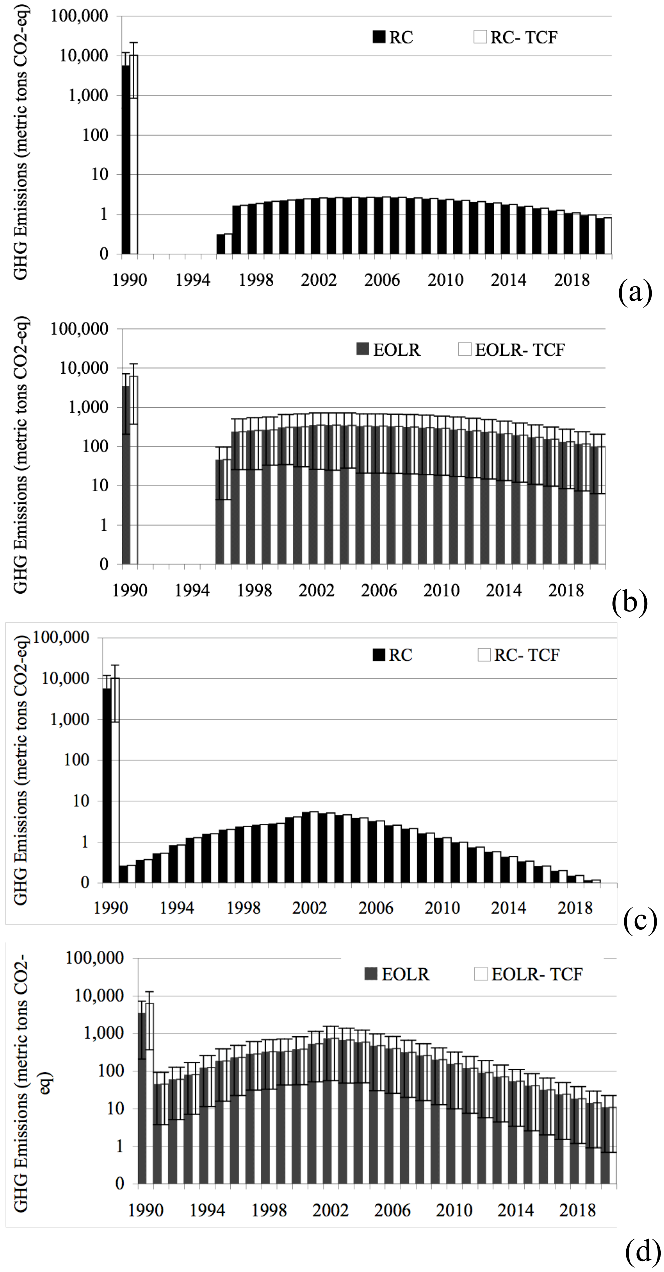


Figure 5.5 Temporal Distribution of GHG Emissions (Base GHG Intensity) Estimated by Recycled Content Approach ((a) Schmoyer (2001) vehicle retirement distribution and (c) Lu (2006) distribution) and by End-of-Life Recycling Approach ((b) Schmoyer (2001) vehicle retirement distribution and (d) Lu (2006) distribution)

Total GHG emissions calculated by each allocation approach are provided in table 5.5. The EOLR estimates are between 67% and 78% larger than the RC estimates with no TCF and between 20% and 26% larger with the TCF. Just as striking are the differences between the fraction of emissions that occur during initial production and during end-of-life management. More than 99% of the emissions occur during initial production under the RC approach, compared to between 35% and 49% for the EOLR approach.

Although the EOLR approach results in larger estimates of total GHGs, its temporal distribution of these emissions understates their global warming impact. The EOLR approach shifts the emissions burden from the initial production period to the future. This is the result of larger metal losses (i.e., primary aluminum according to the EOLR approach) occurring during end-of-life management than during initial production. By shifting the emissions associated with primary production to future years, the EOLR approach captures less of their cumulative warming effect.

The disparity of emissions timing also plays a role in the sensitivity and uncertainty of results estimated by each allocation approach. While the high and low GHG emission intensity of primary aluminum change the RC and EOLR estimates by roughly the same amount (7-9%), the EOLR approach is much more sensitive to the assumed vehicle retirement distribution. As can be deduced from table 5.5, assuming the distribution of Lu (2006) results in an emissions estimate that is less than 0.1% larger with the RC approach and approximately 7% larger with the EOLR approach. The EOLR approach produces emissions estimates with a higher degree of uncertainty due to its elevated sensitivity to vehicle retirement. Data on the uncertainty of the vehicle lifetime distributions would add greatly to this analysis, but, unfortunately, these are not provided by either source.

Table 5.5 Estimated Total GHG Emissions by Allocation Approach

Allocation Approach	Retirement Distribution	TCF?	GHG Emissions: Total (metric tons CO2-eq)	Fraction During Initial Production	Fraction During End-of-Life Management
RC	Schmoyer (2001)	No	5,780	99.1%	0.9%
		Yes	10,300	99.5%	0.5%
	Lu (2006)	No	5,780	99.1%	0.9%
		Yes	10,300	99.5%	0.5%
EOLR	Schmoyer (2001)	No	9,630	35.8%	64.2%
		Yes	12,300	50.1%	49.9%
	Lu (2006)	No	10,300	33.6%	66.4%
		Yes	13,000	47.5%	52.5%

Additional sensitivity analysis of each allocation approach is provided by observing the influence of recycled content. Results are presented as figure 5.6. While it is expected that an increased fraction of secondary aluminum results in lower total GHG emissions using the RC approach, it is somewhat surprising that additional secondary aluminum *increases* total emissions in the EOLR approach. For each one percentage point increase in secondary aluminum content, GHG emissions using the EOLR approach increase by 0.3% to 0.4% compared to a reduction of 1% under the RC approach. This counterintuitive result is based on the fact that the EOLR approach considers emissions only from recycling activities and primary production that replaces material lost to the system. An increase in the fraction of secondary aluminum increases not only the amount of scrap that must be remelted, but also the amount of metal lost (i.e., primary aluminum) during remelting.

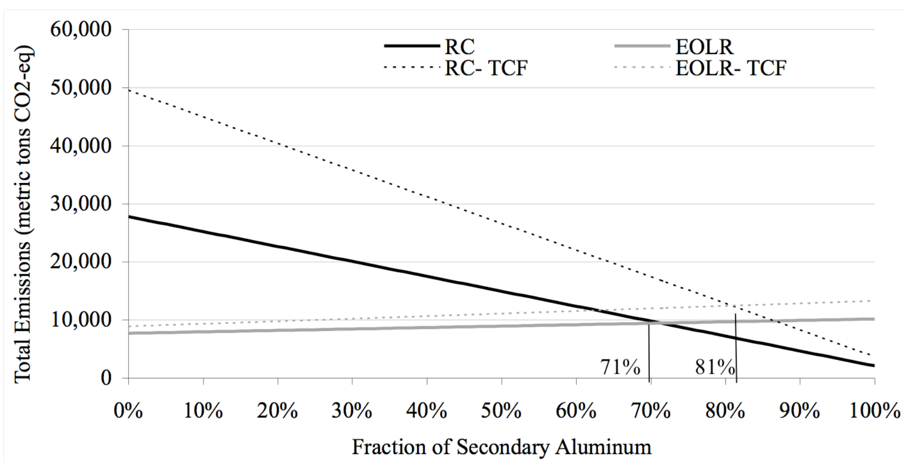


Figure 5.6 Effect of the Fraction of Secondary Aluminum on Total Estimated GHG Emissions by Allocation Approach (Schmoyer (2001) Vehicle Retirement Distribution)

Varying the total fraction of aluminum recovered at the end-of-life reveals that both allocation approaches estimate lower GHG emissions with increased recovery. In this analysis the end-of-life recovery represents the average of ELV collection, shredding, and nonferrous separation recovery; the GHG emission factor is an average of dismantling, shredding, and nonferrous separation. The timing of primary production and its associated emissions again creates a large discrepancy in the emissions estimated by each approach. For each one percentage point increase in the total end-of-life recovery, emissions estimated by the EOLR approach assuming either vehicle retirement distribution decrease 0.9% without a TCF and 0.8% with a TCF, as shown in figure 5.7. The effect on the RC approach is approximately zero.

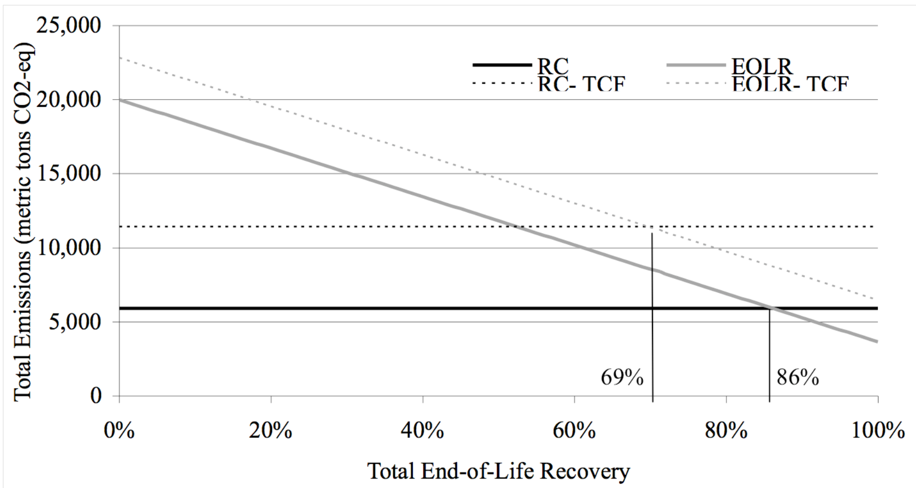


Figure 5.7 Effect of the Total End-of-Life Recovery of Aluminum on Total Estimated GHG Emissions by Allocation Approach (Schmoyer (2001) Vehicle Retirement Distribution

5.5 Discussion

The preceding analyses demonstrate that the MB and VCS approaches are not suitable choices for allocation in LCAs of aluminum product systems. The MB approach is found to lack consideration of the stockholding behavior that is a characteristic of storable commodities markets, and the price ratio of secondary aluminum alloy to primary aluminum is found to be nonstationary, violating a fundamental assumption of the VCS approach. Additionally, the EOLR approach, which has been specifically identified as the appropriate allocation approach for metals (Atherton 2007), is shown to insufficiently characterize international trade in the aluminum beverage can case study and to distort the timing of physical flows and emissions relative to the RC in the case of the automotive engine block.

The results of this work have far-reaching implications for materials other than aluminum, as well as for other allocation approaches. The suitability of the MB approach for evaluating aluminum recycling is called into question by the inconsistent results provided by the analysis of U.S. auto shredder scrap. A more fundamental shortcoming of the approach is its failure to account for stockholding behavior of storable commodities. The assumptions of the VCS approach rely on establishing the stationarity of price ratios, which the original proposal of the approach failed to sufficiently provide. Results of stationarity testing by Xiarchos (2006) indicate that the VCS might potentially be used for copper and lead, but it is not appropriate for zinc. The comparative analysis of the RC and EOLR approaches revealed the EOLR approach's inherent distortions to the timing of mass flows and emissions. By its assumption that primary production occurs only when material is lost from the total material pool, these distortions will occur with any metal.

The temporal nature of metal recycling poses problems for other proposed allocation approaches. For instance, the approach developed for automotive material substitution by Geyer (2008) is static and as such does not distinguish between when scrap is consumed during material production and when scrap is generated during vehicle manufacturing and end-of-life recycling. Acknowledging the timing of scrap consumption and generation may be problematic for the approach in its current form and would create separate values for its scrap balance parameters, siout and siin, at each point in time. These in turn would affect the calculation of the β parameter, which is used to estimate the change in secondary production in other life cycles due to changes in scrap balance.

Our analysis of the aluminum engine block demonstrates that the EOLR approach places between 48% and 66% of the environmental burdens of primary production in the future use phase, compared to less than 1% with the RC approach. This result is consistent with a larger issue relating to the RC and EOLR approaches discussed by Frischknecht (2010), which also focuses on aluminum. This work explores the value choices associated with the RC and EOLR approaches, and the distinct ways that the two approaches treat the temporal nature of metals recycling is described in terms of risk perception. The EOLR approach is classified as risk-seeking based on its assumption that future generations will demand recycled metal. This is considered risk-seeking behavior because the approach credits recycling activity even in light of the uncertainty associated with knowing recycling practices decades into the future. Conversely, the RC approach is classified as risk-averse as it does not make any assumptions about future recycling. This risk-averse behavior implicitly acknowledges the uncertainties associated with knowing future recycling practices and market characteristics by not assigning a recycling credit at the end of life.

The LCA community is largely polarized in its debate over allocation approaches. Attributional and consequential approaches both have merits and deficiencies; it is necessary that quantitative analysis continue to be used to understand their implications for different product systems. Before approaches are advocated for use with specific materials or processes, the suitability of their assumptions must be evaluated. Without objective, quantitative analyses the LCA community risks rendering LCA a "more incredible and disreputable" approach (Heijungs and Guinée 2007, 998).

Ignoring temporal aspects can lead to significant distortions of the flows of materials and emissions. Speculating on future behavior can introduce new layers of complexity and uncertainty. LCA practitioners desiring to utilize an allocation approach for aluminum product systems and other metals and long-lived products, would be best served by considering the implications of ignoring the temporal aspects of recycling systems and commodities markets, as well as the difficulties and uncertainties inherent with assuming future behavior.

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CHAPTER 6

CONCLUSIONS AND FUTURE WORK

6.1 Conclusions

The findings of this dissertation have demonstrated that an improved understanding of the industrial ecology of aluminum can be gained through incorporating temporal and dynamic aspects to LCA and MFA. The life cycle of aluminum is inherently dynamic and the drivers of production, consumption, retirement, recycling, and disposal are more robustly described by dynamic analyses than the traditional, static approaches. Observing and dynamically accounting for these drivers were shown to be critical in capturing the nature of aluminum flows and stocks, and their associated GHG emissions.

The research objectives identified in chapter 1.3 were addressed in four chapters of the dissertation. The following four sections detail the key observations and contributions of these research areas. Chapter 2 detailed a dynamic MFA model that was developed to estimate how changes in consumption affect in which end-use sectors of the U.S. economy aluminum accumulates. Chapter 3 developed a dynamic model of regional GHG emissions from primary aluminum production, consumption, and international trade. Dynamic analysis is used to present evidence that recycling aluminum has a limited ability to displace primary production outside of the UBC system in chapter 4. Chapter 5 detailed a quantitative analysis of four LCA allocation approaches applied to aluminum recycling.

6.1.1 U.S. Dynamic Aluminum MFA Model

The U.S. dynamic MFA model examined how aluminum consumption leads to the accumulation of in-use stocks and the recovery of old scrap in an economy over time. Because product lifetimes differ, a shift in consumption to a different product mix results in changes to the timing of future old scrap availability. The influence of assumed product lifetime was explored by utilizing different product retirement probability distributions.

A global sensitivity analysis was performed using the Fourier Amplitude Sensitivity Test (FAST). The sensitivity of LCA and MFA models are by and large tested using local results, which are unable to quantify linear and interactive effects. FAST results indicated that the product lifetime distribution and the recycling rate are each responsible for 40% of the total model variance. It is recommended that FAST be utilized for sensitivity analysis of future dynamic MFA models, particularly as modeling complexity increases. New analytical methods that can enhance the understanding of material flows, such as econometrics, will affect the sensitivity of MFA results. The FAST method offers a means of comprehensively evaluating these new contributions to model sensitivity.

U.S. in-use stocks have grown from 2,300 metric tons in 1900 to 93.6 million metric tons in 2007. Two periods of logistic growth occurred in the country’s in-use stocks: the first, from 1946 to 1987, was driven by consumption of wrought products in the construction, transportation, electrical, and machinery end-use sectors; the second, from 1987 to 2007, was driven by demand for castings in the automotive market.

As a result of the composition of in-use stocks and associated product lifetimes, recovered old scrap has mostly come from the containers and packaging, and transportation sectors. Aluminum beverage cans constitute the vast majority of containers and packaging category, and are characterized by a closed-loop recycling system where used beverage containers (UBCs) are recovered and remelted to produce new cans. Excluding the packaging and containers sector, the U.S. recovered on average 20% of the mass of aluminum from retired aluminum from 1972 to 2007; with the packaging and containers sector, the average recovery during the same period is 25%.

MFA model estimates of net additions to in-use stock (NAS) were used to examine the macroeconomic drivers of aluminum use in the U.S. First, time series econometrics was used to test for nonstationary data and GDP data are found to contain a unit root process. As a result, the data were log transformed and first differenced to remove the unit root. A quantitative model of a relationship between NAS and GDP was then developed using statistical regression. Model estimates indicated that each one percentage point increase in last year's $\Delta \ln$ NAS and current $\Delta \ln$ GDP is associated, *ceteris paribus*, with a change in current Δln NAS of −0.196 percentage points and 10.6 percentage points, respectively. Overall, the model results indicated that large, statistically significant changes in NAS are associated with changes in economic output as measured by GDP. These results were nearly the same using data measured on a per capita basis.

6.1.2 Dynamic Regional Primary Aluminum Life Cycle GHG Model

Just as the accumulation of aluminum in-use stocks and recovery of old scrap has changed over time in the U.S. economy, shifts in the location of primary production have occurred on a global level. After the oil shocks of the 1970s, the drive to secure low-cost sources of electricity have expanded primary production to regions far removed from the original markets of the U.S. and Western Europe. This spatial variation has created regionally distinct GHG emission factors of production, due to either hydro or coal being the typical low cost source of electricity. Additionally, each region has experienced unique trends in electricity intensity of production (i.e., the electricity required to produce a mass of ingot), intensity of perfluorocarbon emissions, and carbon intensity of coal and natural gas fired electricity generation. All told, there is pronounced regional and temporal variation in the GHG intensity of primary aluminum ingot production. This variation is not captured in most publicly-available life cycle inventory reports, where a new global average value is estimated approximately every five years.

The dynamic primary aluminum GHG model presented in this dissertation estimated that regional GHG intensities in 2005 ranged from 7.07 kg CO_2-eq/kg ingot in Latin America to 21.9 kg CO_2-eq/kg ingot in Asia. This discrepancy was explained by the fact that smelters in Latin America are powered by hydro and the smelters in Asia are inefficient, highly PFC emissions intensive, and powered by coal-fired electricity generation. Asia also had the highest absolute emissions in 2005: approximately 244 million metric tons CO_2-eq. This represents an increase of 450% over 1990 emissions and is 230% greater than the second largest emitter, Europe.

The most substantial change in GHG intensity over time occurred in Africa. Between 1990 and 2005 the average GHG intensity increased 46% from 12.3 kg CO_2-eq/kg to 18.0 kg CO_2-eq/kg. This was the result of a shift in the smelter electricity fuel mix from hydro to coal. The region experienced a near doubling of production during the period and many of the new smelters are based on coal-fired sources of electricity.

The expansion of primary smelters into new regions has coincided with growing demand for primary aluminum in the developing world. New trade patterns have emerged with implications for the GHGs associated with the consumption of primary ingot for a given region. Estimating absolute and relative emissions on a consumption basis reflects the GHGs embodied in net trade. In 2005 North America and Asia imported roughly 25% of their apparent consumption of primary ingot. Because the emission intensities of North American production and imports were similar, the consumption and production emissions intensities did not differ greatly. Conversely, Asian imports were 36% less GHG intensive than domestic production and the consumption weighted GHG intensity of the region was 10% lower than the domestic production intensity.

LCA studies of aluminum products should consider the influence of time frame, trade and other regional factors before selecting a life cycle GHG emissions intensity. Selecting the most representative emission factor is critical not only for the results of the LCA itself, but also for any associated policy recommendations. Recent discussions in the U.S. and European Union of levying carbon taxes on imported goods have highlighted the importance of measuring GHG intensity on regional and consumption bases. Carbon tariffs have been debated as a means of reducing carbon leakage (i.e., the relocation of carbon intensive industries from a region with GHG caps to a region without) by raising the price of goods imported from countries that do not regulate GHG emissions. The regional GHG intensities of primary aluminum production would be a crucial component of trade policy analysis for evaluating how producers and consumers would be affected under a carbon tariff.

6.1.3 Evaluation of Metals Recycling Characteristics

It is the position of the metals industry that metals are recycled at high rates and recycled over and over again. Additionally, secondary production displaces primary production and its associated emissions, and is constrained by scrap availability. These positions were shown to be simplifications or misrepresentations of the current recycling systems for aluminum, copper, and iron and steel. The position of the metals industry should be viewed as theoretical potential to work towards and not as reflecting current practice.

The position that metals are recycled at high rates was examined for aluminum and copper using existing MFA research. The majority of these MFA models have estimated that less than 50% of copper and aluminum and nearly 75% of iron and steel products that are retired are recovered for recycling in Europe and the U.S. The metals industry identified metals as being different from other materials like plastics and paper due to their mature and economically-viable recycling markets. Estimates placed the recycling rates of paper on par or above what has been estimated for these three metals.

The recycling of metals over and over again was shown to be contrary to the results of work that utilized Markov chain analysis to estimate the number of times metal is recycled. These studies have estimated that copper is recycling 1.9 times on a global basis (Eckelman and Daigo 2008) and iron and steel are recycled 2.7 (Matsuno, Daigo, and Adachi 2007) before ultimate disposal. Although a similar estimate is unavailable for aluminum, the analysis provided in Chapter 4 of this dissertation suggested that based on recycling rates and product lifetimes the number of recycling events would be similar to that of copper. These results indicate that while metals have the theoretical potential to be recycled an infinite number of times, current recycling systems fall far short.

The position that the constraint to metals recycling is the availability of scrap is a simplification of the multitude of factors that influence metals recycling. Unless mandated by law, the extent of product recycling is determined by economic considerations. These considerations include the quality and concentration of metal, and the demand for products that are capable of utilizing scrap. Contamination plays a role in the recycling of all metals, but varies in its degree of influence based on the metal and its recycling infrastructure. The strict tolerances of certain aluminum alloys have created two separate markets for wrought and cast products. Old scrap is largely consumed for automotive castings, while wrought products rely on mostly primary aluminum and new scrap. Copper was found to be similar to aluminum, with the use of unrefined copper alloy old scrap being limited to castings. Most iron and steel products are able to utilize old scrap; nonetheless, steel produced with the basic oxygen furnace process is limited to a scrap input of 30% (Fenton 2004).

The possibility that secondary metal production offsets primary production was investigated using the U.S. aluminum market as a case study. A significant growth in non-UBC old scrap consumption was identified from 1986 to 1993. This corresponded to a surge in ingot shipments to the automotive market, which is consistent with the prevailing knowledge that most old scrap is utilized for castings. After 1993 the non-UBC old scrap consumption decreased, while the consumption of automotive castings continued unabated. It was found that during this time an increasing fraction of aluminum was recovered from dross and that secondary ingot producers utilized this increased recovery. It is also likely that additional imports of secondary alloy and castings helped to cover the gap between old scrap recovery and the requirements for automotive castings shipments.

These trends in non-UBC old scrap consumption and ingot production were then compared against the trends in primary aluminum consumption and shipments of semi-finished wrought products (e.g., rolled sheet, extruded wire). This basic comparison revealed that the consumption of primary aluminum, new scrap, and UBC scrap was closely correlated to shipments of wrought semi-finished products, while the consumption of non-UBC old scrap was closely correlated to shipments of ingots for casting. These relationships were expected based on the separation of the two markets and little evidence was found to support the idea that secondary production displaces primary production in the U.S. aluminum market.

6.1.4 Quantitative Analysis of LCA Allocation Approaches for Aluminum

The debate surrounding open-loop recycling allocation in LCA has become polarized. One position supports attributional allocation, which ascribes the functional flows of a system based on mass, market price, energy content, or other physical or economic characteristic. The other position supports consequential allocation, which expands the definition of attributional LCA by including the possible direct and indirect effects associated with changes in the life cycle of a system brought about by a decision (Curran, et al. 2002; Weidema 2000). Numerous attributional and consequential approaches have been proposed, yet many lack quantitative supporting analysis. In particular, the approaches that have been proposed for aluminum and other metals have not addressed their ability to accurately account for the timing of physical flows and emissions.

The market-based (MB), value-corrected substitution (VCS), end-of-life-recycling (EOLR), and recycled content (RC) approaches were quantitatively evaluated for their application to aluminum recycling. The MB approach is based on the assumption that the price elasticities of scrap supply and demand can be used to estimate the impacts of recycling in other product systems. This assumption was found to be inconsistent with the stockholding behavior of storable commodities markets and was deemed inappropriate for application to aluminum. Suppliers and consumers hold stocks of commodities like aluminum based on expected changes in price and to guard against stochastic shocks in market conditions. Because metal supply is inelastic in the short run, stocks play a critical role in equilibrating supply and demand; this behavior is also not captured by the MB approach.

The VCS approach was developed specifically to account for the degradation in quality encountered during aluminum recycling. The approach calculates the environmental burdens associated with primary aluminum production based on the difference between the price ratio of secondary alloy to primary aluminum and the price ratio of old scrap to primary aluminum. The fundamental assumption of the approach is that these price ratios are constant over time. It was found that the original statistical analysis by Werner (2005) was insufficient to support this conclusion. As a result, time series analysis and a battery of unit root and stationarity tests were used to establish that the price ratio of secondary alloy to primary aluminum was nonstationary and the price ratio of old scrap to primary aluminum was stationary. The VCS approach was deemed unsuitable for application to aluminum.

The EOLR approach is supported as a method of capturing the theoretical ability of metals to be recycled without degradation of their inherent atomic properties. The metal under study is treated as being a part of a large material pool and primary production only occurs when metal is not recovered for recycling or is lost during melting. The EOLR approach was quantitatively compared to the RC approach, which accounts for the functional flows of primary and secondary metals based on the mass consumed during product production, using two case studies of aluminum products. The first, the U.S.

aluminum beverage can market, found that the EOLR approach was unable to accurately account for the international trade of used beverage containers (UBCs). Instead of representing a decrease in the need for primary production as in the RC approach, the EOLR approach estimated increased primary aluminum production to cover the losses that occur from remelting the imported UBCs. The EOLR approach estimated total GHGs nearly 18% above the RC approach.

A hypothetical fleet of aluminum engine blocks comprised the second case study. It was found that the losses encountered during end-of-life recovery processes (e.g., dismantling, shredding, non-ferrous separation) cause the EOLR approach to report primary production and associated GHG emissions much later in the product life than the RC approach. The RC approach estimates that 99% of life cycle GHGs of the fleet of aluminum engine blocks occur during production; the EOLR approach estimates that this figure is between 34% and 52%. The timing of emissions has additional implications for GHGs; by shifting primary production into the future, the EOLR approach reduces the warming potential of GHG emissions relative to the timing of the RC approach. Using a time correction factor to account for the timing of CO_2 increases the total GHG emissions estimated by the RC approach by 78%. The same time correction factor increases total GHG emissions of the EOLR approach by 29%. All told, the EOLR approach estimates total GHG emissions from the hypothetical fleet of engine blocks that are between 18% and 77% larger than estimated by the RC approach.

The EOLR approach has significant implications for climate change policy. The effect of shifting primary production GHG emissions to the future not only undercounts the total global warming effect, but it transfers the burdens of emissions reductions to subsequent generations. This exacerbates the intergenerational inequity of climate change: the current generation receives the benefit of using the primary material, but a future generation is assigned the emissions of production.

Distorting the timing of current emissions so that they occur in the future also reduces the urgency of GHG mitigation. Delaying action on emissions explicitly clashes with the climate stabilization paths identified by the IPCC. Stabilizing the atmospheric CO_2 concentration between 350 and 400 ppm, levels that limit global average temperature increase to 2.4°C, requires that emissions peak between 2000 and 2015 (Metz et al. 2007).

Although the RC approach correctly accounts for the timing of material flows and emissions, it is not without its shortcomings. Most notably, it does not acknowledge the environmental benefits of scrap that is generated during the life cycle of a product system. Two product systems, each identical with the exception that one generates usable scrap at the end-of-life and the other does not, are treated the same under the RC approach. In this way the RC approach creates an incentive to consume scrap, instead of an incentive to generate scrap at end-of-life. However, economically speaking, the consumption of scrap is derived from the demand for products that are able to utilize scrap. This view is supported by the work of Blomberg and Hellmer (2000) for the Western European aluminum market; it was estimated that for each 1% increase in automotive production, old scrap demand increased by 0.52%.

Utilizing the RC allocation approach and therefore promoting the use of scrap as a production input could lead to environmental benefits by encouraging increased recovery of old scrap. The increased demand may raise the short-run price of scrap, which would enter into the stockholding decisions of scrap producers and consumers. The price rise may then encourage increased old scrap recovery in the long run. Utilizing the RC approach does not preclude the use of consequential LCA and the effects of these market changes could be explored using the appropriate dynamic economic models and through systems expansion.

Overall, the ability of the RC approach to accurately account for the temporal nature of aluminum material flows and GHG emissions, and to be used in a consequential LCA framework make it the recommended allocation approach. Where appropriate, the RC approach should be extended with systems expansion approaches that are based on sound economic methods and theory.

6.2 Future Work

Although the research contributions developed in this dissertation were extensively focused on aluminum, the methods are appropriate for other metals and materials. In particular, other metals are in characterized by the same basic intertemporal bevhavior and share many of the same dynamic processes that are part of the aluminum life cycle. All metals are exhaustible resources and are brought to market as storable commodities. Their consumption over time is driven by trends in economic activity, have long lives, and accumulate in an economy as in-use stocks.

The same time series analysis methods that were used to quantify the relationship between GDP and aluminum NAS could be applied to existing MFA data for additional metals in the U.S. and other regions. Results could then be compared between metals to highlight and evaluate how economic output affects the stock accumulation of different metals. Further refinements to these econometric models, including the model for aluminum presented in this dissertation, could be made by introducing additional explanatory variables and by testing for variable cointegration. Additional variables could include the disaggregated components of GDP – final consumption, investment, government purchases, and net exports.

Conintegration testing, Granger causality testing (Granger 1969), vector autoregression, and other time series analysis methods would also be applied in the development of an econometric model of the U.S. secondary aluminum market. The most recent econometric models of the scrap aluminum market (Blomberg and Söderholm 2009; Blomberg and Hellmer 2000) neither acknowledge nor utilize time series econometric methods; applying such methods would allow a much more appropriate and robust analysis. Recent work that examines the relationships between the prices and stocks of metals (Xiarchos and Fletcher 2009; Xiarchos 2006) and develops a structural-time series analysis model for the U.S. wheat market (Robledo 2002) serve as example analyses.

This work would be a significant contribution to the literature on scrap metal markets, but would also be useful in the LCA allocation debate. The secondary aluminum econometric model would be used to estimate the cross price elasticities of primary and secondary aluminum. These relationships have yet to be quantified and are essential information in the debate over whether or not the production of secondary aluminum displaces primary production. The argument that secondary production displaces primary production would be supported if the two were found to be substitutes. This would involve finding that a rise in primary price is associated with an increase in demand for secondary alloy. Cointegration and Granger causality testing would also be used to further analyze the relationship between primary and secondary alloy prices.

The LCA community would greatly benefit from the development of a more appropriate recycling allocation approach. The lack of an approach that accurately accounts for the timing of emissions, material production and consumption, while recognizing the benefits of providing a source of secondary material, hinders the utility of LCA for policy development. The research efforts of this dissertation have concluded that while the RC approach accurately accounts for the timing of metals recycling and emissions, it lacks a means of designating the benefits associated with providing secondary material for future consumption. Ideally, an allocation approach would assign benefits to a material that is recycled at its end-of-life without speculating how it is then used (e.g., if it displaces primary production).

An appropriate allocation approach would account for the differences in the old scrap availability and emissions of short-lived and long-lived products. A potential avenue to consider would be to incorporate a discount rate to equilibrate the costs and benefits of producing a material today with the costs and benefits of making scrap available for consumption in the future. The uncertainty of ascribing emissions to scrap generation and consumption decades into the future, when recycling technology and metal consumption are unknown, would expressed by the choice of discount rate. Although, this approach is likely to be complicated by the range of discount rates that would be associated with different emissions and other impacts to human and ecosystem health.

6.3 References

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APPENDIX A

SUPPLEMENTAL MATERIAL: CHAPTER 2

A.1 Indirect Aluminum Flows

A brief analysis was performed for aluminum to gauge the extent to which the model may over- or underestimate additions to in-use stock. Data used for the analysis were available from 1989 to 2007 and are presented in Table A.1. Results of the analysis indicate that by not including these data the model would underestimate the amount of aluminum net imports by as much as 1,383,000 metric tons (61% of aluminum net imports) in 1994 and as little as 574,000 tons (78% of aluminum net imports) in 1991, with an annual average of 841,000 tons (48% of aluminum net imports). This annual average underestimate is equivalent to approximately 13% of apparent consumption. The imports for consumption and domestic exports from 1989 to 2006 were obtained from the U.S. International Trade Commission (USITC, 2009) for the trade categories metal doors, windows, door thresholds and window frames of aluminum (SITC 69121); stranded ropes, wires, and cables, etc. of aluminum (SITC 69313); household articles and parts thereof, n.e.s., of aluminum (SITC 69743); and motor cars and other motor vehicles (SITC 781). These trade categories were chosen as the basis for evaluating the mass of traded aluminum not accounted for by the model. Since the trade flows of motor cars and other motor vehicles are reported in number of units, the total mass of aluminum was calculated from annual estimates of the average mass of aluminum per vehicle (Burgert, 2007).

Table A.1 Indirect Aluminum Flows Added to Apparent Consumption (metric ton, unless noted otherwise) (USITC, 2009)

	SITC - 69121		SITC - 69313		SITC - 69743	
Year	Imports for Consumption	Domestic Exports	Imports for Consumption	Imports for Consumption	Domestic Exports	Domestic Exports
1989	8,881	6,425	11,261	3,122	805	5,971
1990	7,608	6,618	3,114	2,303	1,732	7,268
1991	3,345	6,590	5,412	2,736	2,038	11,310
1992	2,940	8,290	7,016	2,906	2,245	3,395
1993	1,911	9,162	6,844	3,243	1,938	2,111
1994	2,540	8,445	13,811	5,254	2,539	2,967
1995	3,484	8,516	9,484	5,649	1,649	5,170
1996	8,357	9,932	13,985	3,343	2,896	2,549
1997	11,063	11,791	19,582	4,132	1,697	6,211
1998	14,972	12,560	31,241	5,080	1,944	3,921
1999	20,292	11,182	31,295	5,370	2,132	2,395
2000	34,338	9,890	30,802	7,218	3,536	2,925
2001	37,418	9,574	34,959	7,149	1,965	3,303
2002	40,088	8,197	34,282	10,942	1,391	4,813

2003	39,717	11,046	43,458	13,194	2,213	6,578
2004	51,219	10,798	59,351	19,683	2,484	3,739
2005	56,848	15,274	79,048	19,031	3,283	3,103
2006	66,147	12,665	94,413	12,203	5,094	2,071
2007	81,629	10,978	119,280	12,764	4,476	3,437

SITC – 781 (in Units)	
Imports for Consumption	Domestic Exports
9,051	897
9,159	899
8,333	950
9,005	1,112
10,995	1,111
16,239	1,234
10,192	1,246
9,290	1,256
7,569	1,326
7,250	1,231
8,023	1,195
7,651	1,251
7,243	1,306
7,750	1,397
7,568	1,377
8,098	1,409
8,527	1,656
9,451	1,920
9,223	2,259

A.2 Dynamic MFA Model

The cumulative in-use stock for end-use sector i in year t, $S_{i,t}$ is calculated as

$$S_{i,t} = \sum_{t=1900}^{2007} \sum_{s=0}^{t-1900} \left(C_{i,t-s}(1-\delta_{i,s}) + S_{i,t-1} \right) \quad \text{(A.1)}$$

where s is the number of years after initial consumption, $C_{i,t-s}$ is the aluminum apparent consumption by end-use sector i in year t-s and $\delta_{i,s}$ is the distribution parameter for average product retirement in end-use sector i.

Once product enters the waste stream, it is either collected for recycling or released into the environment. In the model the annual recycling rate chosen for each end-use category should ideally reflect the observed fraction of aluminum products recovered from waste streams in that year. Unfortunately, with the exception of 1972-2007 for the containers and packaging category (Aluminum Association, 2008), these annual data are not available. Instead a separate recycling rate for each category obtained from Bruggink (2000) is applied as a constant across all years.

Loss of aluminum metal also occurs after products have been collected for recycling. Remelting aluminum scrap results in the loss of metal and produces dross, a mixture of aluminum and contaminants, which may or may not be processed to partially recover its aluminum content. We refer to the fraction of aluminum metal remaining after remelting as the metallic recovery rate.

The ultimate recovery of old scrap from end-use sector *i* in year *t*, $R_{i,t}$, is calculated as

$$R_{i,t} = \sum_{t=1900}^{2007} \sum_{s=0}^{t-1900} \left(C_{i,t-s} \times \delta_{i,s} \times \eta_i \times \varepsilon_i \right) \tag{A.2}$$

where η_i is the average recycling percentage of end-use sector *i* and ε_i is the average metallic recovery of recycled aluminum from end-use sector *i*.

A.3 Product Lifetime Distribution Assumptions

The dynamic MFA model relies on product lifetime probability distribution functions developed by Melo (1999), shown in Table A.2, and by Müller et al. (2006), for uncertainty analysis. Because Müller et al. developed product lifetimes based only on an average distribution, as shown in Table A.3, estimates were constructed for Weibull distributions and are shown in Table A.4.

Table A.2 Product Lifetime Distribution Data (Melo, 1999)

Distribution and Product Category	Lifetime Interval	Standard Deviation	Mean Lifetime	Most Likely Lifetime
Normal Distribution				
Transportation	[10,16]	1.0	13	na
Machinery & Equipment	[10,20]	1.7	15	na
Electrical	[10,25]	2.5	17.5	na
Construction	[23,40]	2.8	31.5	na
Packaging & Containers	1	0.2	na	na
Consumer Durables	[5,15]	1.7	10	na
Other	[5,15]	1.7	10	na

Distribution and Product Category	Lifetime Interval	Standard Deviation	Mean Lifetime	Most Likely Lifetime
Weibull Distribution				
Transportation	[10,16]	na	12.2	11.8
Machinery & Equipment	[10,20]	na	13.6	12.9
Electrical	[10,25]	na	15.5	14.4
Construction	[23,40]	na	29.3	28
Packaging & Containers	1	na		
Consumer Durables	[5,15]	na	8.6	7.9
Other	[5,15]	na	8.6	7.9
Beta Distribution				
Transportation	[10,16]	na	12	11.2
Machinery & Equipment	[10,20]	na	13	12
Electrical	[10,25]	na	15	14
Construction	[23,40]	na	30	30
Packaging & Containers	1	na	1	1
Consumer Durables	[5,15]	na	8	7
Other	[5,15]	na	8	7

Table A.3 Product Lifetime Distribution Data (Müller et al, 2006)

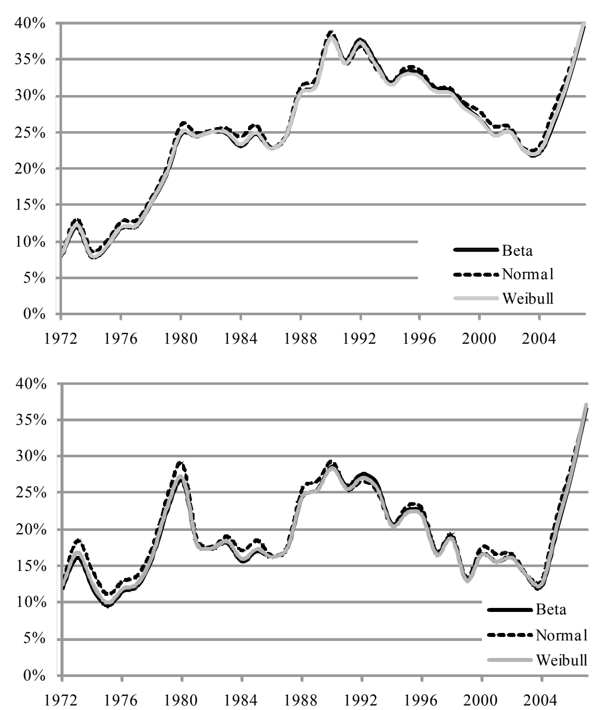
Distribution and Product Category	Standard Deviation	Mean Lifetime
Normal Distribution		
Transportation	7.5	20
Machinery & Equipment	10	30
Electrical	na	na
Construction	20	75
Packaging & Containers	na	na
Consumer Durables	na	na
Other	5	15

Table A.4 Product Lifetime Distribution Data

Distribution and Product Category	Lifetime Interval	Mean Lifetime	Most Likely Lifetime
Weibull Distribution			
Transportation	[10,30]	20	19
Machinery & Equipment	[10,50]	30	28
Electrical	[10,20]	15	16
Construction	[23,127]	75	73
Packaging & Containers	1		
Consumer Durables	[5,15]	10	11
Other	[5,25]	15	14

A.4 Estimating Aluminum Recycling Rate and Domestic Recycling

By correcting the annual domestic old scrap recycling and total old scrap generation for the mass of UBCs recovered and generated (Aluminum Association, 2008), an estimate is generated of the old scrap recycling rate as driven by sectors outside of beverage containers. The effect of removing UBCs is not uniform across years, which can be seen in Figure A.1. During the 1990s when beverage can recycling was at historical highs, removing UBCs lowers the total recycling rate by an average of 11 percentage points. The difference between recycling rates becomes much less prominent after the turn of the century, particularly after 2004 when the overall amount of old scrap recycled increased. This was likely a response to the run-up of aluminum prices that occurred during the same period. The increase in prices even provided incentive for illegal recovery of material that had not reached the end of its useful life (Maag, C., 2008).



(a) (b)

Figure A.1 (a) Estimated Rate of Aluminum Old Scrap Recycling by End of Life Distribution and (b) Adjusted for UBCs.

Table A.5 presents the data and calculations used to estimate the annual amount of old scrap that is domestically recycled and consumed in the U.S.

Table A.5 Calculation of Estimated Domestic Old Scrap Recycling and Consumption (metric tons).

	A	B	C	D
	Secondary Smelter Old Scrap Consumption	Primary Producer, Foundries, Ind. Mill Fab, etc. Old Scrap Consumption	Old Scrap Imports	Old Scrap Exports
Source	USGS (2009)	USGS (2009)	USGS (2009)	USGS (2009)
1946	83,428	47,059	13,148	581
1947	162,088	32,441	14,260	715
1948	87,605	20,203	65,075	397
1949	39,179	8,559	36,397	360
1950	69,923	13,628	61,652	726
1951	70,264	10,513	18,152	1,325
1952	70,344	4,266	6,350	907
1953	77,112	5,997	24,494	4,536
1954	66,703	4,899	13,608	35,381
1955	91,334	4,474	37,195	16,329
1956	90,342	2,420	23,587	17,237
1957	82,126	2,078	14,515	16,329
1958	80,979	1,897	9,001	17,151
1959	99,702	1,884	9,906	29,382
1960	80,204	1,933	5,118	72,133
1961	61,760	956	5,445	74,394
1962	76,002	2,910	5,893	59,451
1963	81,199	3,158	8,442	64,446
	A	B	C	D
	Secondary Smelter Old Scrap Consumption	Primary Producer, Foundries, Ind. Mill Fab, etc. Old Scrap Consumption	Old Scrap Imports	Old Scrap Exports
Source	USGS (2009)	USGS (2009)	USGS (2009)	USGS (2009)
1964	91,256	3,751	7,395	62,246
1965	111,924	6,489	24,520	34,969
1966	96,798	3,205	30,496	44,149
1967	94,141	3,646	27,659	49,470
1968	103,335	7,685	34,038	44,839
1969	107,400	8,511	26,172	78,221
1970	113,985	9,506	33,365	51,854
1971	107,412	17,716	57,007	27,828
1972	106,813	30,924	47,447	59,910
1973	101,424	58,605	42,464	104,435
1974	113,345	64,308	67,806	72,719

	A	B	C	D
	Secondary Smelter Old Scrap Consumption	Primary Producer, Foundries, Ind. Mill Fab, etc. Old Scrap Consumption	Old Scrap Imports	Old Scrap Exports
Source	USGS (2009)	USGS (2009)	USGS (2009)	USGS (2009)
1975	112,632	111,277	49,719	59,645
1976	144,124	138,229	77,758	98,845
1977	167,442	178,829	81,551	92,227
1978	187,387	201,597	83,600	176,455
1979	221,420	215,152	61,975	278,578
1980	260,423	261,759	54,251	403,408
1981	335,767	378,130	74,384	218,778
1982	285,337	487,731	67,438	194,409
1983	280,282	533,221	80,260	215,753
1984	272,891	551,471	137,675	258,404
1985	344,339	466,551	127,501	374,646
1986	268,524	503,608	162,317	350,858
1987	267,557	581,105	188,657	368,510
1988	419,771	653,146	200,517	486,615
1989	399,494	665,938	206,610	575,419
1990	588,493	862,389	214,196	537,312
1991	411,890	1,002,943	208,384	460,820
1992	589,885	1,166,916	265,306	295,239
1993	670,352	1,099,676	309,000	212,000
1994	443,363	1,195,659	390,000	307,000
1995	496,700	1,144,910	419,000	430,000
1996	542,000	1,170,000	402,000	320,000
1997	549,000	1,120,000	454,000	338,000
1998	724,000	910,000	501,000	428,000
1999	643,000	1,050,000	615,000	419,000
2000	582,000	918,000	625,000	625,000
	A	B	C	D
	Secondary Smelter Old Scrap Consumption	Primary Producer, Foundries, Ind. Mill Fab, etc. Old Scrap Consumption	Old Scrap Imports	Old Scrap Exports
Source	USGS (2009)	USGS (2009)	USGS (2009)	USGS (2009)
2001	510,000	823,000	497,000	580,000
2002	506,000	785,000	466,000	613,000
2003	429,000	759,000	440,000	577,000
2004	470,000	778,000	535,000	660,000
2005	399,000	755,000	482,000	1,090,000
2006	498,000	796,000	527,000	1,480,000
2007	488,000	1240000	471,000	1,550,000

Source	E Domestic Old Scrap Recycling A+B-C+D	F Domestic Old Scrap Consumption A+B-C	G Percentage of Domestic Old Scrap Consumed Domestically F/E
1946	117,920	117,339	100%
1947	180,984	180,269	100%
1948	43,131	42,733	99%
1949	11,702	11,342	97%
1950	22,625	21,899	97%
1951	63,949	62,625	98%
1952	69,166	68,259	99%
1953	63,152	58,616	93%
1954	93,375	57,994	62%
1955	74,942	58,613	78%
1956	86,412	69,175	80%
1957	86,018	69,688	81%
1958	91,026	73,875	81%
1959	121,063	91,681	76%
1960	149,152	77,019	52%
1961	131,665	57,271	43%
1962	132,471	73,019	55%
1963	140,361	75,915	54%
1964	149,858	87,611	58%
1965	128,862	93,893	73%
1966	113,656	69,507	61%
1967	119,598	70,128	59%
1968	121,820	76,981	63%
1969	167,960	89,739	53%
1970	141,980	90,126	63%
1971	95,948	68,121	71%
1972	150,200	90,290	60%
1973	222,001	117,566	53%
1974	182,566	109,847	60%
1975	233,835	174,190	74%
1976	303,439	204,594	67%
1977	356,946	264,719	74%
1978	481,839	305,385	63%
1979	653,175	374,597	57%
1980	871,338	467,930	54%
1981	858,291	639,514	75%
1982	900,038	705,629	78%
1983	948,996	733,243	77%

	E	F	G
	Domestic Old Scrap Recycling	Domestic Old Scrap Consumption	Percentage of Domestic Old Scrap Consumed Domestically
Source	A+B-C+D	A+B-C	F/E
1984	945,091	686,687	73%
1985	1,058,035	683,389	65%
1986	960,673	609,815	63%
1987	1,028,515	660,005	64%
1988	1,359,015	872,400	64%
1989	1,434,241	858,822	60%
1990	1,773,998	1,236,686	70%
1991	1,667,269	1,206,449	72%
1992	1,786,734	1,491,495	83%
1993	1,673,028	1,461,028	87%
1994	1,556,022	1,249,022	80%
1995	1,652,610	1,222,610	74%
1996	1,630,000	1,310,000	80%
1997	1,553,000	1,215,000	78%
1998	1,561,000	1,133,000	73%
1999	1,497,000	1,078,000	72%
2000	1,500,000	875,000	58%
2001	1,416,000	836,000	59%
2002	1,438,000	825,000	57%
2003	1,325,000	748,000	56%
2004	1,373,000	713,000	52%
2005	1,762,000	672,000	38%
2006	2,247,000	767,000	34%
2007	2,807,000	1,257,000	45%

	H	I	J	K
	UBC Consumption	UBC Scrap Exports	UBC Scrap Imports	Domestic Recycled Non-UBC Old Scrap Consumed Domestically
Source	USGS (2008)	US ITC (2009)	US ITC (2009)	(F-H+J)/(F-H+J+I)
1946	0	na	na	na
1947	0	na	na	na
1948	0	na	na	na
1949	0	na	na	na
1950	0	na	na	na
1951	0	na	na	na
1952	0	na	na	na
1953	0	na	na	na

	H	I	J	K
	UBC Consumption	UBC Scrap Exports	UBC Scrap Imports	Domestic Recycled Non-UBC Old Scrap Consumed Domestically
Source	USGS (2008)	US ITC (2009)	US ITC (2009)	(F-H+J)/(F-H+J+I)
1954	0	na	na	na
1955	0	na	na	na
1956	0	na	na	na
1957	0	na	na	na
1958	0	na	na	na
1959	0	na	na	na
1960	0	na	na	na
1961	0	na	na	na
1962	0	na	na	na
1963	0	na	na	na
1964	0	na	na	na
1965	0	na	na	na
1966	0	na	na	na
1967	0	na	na	na
1968	0	na	na	na
1969	0	na	na	na
1970	0	na	na	na
1971	0	na	na	na
1972	0	na	na	na
1973	0	na	na	na
1974	0	na	na	na
1975	76,058	na	na	na
1976	98,707	na	na	na
1977	120,349	na	na	na
1978	151,651	na	na	na
1979	163,259	na	na	na
1980	271,470	na	na	na
1981	461,127	na	na	na
1982	511,404	na	na	na
1983	520,889	na	na	na
1984	542,780	na	na	na
1985	591,265	na	na	na
1986	524,385	na	na	na
1987	554,972	na	na	na
1988	667,524	na	na	na
1989	677,995	17,354	22,738	27%
1990	886,657	2,854	37,028	42%
1991	869,283	2,805	31,602	45%
1992	961,198	3,383	44,089	66%
1993	912,002	996	51,849	74%
1994	949,550	565	76,060	55%

	H	I	J	K
	UBC Consumption	UBC Scrap Exports	UBC Scrap Imports	Domestic Recycled Non-UBC Old Scrap Consumed Domestically
Source	USGS (2008)	US ITC (2009)	US ITC (2009)	(F-H+J)/(F-H+J+I)
1995	965,216	2,820	101,054	46%
1996	920,563	6,325	81,471	60%
1997	1,003,183	3,252	103,027	48%
1998	916,545	3,921	112,484	44%
1999	1,038,511	2,006	106,366	26%
2000	877,376	4,863	106,603	14%
2001	791,662	6,097	88,859	19%
2002	758,520	39,796	71,113	19%
2003	723,439	5,743	78,415	15%
2004	784,703	4,003	84,518	2%
2005	701,900	16,754	94,438	6%
2006	700,600	15,107	99,553	10%
2007	696,300	7,329	86,870	30%

As Figure A.2 shows, the percentage of old scrap recovered in the U.S. that is consumed domestically showed a strong upward trend from 1961 until 1994. It was after this point that scrap exports began to rise as U.S. old scrap consumption declined.

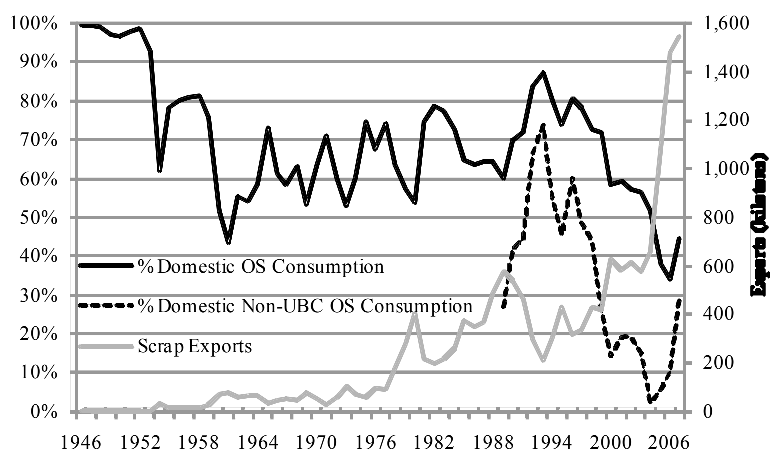


Figure A.2 Percentage of Domestically-Recycled Old Scrap Consumed Domestically and Old Scrap Exports in Kilotons (USGS, 2009)

A.5 Time Series Data

Table A.5 First-Differenced Natural Log (Δln) of Net Additions to In-Use Stock for Construction and Transportation Sectors

Data	dlnNAS Construction	dlnNAS per Capita Construction	dlnNAS Transportation	dlnNAS per Capita Transportation
Source	BEA (2009)	BEA (2009), Census Bureau (2009)	BEA (2009)	BEA (2009), Census Bureau (2009)
1947	0.1110	0.0938	-0.0860	-0.1032
1948	-0.1277	-0.1450	-0.2985	-0.3158
1949	0.3291	0.3087	0.7943	0.7738
1950	-0.1108	-0.1278	0.3542	0.3373
1951	0.0257	0.0086	0.3388	0.3217
1952	0.4230	0.4064	0.1210	0.1045
1953	-0.2733	-0.2908	-4.4790	-4.4966
1954	0.4568	0.4392	2.7965	2.7789
1955	0.1233	0.1056	0.6696	0.6518
1956	-0.0720	-0.0901	-0.1503	-0.1684
1957	-0.1510	-0.1677	0.4607	0.4440
1958	0.4360	0.4193	1.1271	1.1104
1959	0.0300	0.0141	-0.1500	-0.1658
1960	0.2303	0.2137	-0.2415	-0.2581
1961	0.0384	0.0230	0.3312	0.3158
1962	0.1284	0.1140	0.0255	0.0111
1963	-0.0282	-0.0421	-0.3986	-0.4125
1964	0.1876	0.1751	0.3308	0.3183
1965	0.1344	0.1228	0.1587	0.1471
1966	-0.0588	-0.0696	-0.3965	-0.4074
1967	0.1150	0.1050	0.0800	0.0700
1968	0.0611	0.0513	0.0408	0.0310
1969	-0.0473	-0.0590	-0.7399	-0.7515
1970	0.2319	0.2193	0.4996	0.4869
1971	0.1961	0.1854	0.6326	0.6219
1972	0.0675	0.0580	0.3048	0.2952
1973	-0.1630	-0.1721	-0.3212	-0.3304
1974	-0.3671	-0.3770	-1.1707	-1.1806
1975	0.2523	0.2428	0.9938	0.9843
1976	0.0512	0.0411	0.2314	0.2214
1977	0.0666	0.0560	0.1058	0.0952
1978	-0.0860	-0.0971	-0.1111	-0.1221
1979	-0.2151	-0.2247	-1.2192	-1.2288
1980	-0.0643	-0.0741	-0.0892	-0.0990
1981	-0.0698	-0.0793	-1.2430	-1.2525
1982	0.1964	0.1873	1.6875	1.6784
1983	-0.0573	-0.0660	0.3267	0.3181

Data	dlnNAS Construction	dlnNAS per Capita Construction	dlnNAS Transportation	dlnNAS per Capita Transportation
Source	BEA (2009)	BEA (2009), Census Bureau (2009)	BEA (2009)	BEA (2009), Census Bureau (2009)
1984	0.0058	-0.0030	-0.4984	-0.5073
1985	-0.0194	-0.0287	-0.0058	-0.0151
1986	-0.0006	-0.0096	0.5981	0.5892
1987	-0.2093	-0.2184	-0.1098	-0.1189
1988	-0.1805	-0.1899	0.7559	0.7465
1989	-0.0595	-0.0702	0.0649	0.0543
1990	-0.3852	-0.3960	-0.0896	-0.1003
1991	0.2036	0.1923	0.5052	0.4939
1992	0.2132	0.2025	0.5897	0.5789
1993	0.0248	0.0149	0.3430	0.3332
1994	-0.7461	-0.7555	-0.2386	-0.2481
1995	0.2216	0.2125	-0.0138	-0.0229
1996	-0.4107	-0.4203	-0.0510	-0.0606
1997	0.0377	0.0285	0.1068	0.0976
1998	0.1743	0.1653	0.1910	0.1820
1999	-0.3687	-0.4002	-0.0994	-0.1309
2000	-1.6021	-1.6149	-0.4707	-0.4835
2001	-0.8795	-0.8889	0.0232	0.0139
2002	na	na	-0.1250	-0.1336
2003	na	na	-0.1118	-0.1210
2004	na	na	-0.3598	-0.3689
2005	na	na	-0.2833	-0.2927
2006	na	na	-4.4328	-4.4426

Table A.6 First-Differenced Natural Log (Δln) of Gross Domestic Product (GDP)

Data	ΔlnGDP	ΔlnGDP per Capita	ΔlnGDP Construction	ΔlnGDP per Capita Construction	ΔlnGDP Automobile Bodies, etc
Source	BEA (2009)	BEA, Census Bureau (2009)	BEA (2009)	BEA, Census Bureau (2009)	BEA (2009)
1947	-0.0094	-0.0286	0.2187	0.2015	na
1948	0.0427	0.0255	0.0089	-0.0084	na
1949	-0.0052	-0.0225	0.1324	0.1120	na
1950	0.0837	0.0632	0.1771	0.1602	na
1951	0.0746	0.0576	0.0750	0.0579	na
1952	0.0376	0.0204	0.0355	0.0189	na
1953	0.0448	0.0283	0.0116	-0.0060	na
1954	-0.0068	-0.0244	0.0667	0.0490	na
1955	0.0689	0.0513	0.0973	0.0795	na
1956	0.0192	0.0015	0.0476	0.0296	na

Data	ΔlnGDP	ΔlnGDP per Capita	ΔlnGDP Construction	ΔlnGDP per Capita Construction	ΔlnGDP Automobile Bodies, etc
Source	BEA (2009)	BEA, Census Bureau (2009)	BEA (2009)	BEA, Census Bureau (2009)	BEA (2009)
1957	0.0199	0.0018	-0.0141	-0.0308	na
1958	-0.0096	-0.0263	0.0815	0.0648	na
1959	0.0687	0.0520	0.0172	0.0014	na
1960	0.0245	0.0086	0.0418	0.0253	na
1961	0.0230	0.0064	0.0712	0.0558	na
1962	0.0589	0.0435	0.0664	0.0521	na
1963	0.0428	0.0284	0.0921	0.0782	na
1964	0.0565	0.0426	0.0932	0.0807	na
1965	0.0622	0.0497	0.0961	0.0846	na
1966	0.0631	0.0516	0.0422	0.0313	na
1967	0.0248	0.0140	0.0913	0.0813	na
1968	0.0471	0.0371	0.1094	0.0996	na
1969	0.0304	0.0206	0.0595	0.0479	na
1970	0.0017	-0.0099	0.0949	0.0823	na
1971	0.0330	0.0204	0.0999	0.0892	na
1972	0.0516	0.0409	0.1247	0.1151	na
1973	0.0560	0.0465	0.0685	0.0594	na
1974	-0.0051	-0.0142	0.0108	0.0009	na
1975	-0.0019	-0.0118	0.1337	0.1242	na
1976	0.0519	0.0424	0.0969	0.0868	na
1977	0.0451	0.0351	0.1686	0.1580	0.0789
1978	0.0542	0.0436	0.1302	0.1191	-0.0598
1979	0.0311	0.0201	0.0257	0.0161	-0.3313
1980	-0.0023	-0.0119	0.0114	0.0016	0.2524
1981	0.0249	0.0151	-0.0230	-0.0326	-0.0554
1982	-0.0195	-0.0291	0.0820	0.0728	0.2713
1983	0.0442	0.0351	0.1621	0.1534	0.2541
1984	0.0694	0.0607	0.1159	0.1070	0.0284
1985	0.0405	0.0316	0.1179	0.1087	0.0078
1986	0.0341	0.0248	0.0493	0.0404	-0.0187
1987	0.0332	0.0243	0.0643	0.0553	0.0401
1988	0.0405	0.0314	0.0507	0.0412	-0.0449
1989	0.0348	0.0253	0.0150	0.0043	-0.1481
1990	0.0186	0.0079	-0.0765	-0.0872	0.0396
1991	-0.0017	-0.0124	0.0099	-0.0014	0.2389
1992	0.0327	0.0214	0.0657	0.0550	0.1613
1993	0.0264	0.0156	0.0999	0.0901	0.1978
1994	0.0394	0.0296	0.0449	0.0354	-0.0436
1995	0.0247	0.0153	0.0826	0.0734	0.0071

Data	ΔlnGDP	ΔlnGDP per Capita	ΔlnGDP Construction	ΔlnGDP per Capita Construction	ΔlnGDP Automobile Bodies, etc
Source	BEA (2009)	BEA, Census Bureau (2009)	BEA (2009)	BEA, Census Bureau (2009)	BEA (2009)
1996	0.0363	0.0271	0.0798	0.0702	0.0365
1997	0.0440	0.0344	0.1035	0.0943	0.0538
1998	0.0409	0.0317	0.0825	0.0735	0.0589
1999	0.0435	0.0345	0.0696	0.0381	0.0231
2000	0.0360	0.0044	0.0743	0.0615	-0.1300
2001	0.0075	-0.0053	0.0269	0.0175	0.1368
2002	0.0159	0.0065	0.0284	0.0198	0.0428
2003	0.0248	0.0162	0.0831	0.0739	-0.1492
2004	0.0383	0.0291	0.1158	0.1067	-0.0849
2005	0.0317	0.0227	0.0649	0.0555	0.0010
2006	0.0327	0.0232	-0.0560	-0.0658	0.0020

ΔlnGDP per Capita Automobile Bodies, etc
BEA, Census Bureau (2009)
na
na
na
na
na
na
na
na
na
na
na
na
na
na
na
na
na
na
na
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na
na
na
na

ΔlnGDP per Capita Automobile Bodies, etc BEA, Census Bureau (2009)
na
na
na
na
na
na
0.0683
-0.0708
-0.3409
0.2426
-0.0649
0.2621
0.2454
0.0196
-0.0015
-0.0277
0.0310
-0.0543
-0.1588
0.0289
0.2275
0.1505
0.1879
-0.0531
-0.0021
0.0270
0.0447
0.0499
-0.0084
-0.1428
0.1274
0.0342
-0.1584
-0.0940
-0.0084
-0.0077

A.6 Regression Estimates of Disaggregated Net Additions to In-Use Stocks and GDP

The following tables summarize the model estimation results for net additions to in-use stock (NAS) in the construction (Table A.7) and transportation (Table A.8) sectors. Due to non-normality of the first differenced natural log (Δln) of both NAS series, a robust MM-estimator (Yohai, 1987) was used.

Table A.7 Regression Estimates for Construction NAS and GDP

Dependent Variable	Δln Construction NAS	Δln Construction NAS	Δln Construction NAS per Capita	Δln Construction NAS per Capita
Period	1948-2001	1948-2001	1948-2001	1948-2001
Intercept	-0.104 (-1.81)	-0.126** (-2.74)	-0.0871 (-1.72)	-0.0652 (-1.80)
Δln GDP (construction)	1.06 (1.82)	-	-	-
Δln GDP per Capita (construction)	-	-	0.992 (1.70)	-
Δln GDP	-	4.95** (4.43)	-	-
Δln GDP per Capita	-	-	-	4.48** (3.90)
LΔln GDP (construction)	0.837 (1.32)	-	-	-
LΔln GDP per Capita (construction)	-	-	0.804 (1.27)	-
LΔln Construction NAS	-0.0524 (-0.453)	0.0163 (0.168)	-	-
LΔln Construction NAS per Capita	-	-	-0.0623 (-0.533)	-0.0044 (-0.0431)
R^2	0.106	0.219	0.0969	0.187
Breusch-Godfrey (order 1)	0.114 P-value: 0.735	0.117 P-value: 0.732	0.0846 P-value: 0.7712	0.0077 P-value: 0.930
Breusch-Pagan (studentized)	0.860 P-value: 0.835	0.502 P-value: 0.778	1.07 P-value: 0.785	0.3931 P-value: 0.822

T-stats provided in parenthesis. * denotes significance at the 5% level; ** denotes significance at the 1% level. Breusch-Godfrey tests for serial correlation. Breusch-Pagan tests for heteroskedasticity.

Table A.8 Regression Estimates for Transportation NAS and GDP

Dependent Variable	Δln Transportation NAS	Δln Transportation NAS	Δln Transportation NAS per Capita	Δln Transportation NAS per Capita
Period	1978-2006	1948-2006	1978-2006	1948-2006
Intercept	0.0227 (0.215)	-0.388** (-3.33)	0.0246 (0.236)	-0.213* (-2.34)
Δln GDP (transportation)	0.722 (0.883)	-	-	-

Δln GDP per Capita (transportation)	-	-	0.735 (0.897)	-
Δln GDP	-	13.6** (4.71)	-	-
Δln GDP per Capita	-	-	-	13.1** (4.37)
L.Δln Transportation NAS	-0.178 (-1.77)	-0.0158 (-0.237)	-	-
L.Δln Transportation NAS per Capita	-	-	-0.178 (-1.77)	-0.0156 (-0.229)
R^2	0.0988	0.178	0.0990	0.163
Breusch-Godfrey (order 1)	1.46 P-value: 0.227	0.0165 P-value: 0.898	1.50 P-value: 0.220	0.0304 P-value: 0.862
Breusch-Pagan (studentized)	1.17 P-value: 0.557	9.34 P-value: 0.00938	1.16 P-value: 0.559	9.96 P-value: 0.00687

T-stats provided in parenthesis. * denotes significance at the 5% level; ** denotes significance at the 1% level. Breusch-Godfrey tests for serial correlation. Breusch-Pagan tests for heteroskedasticity.

A.7 Environmental Kuznets Curve (EKC) Analysis

Statistical analysis was also performed to test for an environmental Kuznets curve (EKC) in the relationship between stocks and GDP for the period of 1947 – 2007. Testing was conducted using ordinary least squares regression (OLS) on the general form

$$\ln y_t = \alpha + \beta_1 \ln x_t + \beta_2 \left(\ln x_t\right)^2 + \varepsilon_t , \tag{A.3}$$

where y_t is the measure of stocks and x_t is GDP. A negative and statistically significant value for the coefficient of the square of GDP, β_2, indicates the presence of an EKC. Three measures of stocks were used: Δln NAS, Δln NAS per capita, and the annual cumulative in-use stocks. It should be noted that the cumulative in-use stocks and GDP contain a unit root and result in spurious regression using OLS. These results are included in Table A.9 simply for completeness.

Table A.9 presents results of the EKC analysis. No statistically significant coefficient for the squared GDP term was found in any of the regressions of GDP on Δln NAS and Δln NAS per capita. Conversely, regression of GDP on the measure of cumulative in-use stocks did reveal statistically significant support for an EKC; however, these results should be treated with skepticism. In addition to both in-use stocks and GDP being nonstationary, Breusch-Godfrey testing indicates autocorrelated errors. In addition, the significance of both GDP terms disappears when the one-year lagged value of cumulative stocks is included in the regression equation. It is concluded from these results that accumulation of U.S. aluminum in-use stocks does not follow an EKC form.

Table A.9 Regression Estimates for EKC Analysis

Dependent Variable	Δln NAS	Δln NAS per capita	Δln NAS per capita	ln Stock	ln Stock
Intercept	-0.321** (-4.29)	-0.209** (-3.52)	-0.200** (-3.34)	-35.1** (-20.4)	-1.79 (-1.23)
L Δln NAS per capita	-	-	-0.0892 (-0.748)	-	-
L.lnStock	-	-	-	-	0.930** (24.1)
Δln GDP	9.95* (2.44)	-	-	-	-
Δln GDP per capita	-	8.92** (3.27)	8.33** (2.79)	-	-
Δln GDP^2	-3.85 (-0.0630)	-	-	-	-
Δln GDP per $capita^2$	-	14.6 (0.220)	22.4 (0.319)	-	-
lnGDP	-	-		11.1** (26.8)	0.673 (1.51)
$lnGDP^2$	-	-		-0.573** (-23.2)	-0.0368 (-1.57)
R^2	0.32	0.38	0.30	0.99	0.99
Breusch-Godfrey (order 1)	0.18 P-value: (0.670)	0.179 P-value: (0.672)	0.054 P-value: (0.816)	37.3** P-value: (1.01e-09)	11.3** P-value: (0.0007436)
Breusch-Pagan (studentized)	3.90 P-value: (0.142)	5.15 P-value: (0.0762)	5.60 P-value: (0.133)	5.91 P-value: (0.0521)	6.05 P-value: (0.109)

T-stats provided in parenthesis. * denotes significance at the 5% level; ** denotes significance at the 1% level. Breusch-Godfrey tests for serial correlation. Breusch-Pagan tests for heteroskedasticity.

A.8 References

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APPENDIX B

SUPPLEMENTAL MATERIAL: CHAPTER 3

B.1 Model Equations

The life cycle GHG emissions for ingot production in region *i*, GHG_i, are calculated as

$$GHG_i = P_i \times \left[\frac{kWh_i}{(1-\theta_i)} \times \sum_j \left(F_{i,j} \times C_{i,j} \right) + \left(PFC_i + \varphi \right) \right]$$

$$j = \text{coal, natural gas, and fuel oil} \tag{B.1}$$

$$i = \text{Africa, North America, Latin America, Asia, Europe, and Oceania}$$

where P_i is the primary ingot production; kWh_i is the electricity intensity of electrolysis per kg primary ingot; θ_i is the electricity generation-weighted average T&D losses; $F_{i,j}$ is the percentage of electricity generated by fuel *j* consumed by smelters in region *i*; $C_{i,j}$ is the life cycle electricity generation-weighted CO_2 emissions per kWh for electricity produced by fuel *j*, including the upstream emissions associated with producing fuel *j*; PFC_i is the PFC emissions per kg primary ingot; and φ is the sum of life cycle GHG emissions per kg primary ingot from bauxite mining, alumina refining, anode manufacturing, and anode oxidation.

The production-weighted life cycle GHG emissions intensity for region *i*, EFP_i, is calculated as

$$EFP_i = \frac{GHG_i}{P_i} \tag{B.2}$$

The consumption-weighted life cycle GHG emissions intensity for region *i*, EFC_i, is calculated as

$$EFC_i = \frac{\sum_k \left(M_{i,k} \times EFP_k \right) + \left(A_i - \sum_k M_{i,k} \right) \times EFP_i}{A_i} \tag{B.3}$$

where $M_{i,k}$ is the ingot imports of region *i* from region *k*, EFP_k is the life cycle GHG emission factor of region *k*, and A_i is the apparent consumption of ingot in region *i*.

B.2 Sources for Model Data and Model Parameters

Electricity intensity (i.e. the amount of electricity consumed by smelters per unit mass aluminum produced) and electricity fuel mix of primary smelters in the six world regions was compiled for the period of 1990 to 2005 (International Aluminum Institute 2007). It should be noted that these data only represent approximately 70% of the world's smelters and specifically do not include smelters in China and, until 2004, the former Soviet Union (IAI 2007). Chinese and former Soviet Union data are supplemented by utilizing previously published electricity intensity and fuel mix figures for 1998 (IAI 2000). Electricity intensity data for the remaining years were extrapolated by applying an equation for the trend in electricity intensity observed in Asia from 1990 to 2005 for China and in Europe from 1990 to 2003 for the former Soviet Union. Depending on smelter vintage and capital improvements, this approach may under- or over-estimate trends in electricity efficiency. Smelters in the former Soviet Union have likely experienced more significant capital improvements since 1991 than their counterparts in Europe (Propokov 2005) and assuming the same trend in electricity efficiency would underestimate the effects of these improvements. Fuel mix data for both Chinese and Soviet smelters were assumed to remain unchanged from the published 1998 value.

Annual CO_2 emission intensities of coal, oil, and natural gas electricity generation are available on a country level (International Energy Agency 2007a). These data are calculated by the International Energy Agency (IEA) using default net calorific and carbon emission factors for each type of fuel (Houghton et al. 1997). The annual differences in the final CO_2 emission factor for each type of fossil fuel generated electricity represent changes in the generation efficiencies and fuel choices of generating facilities. Using data only from countries producing primary ingot, regional CO_2 intensity values for fossil fuel electricity generation were developed by weighting each CO2 intensity of generation by the total generation of the fuel source for that country (IEA 2007b; IEA 2007c).

The IAI published regional PFC emission factors only for the period of 1994 through 1997 (Gibson 2007). With the exception of North America, Oceania, and Europe the average annual emissions change for each region during this period is used to extrapolate emissions data for 1990 through 1993 and 1998 through 2005. North American PFC emissions were calculated based on a production-weighted average of reported PFC emissions for the U.S. and Canada (U.S. EPA 2006; Environment Canada 2007). PFC emissions for Oceania were calculated from data provided by the Australian Government (Australian Government Department of Climate Change 2007). Estimates for the PFC emission intensity of Europe was calculated using a production weighted average of IAI data and data for Russia, Hungary, Poland, and Romania (U.S. EPA 2008).

The life cycle GHG emissions intensity associated with the energy and process inputs for bauxite mining, alumina refining, anode production, and anode oxidation are assumed to be constant at 0.25 kg CO_2-eq/kg primary ingot, 1.91 kg CO_2-eq/kg primary ingot, 0.37 kg CO_2-eq/kg primary ingot, and 1.626 kg CO_2-eq/kg primary ingot, respectively (IAI 2000).

The model does not include emissions from the transport of ingot from casthouse to port for export due to a lack of data and the source's extremely small estimated contribution to total GHG emissions. Data on the geographic location of the world's smelters and the modes of transporting ingot from smelter to port are limited. It would be possible to examine the satellite images of a sample of smelters (assuming that they have an ingot casthouse on site) and speculate a composite transportation mode and distance for each region. We estimated the contribution of emissions from this portion of transportation to the total emissions and found that it was very small (extreme case: <0.22%). Using data from NREL (2007), the life cycle GHG intensities of diesel combination truck, diesel locomotive, and residual barge are 0.094 kg CO_2-eq/metric ton-km, 0.022 kg CO_2-eq/metric ton-km, and 0.034 kg CO_2-eq/metric ton-km. The arithmetic average of these values is 0.05 kg CO_2-eq/metric ton-km.

Incorporating these transportation emissions would have the greatest impact on the emissions of Latin America. The region is a large exporter of ingot and it has the lowest GHG intensity. In 2005, the region exported 1.3 million metric tons of ingot and it domestic production emitted 16.8 million metric tons CO_2-eq. Smelters in the region are relatively close to sea ports, say a distance of 300 km, and using the average transportation emission factor above results in the addition of 19,500 metric tons CO_2-eq to the region's total. This represents an increase in total emissions of 0.12%. Even if this distance was covered entirely by a diesel truck, the additional emissions would be 36,660 metric tons CO_2-eq (0.22%).

As an additional example, in 2005 Asia exported 134,700 metric tons of ingot and domestic production emitted 243 million metric tons CO2-eq. Assuming that ingot travels 2,000 km (a probable overestimate) to the point of export results in the addition of 13,470 metric tons CO2-eq, an increase of 0.0055%.

B.3 Fuel Mixes for Smelters and Electricity Grids by Region

Smelter fuel mix data from the IAI were then compared with overall electricity fuel mix data obtained from the IEA (2007b; 2007c) in order to identify any differences between the sources of electricity used by a region's smelters and the region as a whole. The country-level IEA data were aggregated on regional bases using only primary aluminum producing countries. In general, from 1999 to 2005 the overall electricity fuel mix of a region exhibits much less annual variation relative to the region's primary smelters. There are, however, striking differences between the fuel mixes of the two data sources, as demonstrated by Figure B.1. In most cases hydro, coal-fired, and nuclear generation constitute nearly 100% of the fuel mix of smelters, yet make up less than 80% of the fuel mix of the overall region. Aluminum smelters in each region receive a much higher portion of their electricity from hydro than the rest of the electricity consumers in their same region. The coal intensive regions include Africa, Asia, and Oceania, while the hydro intensive regions are North America, Latin America, and Europe. In particular, Africa and Oceania rely on coal-fired generation for greater than 60% and 70%, respectively, of their electricity consumption.

The largest changes in fuel mix occurred in Africa and Europe. In Europe, the portion of hydro has grown significantly in the recent years. In 1990 hydro provided approximately 43% of smelter electricity, yet from 2003 to 2005 the fraction of hydro increased from 47% to 70%. Concurrently, the fraction of coal-fired electricity decreased from its peak of nearly 32% in 1990 to 11% in 2005. Europe also distinguishes itself from other regions by the large fraction of electricity consumed from nuclear generation. On average, nuclear accounted for 20% of electricity consumed by European smelters from 1990 to 2005. The fraction of nuclear peaked in 1994, at nearly 26%. The lowest fraction of consumption from nuclear sources occurred in 2005, at 12%.

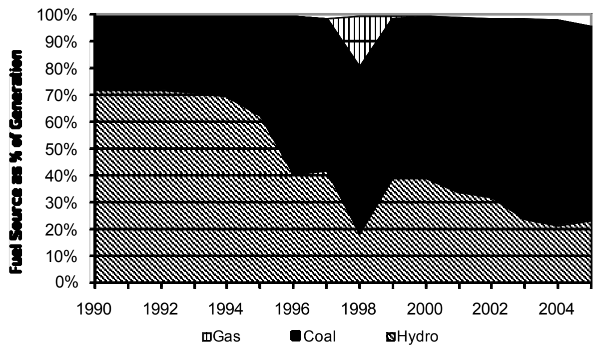
In Africa, coal-fired generation has replaced hydro as the dominant source of electricity for smelters. The fraction of hydro was approximately 72% in 1990 before decreasing to 23% by 2005. The drop in hydro was especially precipitous in 1998, when the fraction decreased from nearly 42% to 18%. Natural gas fired generation was used as a stopgap during this year and constituted 19% of the electricity consumption. During the period preceding and following 1998, the fraction of natural gas did not exceed 1%. As hydro has disappeared as an electricity source, the fraction of electricity from coal-fired generation has increased from approximately 27% in 1990 to 72% in 2005.

In North America, the fraction of electricity obtained from coal-fired generation was at its peak at 37% in 1990 before falling to 30% in 2005. Throughout this period, the fraction of electricity source from hydro has remained between 61% and 69%. A general increase in the consumption of hydro occurred between 1990 and 1999, when its fraction grew from 61% to 74%. After this period, the trend reversed and the fraction decreased to 64% in 2002. It appears that a new increase in hydroelectricity consumption is underway, as its fraction has since grown to 69% in 2005.

As published, the electricity fuel mix data do not include Chinese primary smelters. It has been indicated that in 1998, the electricity fuel mix of smelters in China was 76% coal, 21% hydro, 1.6% oil, and 1.3% nuclear (IAI, 2000). As discussed in the methods section, these figures were assumed to remain constant for the entire period of 1990 through 2005.

Smelter fuel mix data from the IAI were then compared with overall electricity fuel mix data obtained from the IEA (IEAb, 2007) in order to identify any differences between the sources of electricity used by a region’s smelters and the region as a whole. The country-level IEA data were aggregated on regional bases using only primary aluminum producing countries.

In general, from 1999 to 2005 the overall electricity fuel mix of each region exhibits much less variation relative to the region’s primary smelters. Gradual movements from one fuel source to another were noted in the overall regional data, but were much more pronounced in the smelter data.



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Fuel Source as % of Generation
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Coal
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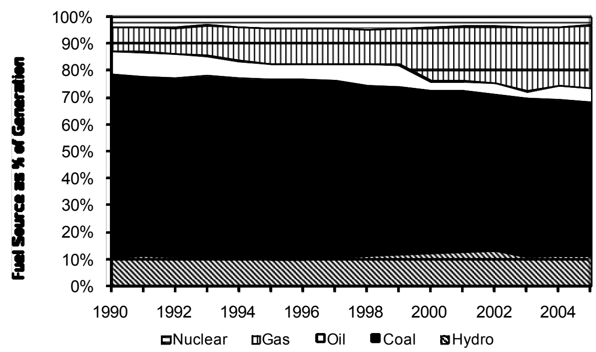
(a)



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Fuel Source as % of Generation
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Nuclear
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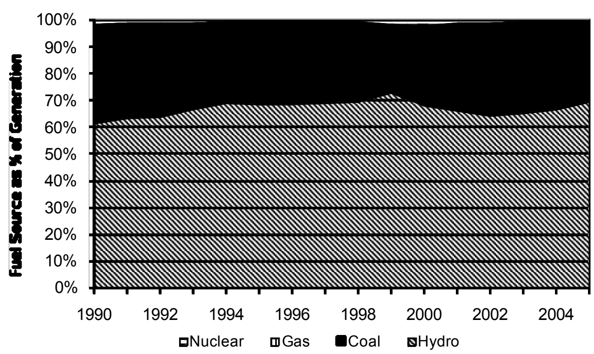
(b)



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Fuel Source as % of Generation
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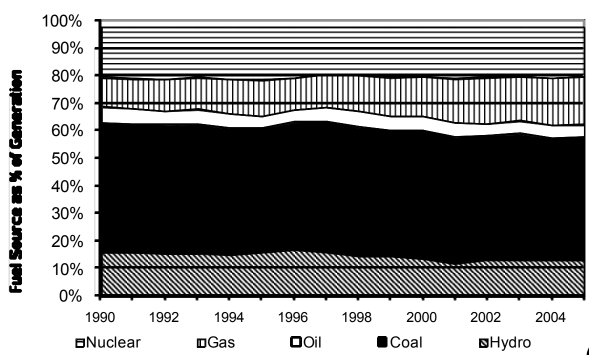
(c)



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0%
Fuel Source as % of Generation
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1996
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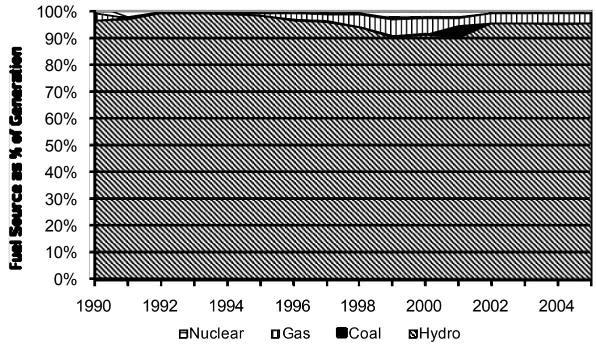
Nuclear
Gas
Oil
Coal
Hydro

(d)


Fuel Source as % of Generation
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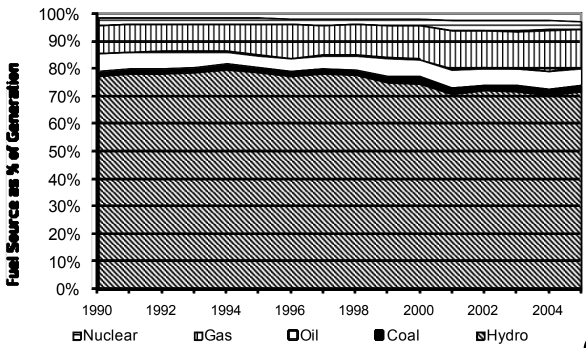
Fuel Source as % of Generation
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Nuclear
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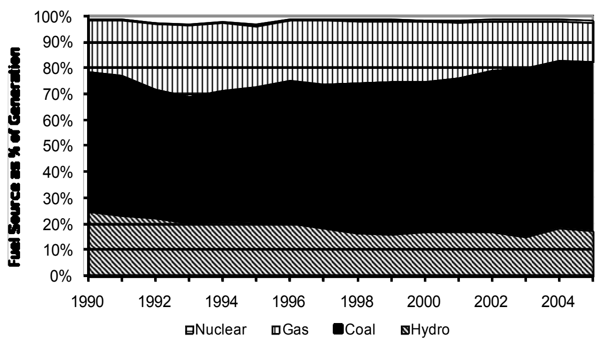
Fuel Source as % of Generation
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90%
80%
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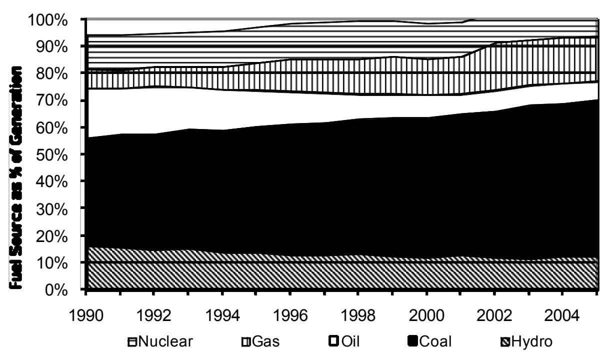
Nuclear
Gas
Coal
Hydro

(g)


Fuel Source as % of Generation
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90%
80%
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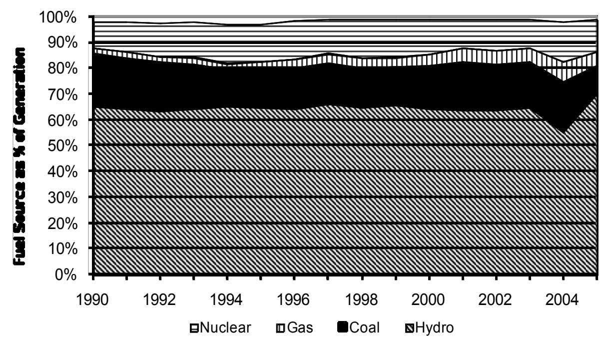
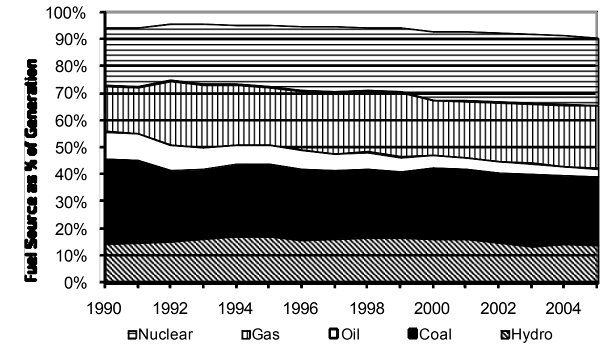
2002
2004
Nuclear
Gas
Oil
Coal
Hydro

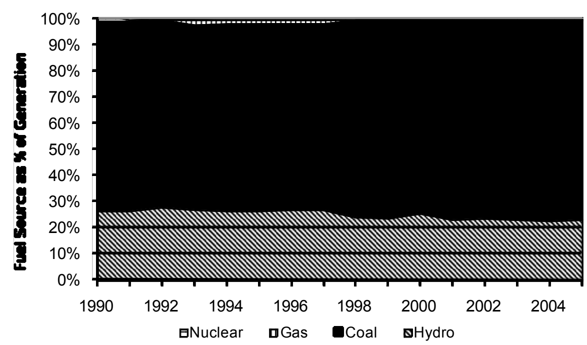
(h)



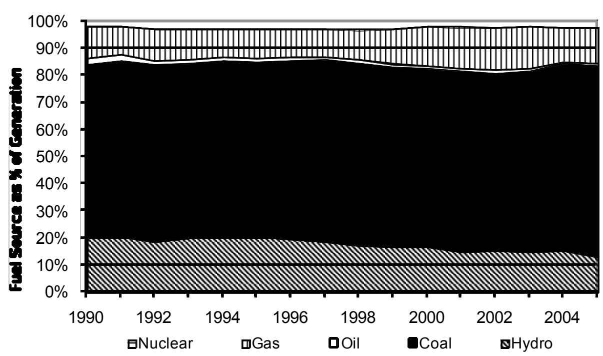
(i)



(j)



(k)



(l)

Figure B.2 Fuel Mixes of Smelters (IAI 2007) and Electricity Grid (IEA 2007b; IEA 2007c) for Africa (a and b), North America (c and d), Latin America (e and f), Asia (g and h), Europe (i and j), and Oceania (k and l).

B.4 Uncertainty Analysis

The electricity intensity of electrolysis (kWh/t), PFC emission intensity (kg CO_2-eq/kg), and CO_2 emission factors of fossil fuel electricity generation (kg CO_2/kWh) were chosen as parameters for the uncertainty analysis based on their contribution to the GHG emissions and the availability of statistical data. A normal distribution was assumed for calculating confidence intervals.

Due to the linearity of the emissions function and the small coefficient of variation of many of the chosen parameters, the annual standard error of GHG emissions for region i, $s_{\overline{GHG_i}}$, is calculated as a first order approximation

$$s_{\overline{GHG_i}} = \sqrt{\sum_j \left(\frac{\partial GHG_i}{\partial x_{i,j}} s_{\bar{x}_{i,j}} \right)^2 + 2 \sum_j r_{x_{i,j} x_{i,j+1}} \left(\frac{\partial GHG_i}{\partial x_{i,j}} s_{\bar{x}_{i,j}} \right) \left(\frac{\partial GHG_i}{\partial x_{i,j+1}} s_{\bar{x}_{i,j+1}} \right)} \tag{B.4}$$

j = kWh/t, PFC kg CO_2eq/kg, kg CO_2 / kWh oil, kg CO_2 / kWh coal, and kg CO_2 / kWh gas.

where $s_{\bar{x}_{i,j}}$ is the standard error of parameter j and $r_{x_{i,j} x_{i,j+1}}$ is the correlation coefficient between parameters j and j+1 for region i.

The resulting 95% confidence interval values are presented in Tables B.1 through B.6.

Table B.1 Production Emissions 95% Confidence Interval Value (kg CO_2-eq)

	Africa	N. America	L. America	Asia	Europe	Oceania	World
1990	1.1E+09	1.1E+10	3.1E+09	1.2E+10	1.7E+10	3.8E+09	1.8E+10
1991	8.0E+08	7.5E+09	2.2E+09	1.6E+10	1.2E+10	3.1E+09	1.6E+10
1992	8.1E+08	7.4E+09	2.1E+09	2.2E+10	1.1E+10	3.1E+09	1.9E+10
1993	8.0E+08	7.1E+09	2.1E+09	1.5E+10	9.5E+09	3.3E+09	1.5E+10
1994	7.9E+08	6.6E+09	2.1E+09	9.7E+09	8.7E+09	3.2E+09	1.1E+10
1995	8.7E+08	6.1E+09	2.1E+09	9.2E+09	8.3E+09	3.1E+09	1.1E+10
1996	1.8E+09	7.4E+09	2.5E+09	1.0E+10	9.5E+09	3.5E+09	1.2E+10
1997	2.1E+09	8.1E+09	2.8E+09	9.8E+09	1.1E+10	4.0E+09	1.3E+10
1998	2.1E+09	7.5E+09	2.6E+09	1.0E+10	1.1E+10	4.4E+09	1.3E+10
1999	2.1E+09	8.8E+09	2.8E+09	1.2E+10	1.1E+10	4.7E+09	1.5E+10
2000	2.1E+09	8.1E+09	2.5E+09	1.3E+10	1.0E+10	4.4E+09	1.4E+10
2001	2.2E+09	5.9E+09	1.7E+09	1.4E+10	9.1E+09	4.2E+09	1.4E+10
2002	2.2E+09	5.0E+09	1.8E+09	1.7E+10	8.4E+09	4.3E+09	1.5E+10
2003	2.5E+09	4.6E+09	1.9E+09	2.1E+10	8.8E+09	4.4E+09	1.7E+10
2004	3.3E+09	4.8E+09	2.2E+09	2.5E+10	1.0E+10	4.3E+09	2.1E+10
2005	2.9E+09	3.7E+09	1.7E+09	3.2E+10	6.7E+09	4.3E+09	2.5E+10

Table B.2 Production Emissions Intensity 95% Confidence Interval Value (kg CO_2-eq/kg)

	Africa	N. America	L. America	Asia	Europe	Oceania	World
1990	1.9	1.9	1.8	6.1	2.2	2.5	0.94
1991	1.3	1.2	1.1	7.0	1.6	2.1	0.82
1992	1.3	1.2	1.1	7.5	1.7	2.1	0.98
1993	1.3	1.2	1.1	5.0	1.5	2.0	0.74
1994	1.3	1.2	1.1	3.0	1.4	2.0	0.58
1995	1.4	1.1	1.0	2.6	1.3	1.9	0.54
1996	1.8	1.2	1.2	2.8	1.5	2.1	0.59
1997	1.9	1.3	1.4	2.5	1.6	2.2	0.60
1998	2.0	1.2	1.3	2.4	1.5	2.3	0.57
1999	1.9	1.4	1.4	2.6	1.5	2.3	0.62
2000	1.8	1.3	1.2	2.5	1.3	2.1	0.58
2001	1.6	1.1	0.87	2.6	1.1	2.0	0.57
2002	1.6	0.89	0.82	2.6	1.0	2.0	0.58
2003	1.8	0.82	0.84	2.5	1.0	2.0	0.62
2004	1.9	0.92	0.94	2.5	1.2	1.9	0.69
2005	1.7	0.67	0.69	2.9	0.74	1.9	0.78

Table B.3 Import Emissions 95% Confidence Interval Value (kg CO_2-eq)

	Africa	N. America	L. America	Asia	Europe	Oceania
1990	1.2E+07	1.7E+08	1.4E+06	2.4E+09	6.0E+08	2.0E+06
1991	1.0E+07	6.2E+07	5.4E+06	1.8E+09	4.9E+08	3.0E+06
1992	8.1E+06	9.7E+07	2.5E+06	1.8E+09	5.0E+08	4.3E+06
1993	8.2E+06	4.6E+08	7.0E+06	1.8E+09	5.0E+08	3.8E+06
1994	8.6E+06	6.8E+08	1.1E+07	1.8E+09	3.8E+08	3.6E+06
1995	7.9E+06	4.0E+08	1.5E+07	1.8E+09	3.2E+08	2.2E+06
1996	6.2E+06	4.2E+08	1.6E+07	2.1E+09	5.2E+08	1.5E+06
1997	7.0E+06	5.0E+08	1.5E+07	2.3E+09	5.0E+08	1.7E+06
1998	9.9E+06	6.7E+08	1.2E+07	2.2E+09	5.5E+08	1.4E+06
1999	4.7E+07	8.7E+08	8.8E+06	2.4E+09	4.6E+08	1.3E+06
2000	6.4E+06	6.9E+08	5.9E+06	2.5E+09	4.5E+08	1.1E+06
2001	5.0E+06	4.1E+08	5.3E+06	2.3E+09	4.1E+08	5.9E+06
2002	3.5E+06	5.0E+08	2.4E+06	2.2E+09	3.9E+08	1.7E+06
2003	6.5E+06	4.8E+08	2.4E+06	2.3E+09	2.9E+08	4.4E+06
2004	8.0E+06	7.2E+08	3.8E+06	2.3E+09	2.9E+08	7.9E+06
2005	7.0E+06	5.1E+08	3.5E+06	2.2E+09	3.4E+08	5.8E+06

Table B. 4 Import Emissions Intensity 95% Confidence Interval Value (kg CO_2-eq/kg)

	Africa	N. America	L. America	Asia	Europe	Oceania
1990	0.81	1.1	0.96	0.87	0.79	1.0
1991	0.50	0.67	0.78	0.63	0.53	0.72
1992	0.49	0.64	0.65	0.62	0.51	0.82
1993	0.55	0.65	0.58	0.60	0.51	0.77
1994	0.65	0.62	0.66	0.58	0.50	0.72
1995	0.56	0.56	0.56	0.50	0.46	0.60
1996	0.65	0.68	0.74	0.61	0.54	0.63
1997	0.60	0.71	0.83	0.67	0.58	0.67

1998	0.68	0.64	0.70	0.81	0.52	0.58
1999	0.86	0.66	0.80	0.74	0.58	0.62
2000	0.50	0.54	0.64	0.60	0.50	0.65
2001	0.41	0.41	0.49	0.62	0.41	1.4
2002	0.39	0.38	0.41	0.64	0.37	1.0
2003	0.37	0.39	0.40	0.62	0.36	1.2
2004	0.45	0.44	0.50	0.59	0.45	1.5
2005	0.46	0.30	0.37	0.58	0.42	1.6

Table B.5 Consumption Emissions 95% Confidence Interval Value (kg CO_2-eq)

	Africa	N. America	L. America	Asia	Europe	Oceania
1990	6.4E+08	6.1E+09	7.3E+08	8.2E+09	1.0E+10	7.4E+08
1991	4.4E+08	3.8E+09	5.3E+08	1.0E+10	6.5E+09	7.2E+08
1992	4.5E+08	4.0E+09	5.5E+08	1.4E+10	5.8E+09	6.6E+08
1993	4.6E+08	3.7E+09	5.2E+08	9.8E+09	4.6E+09	7.7E+08
1994	4.1E+08	3.8E+09	5.6E+08	6.2E+09	3.9E+09	6.8E+08
1995	4.3E+08	3.5E+09	4.1E+08	5.7E+09	3.8E+09	6.6E+08
1996	1.1E+09	4.3E+09	6.6E+08	6.3E+09	4.7E+09	7.2E+08
1997	1.3E+09	4.8E+09	7.9E+08	6.0E+09	4.9E+09	8.5E+08
1998	1.3E+09	4.6E+09	7.1E+08	6.1E+09	5.0E+09	8.5E+08
1999	1.3E+09	5.8E+09	7.3E+08	7.5E+09	4.8E+09	9.8E+08
2000	7.0E+08	5.3E+09	6.0E+08	7.7E+09	4.1E+09	9.7E+08
2001	8.9E+08	3.7E+09	4.3E+08	8.7E+09	4.0E+09	6.8E+08
2002	9.3E+08	3.0E+09	4.6E+08	1.1E+10	3.6E+09	7.8E+08
2003	1.1E+09	2.8E+09	5.1E+08	1.3E+10	3.8E+09	7.5E+08
2004	1.4E+09	3.0E+09	6.6E+08	1.5E+10	4.1E+09	7.8E+08
2005	1.2E+09	2.3E+09	4.6E+08	2.0E+10	3.0E+09	7.7E+08

Table B.6 Consumption Emissions Intensity 95% Confidence Interval Value (kg CO_2-eq/kg)

	Africa	N. America	L. America	Asia	Europe	Oceania
1990	1.3	1.3	1.3	1.8	1.2	1.7
1991	0.88	0.82	0.80	2.1	0.82	1.5
1992	0.89	0.79	0.79	2.6	0.82	1.5
1993	0.88	0.70	0.76	1.7	0.72	1.4
1994	0.91	0.66	0.77	1.00	0.72	1.4
1995	0.93	0.64	0.69	0.81	0.68	1.3
1996	1.3	0.75	0.81	0.92	0.75	1.4
1997	1.3	0.82	0.91	0.85	0.83	1.5
1998	1.4	0.70	0.82	0.90	0.75	1.6
1999	1.2	0.81	0.87	0.97	0.79	1.6
2000	1.2	0.76	0.72	0.86	0.65	1.5
2001	1.1	0.63	0.55	0.99	0.54	1.4
2002	1.1	0.48	0.52	1.1	0.49	1.4
2003	1.2	0.44	0.53	1.1	0.52	1.4
2004	1.3	0.47	0.59	1.1	0.59	1.3
2005	1.1	0.33	0.43	1.4	0.39	1.4

B.5 Global Warming Sensitivity Analysis

The model calculates CO_2 equivalency based on GWPs from the IPCC's Third Assessment Report (TAR). Due to the large change in updated GWP for CF_4, a sensitivity analysis was performed based on latest the 100-year GWPs for CF4 (7,390) and C_2F_6 (12,200) as reported in IPCC's Fourth Assessment Report (FAR). The TAR values used by the model are 5,700 for CF_4 and 11,900 for C_2F_6. It was assumed that the mass fractions of total PFC emissions in each region were 0.84 CF_4 and 0.16 C_2F_6. This assumption was based on the average fractions observed data from Australian smelters from 1990 – 2005. The use of the FAR GWPs resulted in a 22% increase for each region's CO_2-eq PFC emission intensity. For example, Africa's PFC emission intensity in 1990 was 3.01 kg CO_2-eq/kg ingot using the TAR GWPs and 3.67 kg CO_2-eq/kg ingot using the FAR GWPs. The percentage increase in regional primary ingot GHG intensities caused by the change to TAR GWPs for CF_4 and C_2F_6 are shown in Table B.7.

Table B.7 Percentage Increase in Primary Ingot GHG Intensity from TAR GWPs for CF_4 and C_2F_6

	Africa	N. America	L. America	Asia	Europe	Oceania	World
1990	5.4%	6.1%	6.4%	3.9%	5.9%	3.4%	5.4%
1991	5.2%	6.0%	6.4%	3.7%	6.1%	3.3%	5.3%
1992	5.1%	5.5%	6.8%	3.5%	6.2%	3.3%	5.1%
1993	4.9%	5.2%	6.8%	3.5%	6.2%	2.3%	4.8%
1994	4.7%	5.0%	6.7%	3.5%	6.2%	1.7%	4.6%
1995	4.2%	5.1%	5.9%	3.3%	5.7%	1.2%	4.3%
1996	2.0%	5.0%	6.1%	3.4%	6.3%	1.1%	4.4%
1997	2.9%	4.8%	6.2%	3.1%	5.5%	0.8%	4.0%
1998	2.3%	4.4%	5.9%	3.1%	5.4%	1.0%	3.8%
1999	2.7%	4.4%	5.6%	3.2%	5.3%	0.7%	3.7%
2000	2.5%	4.1%	5.6%	3.2%	5.1%	0.7%	3.6%
2001	2.5%	2.7%	5.1%	3.2%	4.8%	1.0%	3.3%
2002	2.5%	2.8%	6.1%	3.4%	4.7%	0.9%	3.3%
2003	2.3%	2.3%	6.1%	3.3%	4.7%	0.9%	3.2%
2004	2.1%	2.2%	6.1%	3.4%	4.3%	0.9%	3.2%
2005	2.2%	2.3%	6.1%	3.5%	5.0%	0.9%	3.3%

B.6 Trade Data Mass Balance Correction for Alloying Materials

Due to differences in the reporting practices between countries, USGS production data are often presented using more than one convention. In general, data are reported on the mass of poured aluminum without the addition of alloying elements and aluminum scrap. There are instances, however, where data include the mass of these additional materials. At the same time, trade data report the combined mass of non-alloyed and alloyed primary ingot. According to the most recent LCI report, approximately 62 kg (5.5%) of alloying metals and outside scrap are used per 1,117 kg of final ingot.

Upon review of the USGS production data and the UN ComTrade bilateral trade data, the bilateral trade data were more likely to include the mass of alloying materials and outside scrap in the reported mass of primary ingot. The percentage of alloying materials and outside scrap was reported as 5.5% by the IAI in the 2007 LCI report, but these materials were not included in the LCI report from 2000. The mass of imports, exports, and inventory adjustments were adjusted downwards using this value. Ingot production data reported by the USGS were not adjusted. The correction changed the calculated consumption-weighted emissions intensities by an average of -0.2% and by no more than 2%.

B.7 Primary Ingot Flow Data

Table B.8 Imports and Exports (United Nations Statistics Division 2007), Production (USGS 2006), Inventory Change (IAI 2007), and Apparent Consumption of Primary Aluminum Ingot by Region (metric tons)

		Percentage of Imports from Exporting Region								
Importing Region	Period	Africa	N. America	L. America	Asia	Europe	Oceania	Total Imports	Total Exports	Production
Africa	1990	na	23%	26%	0%	50%	1%	14,400	144,700	605,000
	1991	na	18%	36%	0%	39%	8%	20,200	106,700	605,000
	1992	na	29%	38%	0%	31%	2%	16,600	140,900	613,000
	1993	na	19%	20%	0%	59%	1%	15,100	102,600	615,000
	1994	na	3%	18%	2%	76%	0%	13,200	154,100	590,000
	1995	na	17%	10%	2%	70%	1%	14,200	121,100	623,000
	1996	na	0%	24%	9%	68%	0%	9,500	132,900	968,000
	1997	na	15%	15%	6%	56%	9%	11,700	119,000	1,094,000
	1998	na	4%	7%	19%	70%	0%	14,700	103,800	1,030,000
	1999	na	0%	3%	1%	95%	0%	54,500	113,500	1,092,000
	2000	na	0%	43%	6%	51%	0%	12,600	583,700	1,172,000
	2001	na	1%	44%	11%	43%	1%	12,200	548,200	1,348,000
	2002	na	0%	30%	9%	52%	9%	9,100	569,600	1,354,000
	2003	na	1%	44%	2%	47%	6%	17,400	525,300	1,434,800
	2004	na	0%	28%	1%	62%	9%	17,600	647,700	1,713,900
	2005	na	1%	22%	4%	44%	29%	15,300	723,700	1,753,600
N. America	1990	0%	na	90%	0%	6%	4%	149,100	1,098,000	5,688,000
	1991	0%	na	80%	1%	18%	2%	92,700	1,398,100	5,991,000
	1992	0%	na	75%	1%	24%	0%	150,300	1,168,100	6,035,000
	1993	0%	na	27%	1%	73%	0%	712,200	1,128,300	6,003,000
	1994	0%	na	31%	1%	68%	0%	1,104,100	922,800	5,554,000

Importing Region	Period	Percentage of Imports from Exporting Region: Africa	N. America	L. America	Asia	Europe	Oceania	Total Imports	Total Exports	Production
	1995	0%	na	34%	1%	65%	0%	713,000	981,700	5,557,000
	1996	0%	na	27%	1%	73%	0%	623,000	937,600	5,921,000
	1997	0%	na	28%	3%	68%	0%	708,200	805,400	5,996,000
	1998	0%	na	22%	3%	68%	7%	1,055,300	691,200	6,149,000
	1999	0%	na	23%	3%	68%	6%	1,319,600	538,500	6,232,000
	2000	2%	na	27%	3%	66%	2%	1,274,700	552,300	6,102,000
	2001	2%	na	35%	1%	47%	14%	996,800	512,100	5,271,200
	2002	1%	na	33%	2%	55%	9%	1,315,900	491,900	5,454,492
	2003	1%	na	37%	0%	54%	8%	1,245,400	473,400	5,495,243
	2004	2%	na	32%	1%	58%	6%	1,663,000	349,600	5,108,522
	2005	5%	na	33%	5%	51%	6%	1,715,100	346,800	5,375,203
L. America	1990	0%	62%	na	0%	38%	0%	1,500	1,177,000	1,719,000
	1991	0%	92%	na	0%	8%	0%	6,900	1,226,100	1,935,000
	1992	0%	75%	na	1%	25%	0%	3,800	1,305,700	1,948,000
	1993	0%	55%	na	0%	45%	0%	12,100	1,238,500	1,943,000
	1994	0%	79%	na	1%	20%	0%	16,100	1,285,900	1,972,000
	1995	0%	70%	na	0%	30%	0%	26,700	1,436,400	2,032,000
	1996	0%	86%	na	0%	14%	0%	21,700	1,276,100	2,040,000
	1997	0%	89%	na	0%	11%	0%	18,000	1,202,000	2,053,000
	1998	0%	81%	na	0%	19%	0%	17,000	1,121,200	2,009,000
	1999	0%	78%	na	0%	22%	0%	11,100	1,243,100	2,032,000
	2000	0%	61%	na	0%	38%	0%	9,200	1,274,100	2,102,000
	2001	19%	54%	na	0%	27%	0%	10,600	1,155,200	1,955,632
	2002	0%	54%	na	0%	45%	0%	5,700	1,287,000	2,192,495
	2003	0%	47%	na	0%	53%	0%	6,000	1,314,000	2,254,259
	2004	5%	21%	na	0%	74%	0%	7,600	1,257,800	2,352,988
	2005	0%	16%	na	3%	82%	0%	9,400	1,313,300	2,379,500
Asia	1990	0%	32%	24%	na	4%	39%	2,712,800	8,200	2,013,000
	1991	0%	37%	23%	na	5%	35%	2,886,000	11,900	2,288,000
	1992	0%	27%	25%	na	12%	36%	2,899,200	21,300	2,901,000
	1993	0%	22%	22%	na	19%	37%	2,954,300	35,400	3,038,000
	1994	0%	17%	24%	na	22%	36%	3,019,600	12,800	3,281,000
	1995	0%	17%	27%	na	27%	29%	3,665,200	25,700	3,570,000
	1996	0%	17%	19%	na	29%	35%	3,380,900	12,900	3,592,000
	1997	0%	12%	19%	na	33%	36%	3,421,500	37,100	3,900,000
	1998	0%	11%	14%	na	27%	48%	2,775,400	75,700	4,266,000
	1999	0%	8%	16%	na	35%	40%	3,314,200	74,600	4,632,000
	2000	10%	7%	11%	na	38%	34%	4,218,100	78,700	5,090,500
	2001	9%	7%	10%	na	34%	41%	3,635,700	47,200	5,580,111
	2002	11%	5%	8%	na	33%	42%	3,462,500	48,600	6,731,342
	2003	11%	7%	12%	na	30%	41%	3,765,700	36,300	8,130,703
	2004	12%	6%	10%	na	34%	38%	3,926,500	45,800	9,643,543
	2005	10%	6%	10%	na	34%	40%	3,846,700	134,700	11,134,940
Europe	1990	19%	28%	51%	1%	na	0%	756,600	134,400	7,782,000
	1991	11%	36%	52%	1%	na	0%	921,900	171,700	7,267,000
	1992	14%	38%	46%	2%	na	0%	982,200	384,900	6,480,000
	1993	10%	48%	39%	3%	na	0%	993,900	1,085,600	6,516,000

Importing Region	Period	Percentage of Imports from Exporting Region: Africa	N. America	L. America	Asia	Europe	Oceania	Total Imports	Total Exports	Production
	1994	20%	51%	28%	1%	na	0%	752,800	1,439,200	6,239,000
	1995	17%	49%	31%	3%	na	0%	694,700	1,491,600	6,327,000
	1996	14%	37%	48%	1%	na	0%	959,100	1,455,600	6,534,000
	1997	14%	44%	41%	2%	na	0%	857,500	1,633,900	6,734,000
	1998	10%	35%	48%	4%	na	3%	1,046,000	1,485,700	7,211,000
	1999	13%	31%	51%	4%	na	0%	797,600	2,109,800	7,604,000
	2000	16%	28%	52%	4%	na	0%	901,500	2,464,000	7,832,000
	2001	21%	27%	44%	3%	na	5%	990,600	1,721,800	8,013,325
	2002	15%	28%	53%	2%	na	2%	1,079,600	1,876,500	8,181,568
	2003	13%	28%	52%	4%	na	3%	802,100	1,831,700	8,466,566
	2004	22%	16%	54%	3%	na	5%	647,500	2,328,300	8,823,871
	2005	30%	14%	47%	5%	na	4%	794,400	2,182,000	8,969,728
Oceania	1990	0%	12%	8%	6%	74%	na	2,000	1,074,100	1,490,000
	1991	0%	22%	2%	0%	75%	na	4,200	1,017,400	1,488,000
	1992	0%	10%	3%	2%	85%	na	5,300	1,036,300	1,483,000
	1993	0%	24%	4%	17%	55%	na	4,900	1,102,300	1,658,000
	1994	0%	10%	2%	2%	86%	na	5,000	1,095,800	1,586,000
	1995	0%	22%	2%	2%	75%	na	3,700	1,061,000	1,570,000
	1996	0%	68%	2%	13%	17%	na	2,400	1,181,600	1,655,000
	1997	0%	66%	3%	3%	28%	na	2,600	1,222,000	1,805,000
	1998	0%	44%	0%	6%	49%	na	2,500	1,433,300	1,945,000
	1999	0%	42%	0%	12%	46%	na	2,100	1,419,700	2,045,000
	2000	0%	38%	1%	32%	28%	na	1,800	1,465,000	2,097,000
	2001	0%	6%	1%	87%	6%	na	4,200	1,665,600	2,119,359
	2002	0%	12%	0%	61%	27%	na	1,700	1,600,900	2,170,982
	2003	0%	16%	1%	73%	10%	na	3,800	1,659,800	2,197,499
	2004	0%	3%	0%	92%	4%	na	5,300	1,638,400	2,244,400
	2005	0%	10%	1%	84%	5%	na	3,600	1,684,100	2,254,400

Importing Region	Period	Inventory Change	Apparent Consumption
Africa	1990	-20,000	494,700
	1991	22,000	496,500
	1992	-20,000	508,700
	1993	4,000	523,500
	1994	-8,000	457,100
	1995	58,000	458,100
	1996	-33,000	877,700
	1997	-5,000	991,700
	1998	17,000	923,900
	1999	-21,000	1,054,000
	2000	26,000	574,900
	2001	22,000	790,400
	2002	-46,000	839,400
	2003	-23,000	949,900
	2004	28,000	1,055,900
	2005	19,000	1,026,200
N.	1990	-26,000	4,765,100

Importing Region	Period	Inventory Change	Apparent Consumption
America			
	1991	7,000	4,678,600
	1992	-31,000	5,048,200
	1993	240,000	5,347,000
	1994	-28,000	5,763,300
	1995	-139,000	5,427,300
	1996	-149,000	5,755,400
	1997	-7,000	5,905,800
	1998	-82,000	6,595,100
	1999	-156,000	7,169,200
	2000	-146,000	6,970,400
	2001	-129,000	5,885,000
	2002	25,000	6,253,500
	2003	-76,000	6,343,300
	2004	10,000	6,411,900
	2005	-21,000	6,764,500
L. America	1990	-38,000	581,500
	1991	47,000	668,700
	1992	-40,000	686,100
	1993	29,000	687,700
	1994	-24,000	726,200
	1995	26,000	596,400
	1996	-26,000	811,600
	1997	-6,000	875,100
	1998	48,000	856,800
	1999	-37,000	837,000
	2000	-4,000	841,100
	2001	27,000	784,000
	2002	14,000	897,200
	2003	-9,000	955,200
	2004	1,000	1,101,800
	2005	3,000	1,072,600
Asia	1990	123,000	4,594,600
	1991	185,000	4,977,100
	1992	195,000	5,583,800
	1993	161,000	5,795,900
	1994	118,000	6,169,800
	1995	158,000	7,051,500
	1996	173,000	6,787,000
	1997	168,000	7,116,300
	1998	168,000	6,797,700
	1999	158,000	7,713,600
	2000	283,000	8,946,900
	2001	336,000	8,832,600
	2002	257,000	9,888,200
	2003	301,000	11,559,200
	2004	320,000	13,204,300
	2005	422,000	14,425,000

Importing Region	Period	Inventory Change	Apparent Consumption
Europe	1990	3,000	8,401,200
	1991	92,000	7,925,200
	1992	16,000	7,061,300
	1993	7,000	6,417,300
	1994	151,000	5,401,600
	1995	-20,000	5,550,200
	1996	-210,000	6,247,500
	1997	17,000	5,940,600
	1998	40,000	6,731,300
	1999	164,000	6,127,700
	2000	7,000	6,262,500
	2001	-16,000	7,298,100
	2002	-4,000	7,388,700
	2003	95,000	7,342,000
	2004	125,000	7,018,000
	2005	-104,000	7,686,200
Oceania	1990	-10,000	427,900
	1991	-19,000	493,800
	1992	-1,000	452,900
	1993	7,000	553,700
	1994	-4,000	499,100
	1995	22,000	490,600
	1996	-26,000	501,900
	1997	31,000	554,600
	1998	-25,000	539,200
	1999	8,000	619,400
	2000	-25,000	658,700
	2001	-29,000	486,900
	2002	10,000	561,800
	2003	2,000	539,500
	2004	26,000	585,300
	2005	8,000	565,900

B.8 Time Series GHG Emissions and Emissions Intensities

Table B.9 Primary Aluminum Ingot Production Life Cycle GHG Emissions by Region (Mt CO_2-eq)

Year	Africa	N. America	L. America	Asia	Europe	Oceania	World
1990	7.45	89.8	12.7	44.1	98.3	31.1	283
1991	7.42	89.2	14.1	50.1	84.1	31.3	276
1992	7.51	86.4	13.4	62.0	69.8	31.0	270
1993	7.64	80.5	13.3	64.0	68.2	32.0	266
1994	7.50	71.1	13.6	68.2	63.7	29.1	253
1995	8.74	71.9	13.5	76.2	67.0	28.3	266
1996	15.8	75.8	14.3	77.2	68.6	30.2	282
1997	18.3	75.2	14.7	84.3	67.8	33.5	294
1998	21.0	74.5	15.0	94.5	71.4	38.5	315
1999	19.0	68.7	16.1	100	74.0	39.6	318

2000	20.8	72.0	16.4	108	76.3	39.7	334
2001	23.5	60.9	16.7	121	81.2	40.8	344
2002	23.4	65.1	15.6	146	80.9	41.7	374
2003	26.5	63.5	16.0	182	81.2	42.0	412
2004	32.8	57.6	16.8	217	88.5	40.9	454
2005	31.5	57.6	16.8	243	74.6	43.6	468

Table B. 10 Primary Aluminum Ingot Production Life Cycle GHG Emission Intensities by Region (kg CO_2-eq/kg)

Year	Africa	N. America	L. America	Asia	Europe	Oceania	World
1990	12.3	15.8	7.36	21.9	12.6	20.9	14.7
1991	12.3	15.0	7.27	21.9	11.6	21.0	14.1
1992	12.3	14.3	6.88	21.4	10.8	20.9	13.9
1993	12.4	13.4	6.83	21.1	10.5	19.3	13.4
1994	12.7	12.8	6.89	20.8	10.2	18.3	13.2
1995	14.0	13.0	6.63	21.3	10.6	18.0	13.5
1996	16.3	12.8	7.03	21.5	10.5	18.3	13.6
1997	16.7	12.6	7.18	21.6	10.1	18.6	13.6
1998	20.4	12.1	7.48	22.2	9.90	19.8	13.9
1999	17.4	11.0	7.94	21.7	9.72	19.4	13.4
2000	17.7	11.8	7.79	21.3	9.75	18.9	13.7
2001	17.4	11.6	8.53	21.7	10.1	19.2	14.2
2002	17.3	12.0	7.14	21.8	9.88	19.2	14.3
2003	18.4	11.6	7.12	22.4	9.59	19.1	14.7
2004	19.1	11.3	7.13	22.5	10.0	18.2	15.2
2005	18.0	10.7	7.07	21.9	8.31	19.3	14.7

Table B.11 Primary Aluminum Ingot Export Life Cycle GHG Emissions by Region (kg CO_2-eq)

Year	Africa	N. America	L. America	Asia	Europe	Oceania
1990	1.80E+09	1.75E+10	8.78E+09	1.82E+08	1.73E+09	2.25E+10
1991	1.32E+09	2.11E+10	9.04E+09	2.64E+08	2.03E+09	2.15E+10
1992	1.75E+09	1.69E+10	9.11E+09	4.61E+08	4.24E+09	2.17E+10
1993	1.29E+09	1.53E+10	8.59E+09	7.55E+08	1.15E+10	2.14E+10
1994	1.98E+09	1.20E+10	8.99E+09	2.70E+08	1.49E+10	2.02E+10
1995	1.72E+09	1.29E+10	9.66E+09	5.56E+08	1.61E+10	1.92E+10
1996	2.19E+09	1.22E+10	9.10E+09	2.80E+08	1.55E+10	2.16E+10
1997	2.01E+09	1.02E+10	8.77E+09	8.11E+08	1.67E+10	2.28E+10
1998	2.13E+09	8.48E+09	8.51E+09	1.70E+09	1.49E+10	2.85E+10
1999	1.99E+09	6.02E+09	1.00E+10	1.63E+09	2.08E+10	2.76E+10
2000	1.04E+10	6.61E+09	1.01E+10	1.69E+09	2.44E+10	2.78E+10
2001	9.61E+09	6.00E+09	9.97E+09	1.04E+09	1.78E+10	3.21E+10
2002	9.91E+09	5.95E+09	9.31E+09	1.07E+09	1.89E+10	3.08E+10
2003	9.77E+09	5.54E+09	9.49E+09	8.22E+08	1.79E+10	3.18E+10
2004	1.25E+10	4.00E+09	9.10E+09	1.04E+09	2.37E+10	2.99E+10
2005	1.31E+10	3.77E+09	9.43E+09	2.97E+09	1.85E+10	3.27E+10

Table B. 12 Primary Aluminum Ingot Import Life Cycle GHG Emissions by Region (kg CO_2-eq)

Year	Africa	N. America	L. America	Asia	Europe	Oceania
1990	1.75E+08	1.23E+09	2.21E+07	4.27E+10	8.24E+09	2.66E+07
1991	2.32E+08	7.73E+08	1.02E+08	4.41E+10	9.97E+09	5.27E+07
1992	1.76E+08	1.20E+09	5.24E+07	4.21E+10	1.06E+10	6.04E+07
1993	1.60E+08	6.85E+09	1.48E+08	4.06E+10	1.10E+10	6.44E+07
1994	1.33E+08	1.02E+10	2.01E+08	3.91E+10	8.51E+09	5.41E+07
1995	1.56E+08	6.67E+09	3.31E+08	4.49E+10	7.99E+09	4.19E+07
1996	1.02E+08	6.04E+09	2.75E+08	4.43E+10	1.02E+10	3.32E+07
1997	1.34E+08	6.86E+09	2.24E+08	4.45E+10	9.56E+09	3.17E+07
1998	1.81E+08	1.11E+10	2.02E+08	4.07E+10	1.20E+10	2.98E+07
1999	5.43E+08	1.36E+10	1.21E+08	4.51E+10	8.64E+09	2.54E+07
2000	1.22E+08	1.27E+10	1.03E+08	5.79E+10	1.01E+10	2.53E+07
2001	1.33E+08	1.12E+10	1.32E+08	5.29E+10	1.21E+10	8.53E+07
2002	1.00E+08	1.35E+10	6.36E+07	5.09E+10	1.14E+10	3.03E+07
2003	1.65E+08	1.19E+10	6.37E+07	5.43E+10	8.76E+09	7.41E+07
2004	1.80E+08	1.67E+10	8.26E+07	5.57E+10	7.47E+09	1.16E+08
2005	1.82E+08	1.69E+10	8.70E+07	5.35E+10	9.66E+09	7.25E+07

Table B. 13 Primary Aluminum Ingot Import Life Cycle GHG Emissions Intensity by Region (kg CO_2-eq/kg)

Year	Africa	N. America	L. America	Asia	Europe	Oceania
1990	12.1	8.26	14.8	15.8	10.9	13.5
1991	11.5	8.34	14.8	15.3	10.8	12.5
1992	10.6	7.99	13.6	14.5	10.8	11.5
1993	10.6	9.62	12.3	13.7	11.1	13.1
1994	10.0	9.28	12.5	13.0	11.3	10.9
1995	11.0	9.35	12.4	12.2	11.5	11.4
1996	10.7	9.70	12.7	13.1	10.7	13.6
1997	11.5	9.68	12.4	13.0	11.2	12.2
1998	12.4	10.6	11.9	14.7	11.4	11.9
1999	9.96	10.3	10.9	13.6	10.8	11.9
2000	9.69	10.0	11.2	13.7	11.2	14.4
2001	10.9	11.2	12.4	14.5	12.2	20.4
2002	11.1	10.2	11.2	14.7	10.6	17.6
2003	9.47	9.59	10.7	14.4	10.9	19.5
2004	10.2	10.1	10.9	14.2	11.5	21.7
2005	11.9	9.87	9.22	13.9	12.2	20.1

Table B. 14 Primary Aluminum Ingot Consumption Life Cycle GHG Emissions by Region (Mt CO_2-eq)

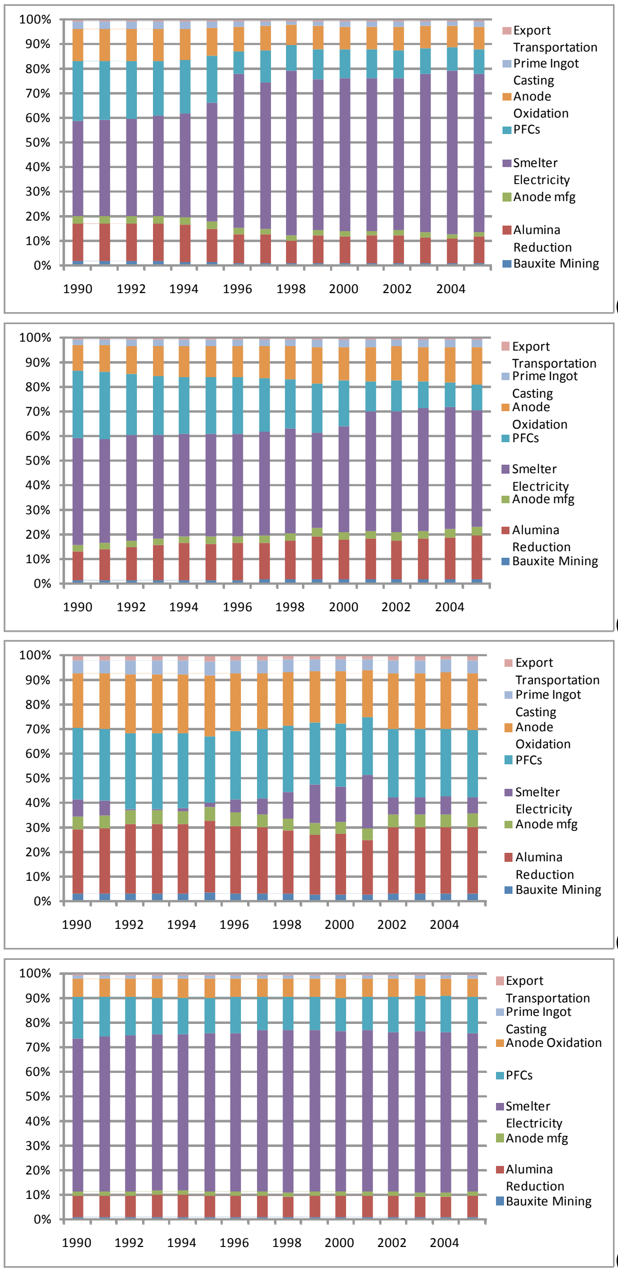
Year	Africa	N. America	L. America	Asia	Europe	Oceania
1990	6.07E+09	7.39E+10	4.18E+09	8.40E+10	1.05E+11	8.86E+09
1991	6.06E+09	6.91E+10	4.79E+09	8.99E+10	9.10E+10	1.03E+10
1992	6.18E+09	7.13E+10	4.61E+09	9.94E+10	7.60E+10	9.35E+09
1993	6.46E+09	6.90E+10	4.64E+09	1.00E+11	6.76E+10	1.06E+10
1994	5.75E+09	6.99E+10	4.96E+09	1.05E+11	5.58E+10	9.06E+09
1995	6.36E+09	6.77E+10	3.98E+09	1.17E+11	5.92E+10	8.76E+09

1996	1.42E+10	7.18E+10	5.68E+09	1.17E+11	6.55E+10	9.09E+09
1997	1.65E+10	7.20E+10	6.23E+09	1.24E+11	6.05E+10	1.02E+10
1998	1.87E+10	7.83E+10	6.36E+09	1.30E+11	6.80E+10	1.06E+10
1999	1.79E+10	7.81E+10	6.55E+09	1.40E+11	6.02E+10	1.19E+10
2000	1.00E+10	7.99E+10	6.46E+09	1.58E+11	6.19E+10	1.24E+10
2001	1.36E+10	6.77E+10	6.62E+09	1.66E+11	7.57E+10	9.30E+09
2002	1.44E+10	7.24E+10	6.31E+09	1.91E+11	7.35E+10	1.07E+10
2003	1.73E+10	7.08E+10	6.68E+09	2.29E+11	7.12E+10	1.02E+10
2004	1.99E+10	7.02E+10	7.74E+09	2.64E+11	7.10E+10	1.06E+10
2005	1.82E+10	7.10E+10	7.47E+09	2.85E+11	6.66E+10	1.09E+10

Table B.15 Primary Aluminum Ingot Consumption Life Cycle GHG Emissions Intensity by Region (Mt CO_2-eq/kg)

Year	Africa	N. America	L. America	Asia	Europe	Oceania
1990	12.3	15.5	7.18	18.3	12.5	20.7
1991	12.2	14.8	7.16	18.1	11.5	20.8
1992	12.2	14.1	6.72	17.8	10.8	20.7
1993	12.3	12.9	6.75	17.3	10.5	19.1
1994	12.6	12.2	6.83	16.9	10.3	18.1
1995	13.9	12.5	6.67	16.6	10.7	17.8
1996	16.2	12.5	7.00	17.3	10.5	18.1
1997	16.6	12.2	7.12	17.5	10.2	18.4
1998	20.2	11.9	7.43	19.1	10.1	19.6
1999	17.0	10.9	7.82	18.2	9.82	19.2
2000	17.4	11.5	7.68	17.7	9.89	18.8
2001	17.2	11.5	8.44	18.8	10.4	19.1
2002	17.1	11.6	7.03	19.3	9.94	19.1
2003	18.2	11.2	7.00	19.8	9.69	19.0
2004	18.9	11.0	7.03	20.0	10.1	18.1
2005	17.8	10.5	6.96	19.7	8.66	19.2

B.9 Breakdown of Contributions to Regional GHG Intensities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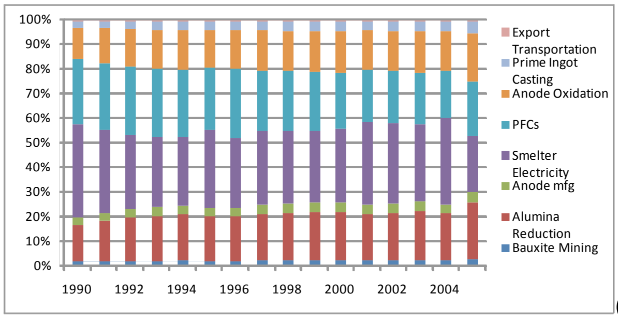
90%
80%
70%
60%
50%
40%
30%
20%
10%
0%
1990
1992
1994
1996
1998
2000
2002
2004
Export Transportation
Prime Ingot Casting
Anode Oxidation
PFCs
Smelter Electricity
Anode mfg
Alumina Reduction
Bauxite Mining

(a)

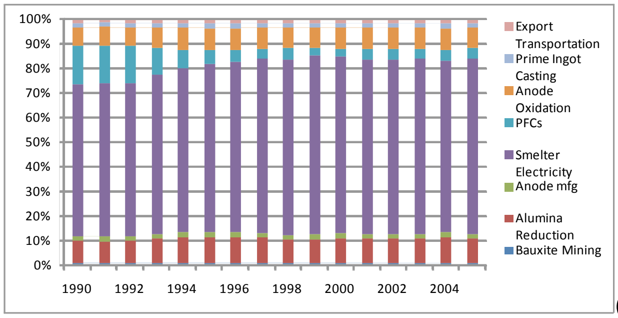
(b)

(c)

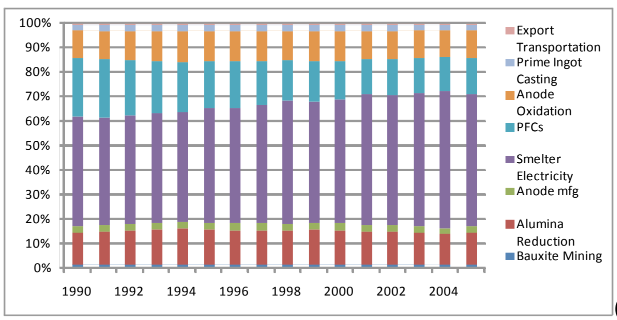
(d)



(e)



(f)



(g)

Figure B.3 Source Contribution to GHG Intensity for Africa (a), North America (b), Latin America (c), Asia (d), Europe (e), Oceania (f), and the Global Average (g)

B.10 Summary of Primary Aluminum Ingot GHG Emissions Intensities from Common LCA Databases

Table B.16 Summary of Primary Aluminum Ingot GHG Emissions Intensities from Common LCA Databases

Database	Value (kg CO2-eq/kg)	Production- or Consumption-Based	Geographic Representation	Data Year
ETH (19)	13.7	Production	Europe	1996
BUWAL 250 (1993)	11.0	Production	Europe	1993
Ecoinvent (2007)	12.2	Production	Europe	2007
IDEMAT 2001 (Delft University of Technology 2001)	13.3	Production	Europe	1996

B.12 Discussion of Additional Trends Affecting Variation in Primary Aluminum GHG Emissions

B.12.1 Portion of Electricity Purchased and Self-Generated

As major users of electricity, primary aluminum smelters may generate their own electricity and make purchases from the grid. The fraction of electricity that is self-generated and purchased differs by region, but the global average has remained around 27% self-generated (73% purchased) during 1990 – 2005. In China, which does not appear in the global average, approximately 18% of primary aluminum capacity has dedicated electricity.

Electricity fuel mix data are reported on world and regional levels for both self-generated and purchased electricity consumption. On the global scale, there is an evolving picture of the fuel sources for self-generated electricity. The fraction of self-generated hydroelectricity has been in a downward trend, decreasing steadily from approximately 65% in 1990 to 46% in 2005. Concurrently, the fraction of electricity generated from natural gas has increased, while the fraction of coal-fired electricity has remained relatively constant at approximately 27%.

Different fuel mix trends are observed in purchased electricity. The fraction of hydroelectricity has increased between 1990 and 2005, from approximately 52.7% to 61%. From 1990 to 2000, the portion of hydroelectricity remained relatively constant, within the range of 53% and 59%. The portion then dropped from 56% to 49% between 2000 and 2001 before increasing to its 2005 level. Concurrently, the fraction of electricity generated from coal has decreased from 37.4% in 1990 to 28% in 2005. The peak fraction occurred in 2001, when coal-fired generation accounted for nearly 40% of purchased electricity.

These global trends indicate that the GHG intensity of electricity generation has been increasing for self-generated electricity and decreasing for purchased electricity. In the case of self-generated electricity, the increase in GHG intensity is being driven by the drop in the fraction of hydroelectricity and the rise in electricity generated from natural gas. Regarding purchased electricity, the decrease in GHG intensity has been shaped by the use of a greater fraction of hydropower and a lower fraction of coal-fired generation.

B.12.2 Primary Aluminum Smelter Capacity

On average, global primary aluminum capacity has kept pace with the increase in production. Estimated capacity increased from 12.163 million metric tons in 1990 to 34.877 million metric tons in 2005. As would be expected given the discussion of production trends, the largest increase in capacity occurred in Asia. Between 1990 and 2005 Asia added over 10 million metric tons of primary aluminum capacity, an increase of nearly 425%. Throughout this period, Asian aluminum production ran at approximately 90% of capacity.

It is estimated that the next largest increase in capacity occurred in Europe. Since data for 1990 are not available for the entire region, the year 1995 was used to gain a sense of the trends in capacity. From 1995 to 2005 Europe added 1.976 million metric tons to existing capacity, an increase of 28%. During this same time period, European primary ingot production ran at an average of 98% of capacity.

Capacity in Africa increased by 1.457 million metric tons from 1990 to 2005. This represents a rise of nearly 230% and marks Africa as the region to add the third largest amount to its existing capacity since 1990. On average, Africa has produced primary ingot at 87% of capacity from 1990 to 2005. The region maintained production at over 90% of capacity from 1990 through 1994, but has not met this level since.

North America added the fourth largest amount to existing capacity, 972,000 metric tons, from 1990 to 2005. This represents the smallest percentage increase in capacity of any region. The years of 1999 to 2005 have seen the region producing at an average of 77% of capacity. Additionally, for the entire 1990-2005 period, Africa is the only region to operate at a lower percentage of capacity than North America (73%). These capacity trends, along with the trends in production, are an indication of North America's diminishing role in global primary ingot production.

B.12.3 Reliance on Imported Ingot

In order to provide a measure of a region's dependence on primary ingot produced in other countries, the annual percentage of apparent consumption due to domestically produced ingot is calculated. This metric is derived by subtracting the amount of imported ingot from apparent consumption and then dividing by apparent consumption.

The major primary ingot consuming regions, Asia, Europe, and North America, each exhibit different trends in this measurement. It is estimated that in 1990, North America produced nearly 98% of the primary ingot it consumed. By 2005, however, this figure had decreased to approximately 75%.

Asia, conversely, is increasingly meeting its apparent consumption with domestically produced primary ingot. In 1990, approximately 43% of the aluminum consumed in the region was domestically produced; by 2005, domestic production satisfied over 71% of Asia's apparent consumption.

Since 1990, Europe has showed only moderate changes in this metric. The region's domestic production accounted for over 90% of apparent consumption from 1990 through 1993, but then decreased and remained around 86% for the subsequent years. The three remaining regions, Africa, Latin America, and Oceania, which all relatively small consumers of ingot, consistently exhibited values of over 90% for the 1990 – 2005 period.

The exploration and in-depth discussion of the inverted U hypothesis of primary ingot consumption is not within the scope of this paper, it will be noted that North America and Europe, whose constituent nations are largely mature, serviced-based economies, exhibited the smallest intensities of use from 1990 to 2005, each averaging around 950 kg primary ingot per million US$. Interestingly, it was calculated that Oceania consumed an average of approximately 1,500 kg primary ingot per million US$ from 1990 to 2000, before decreasing in the years following. This level of intensity of use is similar to what was observed in regions consisting of mostly emerging economies. In keeping with the rapid expansion of the emerging economies in Asia, the intensity of use for the region climbed from 1,178 kg primary ingot/million US$ in 1990 to 2,397 kg primary ingot/million US$ in 2005.

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APPENDIX C

SUPPLEMENTAL MATERIAL: CHAPTER 4

C.1 Comparison of Factors Relating to Markov Chain Analysis of Iron and Steel, Aluminum, and Copper Recycling

Table C6.1 Average Product Lifetime by End-Use Category (years)

Metal	Region /Country	Build & Const	Trans	Consumer Durables	Infra-structure	Machinery	Containers	Other
Copper	N. America	40	15	10	50	20	-	-
	L. America	50	17	15	35	30	-	-
	Europe	35	13	8	30	12	-	-
	Middle East	35	15	12	30	30	-	-
	Africa	50	20	15	50	30	-	-
	Asia	35	15	12	35	25	-	-
	CIS	40	13	15	50	25	-	-
Steel	Japan	30	13	-	-	12	1	12
Aluminum	U.S.	40	13-30	15	35	25	1	40
	Germany	31.5	13	10	17.5	15	1	31.5
	China	32.5	17	16	21.5	17.5	1	32.5

Sources: Copper (Ruhrberg 2006; Spatari *et al*. 2005; van Beers and Graedel 2007); steel (Toi and Sato 1997); and aluminum (Bruggink 2000; Melo 1999; Xiong 2005)

Table C6.2 U.S. Consumption by End-Use Category in 2003

Metal	Build & Const	Service Centers	Trans	Consumer Durables	Electrical	Machinery	Containers	Other
Iron & steel	22%	27%	15%	-	-	-	3%	33%
Copper	48%	-	10%	11%	21%	10%	-	-
Aluminum	16%	-	36%	7%	7%	6%	23%	4%

Sources: iron and steel (USGS 2005); copper (USGS 2005) and aluminum (USGS 2005).

Table C6.3 End-of-Life Old Scrap Recovery by End-Use Category

Metal	Region /Country	Build & Const	Trans	Machinery	Containers	Consumer Durables	Other
Steel	Japan	50%	90%	80%	92%	-	80%
Aluminum	U.S.	15%	30-80%	15%	25-60%	20%	20%
Aluminum	Germany	85%	90%	80%	40%	20%	NA

Sources: steel (Diago *et al*. 2005) and aluminum (Bruggink 2000; Rink 1994).

C.2 Backing Data for Figures and Analyses

Table C6.4 Backing Data for Figure 4.1

Data	Sec Smelter Old Scrap Consumption	Sec Smelter UBC Consumption	All Others Old Scrap Consumption	All Others UBC Consumption	Total Scrap Consumption Less UBCs
Source	USGS (2009)	USGS (2009)	USGS (2009)	USGS (2009)	Own Calculation

Data	Sec Smelter Old Scrap Consumption	Sec Smelter UBC Consumption	All Others Old Scrap Consumption	All Others UBC Consumption	Total Scrap Consumption Less UBCs
Source	USGS (2009)	USGS (2009)	USGS (2009)	USGS (2009)	Own Calculation
Year					
1960	80,204	NA	1,933	NA	82,137
1961	61,760	NA	956	NA	62,716
1962	76,002	NA	2,910	NA	78,912
1963	81,199	NA	3,158	NA	84,357
1964	91,256	NA	3,751	NA	95,007
1965	111,924	NA	6,489	NA	118,413
1966	96,798	NA	3,205	NA	100,003
1967	94,141	NA	3,646	NA	97,787
1968	103,335	NA	7,685	NA	111,019
1969	107,400	NA	8,511	NA	115,911
1970	113,985	NA	9,506	NA	123,491
1971	107,412	NA	17,716	NA	125,128
1972	106,813	NA	30,924	NA	137,737
1973	101,424	NA	58,605	NA	160,029
1974	113,345	NA	64,308	NA	177,653
1975	112,632	6,303	111,277	69,755	147,851
1976	144,124	7,413	138,229	91,295	183,645
1977	167,442	14,447	178,829	105,902	225,922
1978	187,387	24,673	201,597	126,979	237,333
1979	221,420	26,101	215,152	137,158	273,313
1980	260,423	84,530	261,759	186,940	250,712
1981	335,767	160,720	378,130	300,407	252,771
1982	285,337	100,384	487,731	411,020	261,664
1983	280,282	92,857	533,221	428,032	292,614
1984	272,891	79,932	551,471	462,848	281,582
1985	344,339	178,083	466,551	413,182	219,625
1986	268,524	75,527	503,608	448,858	247,747
1987	267,557	58,787	581,105	496,185	293,690
1988	419,771	122,924	653,146	544,600	405,393
1989	399,494	109,401	665,938	568,594	387,437
1990	588,493	198,865	862,389	687,792	564,225
1991	411,890	103,018	1,002,943	766,264	545,551
1992	589,885	97,775	1,166,916	863,423	795,603
1993	670,352	100,812	1,099,676	811,190	858,026

Data Source	Sec Smelter Old Scrap Consumption	Sec Smelter UBC Consumption	All Others Old Scrap Consumption	All Others UBC Consumption	Total Scrap Consumption Less UBCs
Source	USGS (2009)	USGS (2009)	USGS (2009)	USGS (2009)	Own Calculation
1994	443,363	100,426	1,195,659	849,124	689,472
1995	496,700	118,000	1,144,910	847,216	676,394
1996	542,000	69,500	1,170,000	851,063	791,438
1997	549,000	88,100	1,120,000	915,083	665,817
1998	724,000	133,000	910,000	783,545	717,455
1999	643,000	119,000	1,050,000	919,511	654,489
2000	582,000	88,000	918,000	789,376	622,624
2001	510,000	94,300	823,000	697,362	541,338
2002	506,000	93,600	785,000	664,920	532,480
2003	429,000	62,600	759,000	660,839	464,561
2004	470,000	68,700	778,000	716,003	463,297
2005	399,000	44,900	755,000	657,000	452,100
2006	498,000	40,600	796,000	660,000	593,400
2007	488,000	43,300	1,240,000	653,000	1,031,700

Data	Index of Total Scrap Consumption Less UBCs
Source	Own Calculation
Year	
1960	100
1961	76
1962	96
1963	103
1964	116
1965	144
1966	122
1967	119
1968	135
1969	141
1970	150
1971	152
1972	168
1973	195

Data	Index of Total Scrap Consumption Less UBCs
Source	Own Calculation
1974	216
1975	180
1976	224
1977	275
1978	289
1979	333
1980	305
1981	308
1982	319
1983	356
1984	343
1985	267
1986	302
1987	358
1988	494
1989	472
1990	687
1991	664
1992	969
1993	1045
1994	839
1995	823
1996	964
1997	811
1998	873
1999	797
2000	758
2001	659
2002	648
2003	566
2004	564
2005	550
2006	722
2007	1256

Table C6.5 Backing Data for Figure 4.1, continued

Data	Total Shipments	Total Shipments to Containers & Pkg	Total Shipments Less Cont & Pkg	Index of Total Shipments Less Cont & Pkg
Source	AA (2008b)	AA (2008b)	Own Calculation	Own Calculation
Year				
1960	2,146,421	145,605	2,000,816	100
1961	2,254,377	159,213	2,095,165	105
1962	2,618,616	171,913	2,446,702	122
1963	2,892,588	224,984	2,667,604	133
1964	3,252,744	260,365	2,992,380	150
1965	3,695,909	298,013	3,397,895	170
1966	4,095,074	335,662	3,759,412	188
1967	4,057,425	395,537	3,661,889	183
1968	4,524,177	467,205	4,056,972	203
1969	4,909,281	543,409	4,365,871	218
1970	4,585,412	668,148	3,917,264	196
1971	4,728,749	689,921	4,038,828	202
1972	5,457,679	825,093	4,632,586	232
1973	6,551,755	938,039	5,613,717	281
1974	6,121,292	1,034,201	5,087,091	254
1975	4,491,518	913,544	3,577,973	179
1976	5,681,303	1,172,548	4,508,754	225
1977	6,069,128	1,261,000	4,808,128	240
1978	6,665,155	1,424,748	5,240,406	262
1979	6,669,691	1,461,943	5,207,747	260
1980	6,399,800	1,512,292	4,887,508	244
1981	6,118,570	1,593,033	4,525,538	226
1982	5,495,328	1,618,434	3,876,894	194
1983	6,410,687	1,777,193	4,633,494	232
1984	6,836,614	1,828,450	5,008,165	250
1985	6,924,159	1,862,016	5,062,143	253
1986	6,937,766	1,926,427	5,011,340	250
1987	7,437,630	2,052,073	5,385,557	269
1988	7,624,059	2,036,197	5,587,862	279
1989	7,815,023	2,115,577	5,699,447	285
1990	7,796,426	2,164,565	5,631,861	281
1991	7,705,706	2,228,069	5,477,638	274
1992	8,057,244	2,269,346	5,787,898	289

Data Source	Total Shipments	Total Shipments to Containers & Pkg	Total Shipments Less Cont & Pkg	Index of Total Shipments Less Cont & Pkg
	AA (2008b)	AA (2008b)	Own Calculation	Own Calculation
1993	8,386,102	2,193,595	6,192,507	309
1994	9,342,284	2,273,428	7,068,856	353
1995	9,534,156	2,307,902	7,226,254	361
1996	9,595,845	2,175,451	7,420,394	371
1997	10,211,830	2,220,357	7,991,472	399
1998	10,518,461	2,273,428	8,245,033	412
1999	11,188,424	2,316,066	8,872,358	443
2000	11,111,313	2,264,356	8,846,956	442
2001	10,210,469	2,250,295	7,960,174	398
2002	10,770,208	2,258,460	8,511,748	425
2003	10,651,365	2,241,223	8,410,142	420
2004	11,317,246	2,312,438	9,004,808	450
2005	11,587,136	2,320,149	9,266,987	463
2006	11,780,368	2,319,242	9,461,127	473
2007	11,166,652	2,220,357	8,946,294	447

Table S6.6 Backing Data for Figure 4.2

Data Source	Total Domestic Shipments	Total Domestic Shipments Less Cont & Pkg	Total Domestic Ingot Shipments	Ingot Shipments as % of Total Domestic Shipments Less Cont & Pkg
	AA (2008b)	Own Calculation	AA (2008b)	Own Calculation
Year				
1960	1,866,552	1,720,947	470,380	27%
1961	2,106,505	1,947,292	579,697	30%
1962	2,430,826	2,258,913	704,890	31%
1963	2,679,851	2,454,867	771,115	31%
1964	2,988,751	2,728,386	820,103	30%
1965	3,436,905	3,138,891	873,628	28%
1966	3,831,534	3,495,872	889,504	25%
1967	3,762,134	3,366,597	937,585	28%
1968	4,233,421	3,766,216	1,058,242	28%
1969	4,455,684	3,912,274	1,070,035	27%
1970	4,062,869	3,394,720	804,681	24%
1971	4,475,642	3,785,721	997,913	26%
1972	5,207,294	4,382,201	1,094,530	25%

Data	Total Domestic Shipments	Total Domestic Shipments Less Cont & Pkg	Total Domestic Ingot Shipments	Ingot Shipments as % of Total Domestic Shipments Less Cont & Pkg
Source	AA (2008b)	Own Calculation	AA (2008b)	Own Calculation
1973	6,132,178	5,194,140	1,362,152	26%
1974	5,700,807	4,666,606	1,161,208	25%
1975	4,133,176	3,219,632	907,194	28%
1976	5,308,900	4,136,351	1,091,354	26%
1977	5,745,714	4,484,714	1,157,126	26%
1978	6,326,318	4,901,569	1,236,052	25%
1979	6,205,661	4,743,718	1,246,938	26%
1980	5,405,062	3,892,770	941,214	24%
1981	5,497,142	3,904,110	1,005,171	26%
1982	4,907,920	3,289,486	856,845	26%
1983	5,847,773	4,070,580	1,012,429	25%
1984	6,348,544	4,520,094	1,172,548	26%
1985	6,378,028	4,516,012	1,169,827	26%
1986	6,524,540	4,598,113	1,145,332	25%
1987	6,868,366	4,816,293	1,166,652	24%
1988	6,837,522	4,801,325	1,209,743	25%
1989	6,755,420	4,639,844	1,146,240	25%
1990	6,639,300	4,474,735	1,204,300	27%
1991	6,348,544	4,120,475	1,183,888	29%
1992	6,821,646	4,552,300	1,350,358	30%
1993	7,294,747	5,101,152	1,650,186	32%
1994	8,146,603	5,873,174	1,870,634	32%
1995	8,233,693	5,925,792	2,008,528	34%
1996	8,306,722	6,131,271	2,094,711	34%
1997	8,855,575	6,635,217	2,317,427	35%
1998	9,234,328	6,960,900	2,489,794	36%
1999	9,842,602	7,526,535	2,682,119	36%
2000	9,833,983	7,569,627	2,798,694	37%
2001	9,310,533	7,060,238	2,707,521	38%
2002	9,666,606	7,408,147	2,903,021	39%
2003	9,755,511	7,514,288	2,944,298	39%
2004	10,387,825	8,075,388	2,996,462	37%
2005	10,461,762	8,141,613	2,885,331	35%
2006	10,508,936	8,189,694	2,866,733	35%
2007	9,712,419	7,492,062	2,775,560	37%

Data	Ingot Shipments to Transportation
Source	AA (2008b)
Year	
1960	187,789
1961	233,149
1962	293,024
1963	328,858
1964	343,373
1965	402,341
1966	383,289
1967	382,382
1968	425,928
1969	421,392
1970	326,136
1971	405,062
1972	464,030
1973	590,583
1974	450,875
1975	376,032
1976	521,183
1977	571,532
1978	594,666
1979	611,902
1980	435,453
1981	461,308
1982	402,794
1983	531,616
1984	644,108
1985	641,840
1986	640,025
1987	665,880
1988	694,911
1989	638,211
1990	703,983
1991	717,137
1992	857,298

Data	Ingot Shipments to Transportation
Source	AA (2008b)
1993	1,102,241
1994	1,255,557
1995	1,411,594
1996	1,481,901
1997	1,676,948
1998	1,823,914
1999	1,993,559
2000	2,090,629
2001	1,947,292
2002	2,148,236
2003	2,185,430
2004	2,256,192
2005	2,190,874
2006	2,132,360
2007	1,988,116

Table C6.7 Backing Data for Figure 4.3

Data	Sheet and Plate (Flat Rolled) Shipments to Building & Construction	Mill Product Shipments to Trucks & Buses	Mill Product Shipments to Passenger Cars	Ingot Shipments to Trucks and Buses	Ingot Shipments to Passenger Cars
Source	AA (2008b)	AA (2008b)	AA (2008b)	AA (2008b)	AA (2008b)
Year					
1960	264,447	NA	NA	NA	NA
1961	270,344	NA	NA	NA	NA
1962	292,116	NA	NA	NA	NA
1963	337,023	NA	NA	NA	NA
1964	362,878	NA	NA	NA	NA
1965	405,062	NA	NA	NA	NA
1966	433,185	NA	NA	NA	NA
1967	395,083	28,577	65,318	34,927	294,838
1968	474,009	39,463	76,204	43,545	318,879
1969	529,348	43,092	80,287	51,710	310,714
1970	517,554	35,834	68,493	39,463	258,097

Data Source	Sheet and Plate (Flat Rolled) Shipments to Building & Construction	Mill Product Shipments to Trucks & Buses	Mill Product Shipments to Passenger Cars	Ingot Shipments to Trucks and Buses	Ingot Shipments to Passenger Cars
	AA (2008b)	AA (2008b)	AA (2008b)	AA (2008b)	AA (2008b)
1971	615,531	49,896	88,451	55,792	317,064
1972	735,734	64,864	115,214	61,236	360,610
1973	871,360	100,245	151,048	82,101	452,690
1974	756,600	90,266	123,378	72,576	326,136
1975	520,729	34,020	102,967	39,917	293,931
1976	718,044	89,359	181,439	76,658	401,887
1977	731,198	109,770	246,757	86,637	438,628
1978	793,795	113,399	265,354	90,266	454,958
1979	724,848	127,461	230,881	97,977	457,226
1980	567,450	68,493	139,254	61,689	338,383
1981	540,234	73,483	151,501	67,132	354,259
1982	464,483	55,339	117,028	68,493	300,281
1983	646,829	78,926	169,192	83,008	415,948
1984	587,862	115,214	196,861	96,163	478,998
1985	612,810	99,338	194,140	90,719	503,039
1986	635,489	107,049	195,954	91,627	499,410
1987	681,303	120,203	209,108	93,895	523,451
1988	586,501	122,018	219,087	102,059	543,863
1989	586,955	110,224	198,222	95,255	495,328
1990	575,161	106,142	189,150	89,359	566,089
1991	509,843	87,544	166,924	78,472	596,026
1992	546,131	103,874	195,047	96,163	712,601
1993	583,326	136,986	240,406	131,543	920,802
1994	696,725	170,552	306,178	168,738	1,030,572
1995	566,543	179,171	336,115	171,913	1,179,806
1996	642,293	166,017	379,207	167,831	1,256,917
1997	614,624	176,449	411,413	186,882	1,421,119
1998	670,416	190,964	420,031	190,057	1,562,188
1999	725,755	224,984	499,410	199,583	1,720,947
2000	694,457	196,408	505,307	189,604	1,832,986
2001	657,262	146,965	483,081	161,934	1,727,751
2002	730,745	166,017	519,369	181,439	1,908,283
2003	739,817	171,006	537,512	188,696	1,927,787
2004	811,031	226,799	571,532	192,779	1,995,827

Data	Sheet and Plate (Flat Rolled) Shipments to Building & Construction	Mill Product Shipments to Trucks & Buses	Mill Product Shipments to Passenger Cars	Ingot Shipments to Trucks and Buses	Ingot Shipments to Passenger Cars
Source	AA (2008b)	AA (2008b)	AA (2008b)	AA (2008b)	AA (2008b)
2005	770,208	265,354	573,347	217,727	1,905,108
2006	736,188	283,498	565,635	222,263	1,809,852
2007	629,139	212,283	547,945	234,510	1,693,278

Data	Car and Truck Total Shipments
Source	Own Calculation
Year	
1960	NA
1961	NA
1962	NA
1963	NA
1964	NA
1965	NA
1966	NA
1967	423,660
1968	478,091
1969	485,802
1970	401,887
1971	511,204
1972	601,923
1973	786,084
1974	612,356
1975	470,834
1976	749,342
1977	881,793
1978	923,977
1979	913,544
1980	607,820
1981	646,376
1982	541,141
1983	747,074

Data	Car and Truck Total Shipments
Source	Own Calculation
1984	887,236
1985	887,236
1986	894,040
1987	946,657
1988	987,027
1989	899,029
1990	950,739
1991	928,967
1992	1,107,684
1993	1,429,738
1994	1,676,041
1995	1,867,005
1996	1,969,972
1997	2,195,863
1998	2,363,240
1999	2,644,924
2000	2,724,304
2001	2,519,731
2002	2,775,107
2003	2,825,002
2004	2,986,936
2005	2,961,535
2006	2,881,248
2007	2,688,016

Table C6.8 Backing Data for Figure 4.3, continued

Data	Secondary Smelter Total Scrap Consumption	Secondary Smelter Total Scrap Consumption Less UBCs	All Others Total Scrap Consumption	All Others Total Scrap Consumption Less UBCs
Source	USGS (2009)	Own Calculation	USGS (2009)	Own Calculation
Year				

Data Source	Secondary Smelter Total Scrap Consumption	Secondary Smelter Total Scrap Consumption Less UBCs	All Others Total Scrap Consumption	All Others Total Scrap Consumption Less UBCs
	USGS (2009)	Own Calculation	USGS (2009)	Own Calculation
1960	321,043	321,043	79,460	79,460
1961	300,918	300,918	150,966	150,966
1962	401,128	401,128	138,532	138,532
1963	488,113	488,113	139,999	139,999
1964	488,965	488,965	157,178	157,178
1965	526,026	526,026	214,800	214,800
1966	579,471	579,471	233,670	233,670
1967	559,865	559,865	240,994	240,994
1968	634,384	634,384	286,472	286,472
1969	673,238	673,238	332,918	332,918
1970	589,967	589,967	292,303	292,303
1971	580,516	580,516	331,004	331,004
1972	640,912	640,912	404,558	404,558
1973	668,431	668,431	476,834	476,834
1974	571,729	571,729	522,391	522,391
1975	554,780	548,477	563,012	493,257
1976	688,546	681,133	640,899	549,604
1977	763,176	748,730	659,484	553,582
1978	800,769	776,097	665,708	538,729
1979	836,569	810,468	710,210	573,051
1980	802,183	717,653	734,255	547,315
1981	885,728	725,008	939,911	639,504
1982	806,331	705,948	988,886	577,866
1983	821,346	728,489	1,090,664	662,631
1984	792,907	712,975	1,105,751	642,903
1985	858,722	680,639	1,024,822	611,640
1986	808,869	733,342	1,073,686	624,828
1987	803,188	744,401	1,292,883	796,698
1988	871,454	748,530	1,363,328	818,728
1989	804,995	695,594	1,368,994	800,400
1990	982,011	783,146	1,579,366	891,574
1991	673,531	570,513	1,782,795	1,016,531
1992	945,627	847,852	2,015,047	1,151,624
1993	1,134,840	1,034,028	2,051,574	1,240,384

Data Source	Secondary Smelter Total Scrap Consumption	Secondary Smelter Total Scrap Consumption Less UBCs	All Others Total Scrap Consumption	All Others Total Scrap Consumption Less UBCs
	USGS (2009)	Own Calculation	USGS (2009)	Own Calculation
1994	1,150,786	1,050,360	2,185,321	1,336,197
1995	1,297,040	1,179,040	2,183,710	1,336,494
1996	1,430,000	1,360,500	2,230,000	1,378,938
1997	1,750,000	1,661,900	2,370,000	1,454,917
1998	2,080,000	1,947,000	1,920,000	1,136,455
1999	2,180,000	2,061,000	2,130,000	1,210,489
2000	1,960,000	1,872,000	1,970,000	1,180,624
2001	1,640,000	1,545,700	1,740,000	1,042,638
2002	1,680,000	1,586,400	1,690,000	1,025,080
2003	1,520,000	1,457,400	1,700,000	1,039,161
2004	1,630,000	1,561,300	1,760,000	1,043,997
2005	1,550,000	1,505,100	1,750,000	1,093,000
2006	2,060,000	2,019,400	1,930,000	1,270,000
2007	1840000	1,796,700	2490000	1,837,000

Table C6.9 Backing Data for Figure 4.4

Data Source	Secondary Smelter New Scrap Consumption	Secondary Smelter Old Scrap Consumption	Secondary Smelter UBC Scrap Consumption	Secondary Smelter Old Scrap Consumption Less UBCs
	USGS (2009)	USGS (2009)	USGS (2009)	Own Calculation
1967	427,221	94,141	na	94,141
1968	527,317	103,335	na	103,335
1969	518,343	107,400	na	107,400
1970	433,006	113,985	na	113,985
1971	420,356	107,412	na	107,412
1972	469,427	106,813	na	106,813
1973	516,368	101,424	na	101,424
1974	401,193	113,345	na	113,345
1975	383,192	112,632	6,303	106,329
1976	469,937	144,124	7,413	136,711
1977	507,885	167,442	14,447	152,995
1978	530,285	187,387	24,673	162,714
1979	549,184	221,420	26,101	195,320
1980	484,762	260,423	84,530	175,893

Data	Secondary Smelter New Scrap Consumption	Secondary Smelter Old Scrap Consumption	Secondary Smelter UBC Scrap Consumption	Secondary Smelter Old Scrap Consumption Less UBCs
Source	USGS (2009)	USGS (2009)	USGS (2009)	Own Calculation
1981	490,340	335,767	160,720	175,048
1982	457,421	285,337	100,384	184,953
1983	472,061	280,282	92,857	187,425
1984	459,333	272,891	79,932	192,959
1985	432,168	344,339	178,083	166,256
1986	473,346	268,524	75,527	192,997
1987	476,765	267,557	58,787	208,770
1988	410,048	419,771	122,924	296,847
1989	384,894	399,494	109,401	290,093
1990	377,018	588,493	198,865	389,628
1991	255,344	411,890	103,018	308,872
1992	353,948	589,885	97,775	492,110
1993	462,877	670,352	100,812	569,540
1994	704,842	443,363	100,426	342,937
1995	796,000	496,700	118,000	378,700
1996	885,000	542,000	69,500	472,500
1997	1,190,000	549,000	88,100	460,900
1998	1,350,000	724,000	133,000	591,000
1999	1,520,000	643,000	119,000	524,000
2000	1,370,000	582,000	88,000	494,000
2001	1,120,000	510,000	94,300	415,700
2002	1,170,000	506,000	93,600	412,400
2003	1,090,000	429,000	62,600	366,400
2004	1,150,000	470,000	68,700	401,300
2005	1,140,000	399,000	44,900	354,100
2006	1,550,000	498,000	40,600	457,400
2007	1,350,000	488,000	43,300	444,700

Table C6.10 Backing Data for Figure 4.5

Data	Imports: Metals and Alloys, Crude	Estimated Fraction Alloys	Exports: Metals and Alloys, Crude	Estimated Fraction of Alloys	Estimated Net Imports of Alloys
Source	USGS (2009)	Own Calculation	USGS (2009)	Own Calculation	Own Calculation
Year					
1960	140,348	50%	258,531	30%	-7,385
1961	180,734	50%	116,902	30%	55,296
1962	282,097	50%	137,165	30%	99,899
1963	377,092	50%	149,995	30%	143,547

Data	Imports: Metals and Alloys, Crude	Estimated Fraction Alloys	Exports: Metals and Alloys, Crude	Estimated Fraction of Alloys	Estimated Net Imports of Alloys
Source	USGS (2009)	Own Calculation	USGS (2009)	Own Calculation	Own Calculation
1964	357,945	50%	189,261	30%	122,194
1965	478,320	50%	184,743	30%	183,737
1966	472,667	50%	170,770	30%	185,103
1967	407,980	50%	189,612	30%	147,106
1968	622,062	50%	163,548	30%	261,967
1969	424,781	50%	312,450	30%	118,655
1970	317,572	50%	370,545	30%	47,623
1971	502,774	50%	101,873	30%	220,825
1972	599,693	50%	98,266	30%	270,367
1973	460,877	50%	208,272	30%	167,957
1974	461,438	50%	188,541	30%	174,157
1975	393,830	50%	168,602	30%	146,334
1976	521,954	50%	138,226	30%	219,509
1977	608,001	50%	88,697	30%	277,392
1978	686,829	50%	114,866	30%	308,955
1979	517,676	50%	182,028	30%	204,229
1980	526,640	50%	648,558	30%	68,752
1981	644,703	50%	312,221	30%	228,685
1982	616,325	50%	363,943	30%	198,980
1983	673,765	50%	327,229	30%	238,714
1984	881,956	50%	259,598	30%	363,099
1985	868,674	50%	347,292	30%	330,149
1986	1,348,816	50%	209,794	30%	611,470
1987	1,245,638	50%	218,816	30%	557,174
1988	1,027,246	50%	400,057	30%	393,606
1989	923,030	NA	593,103	NA	NA
1990	959,615	NA	679,803	NA	NA
1991	1,024,732	NA	792,794	NA	NA
1992	1,155,515	NA	603,818	NA	NA
1993	1,840,000	NA	400,000	NA	NA
1994	2,480,000	NA	339,000	NA	NA
1995	1,930,000	NA	369,000	NA	NA
1996	1,910,000	NA	417,000	NA	NA
1997	2,060,000	NA	352,000	NA	NA
1998	2,400,000	NA	265,000	NA	NA

Data	Imports: Metals and Alloys, Crude	Estimated Fraction Alloys	Exports: Metals and Alloys, Crude	Estimated Fraction of Alloys	Estimated Net Imports of Alloys
Source	USGS (2009)	Own Calculation	USGS (2009)	Own Calculation	Own Calculation
1999	2,650,000	NA	318,000	NA	NA
2000	2,490,000	NA	273,000	NA	NA
2001	2,560,000	NA	192,000	NA	NA
2002	2,790,000	NA	206,000	NA	NA
2003	2,870,000	NA	214,000	NA	NA
2004	3,250,000	NA	298,000	NA	NA
2005	3,660,000	NA	329,000	NA	NA
2006	3,440,000	NA	346,000	NA	NA
2007	2,950,000	NA	349,000	NA	NA

Data	Estimated Net Import of Unalloyed (Primary)
Source	Own Calculation
Year	
1960	-110,798
1961	8,536
1962	45,033
1963	83,549
1964	46,490
1965	109,840
1966	116,794
1967	71,262
1968	196,547
1969	-6,325
1970	-100,595
1971	180,076
1972	231,060
1973	84,648
1974	98,740
1975	78,894
1976	164,219
1977	241,913
1978	263,008
1979	131,418

Data	Estimated Net Import of Unalloyed (Primary)
Source	Own Calculation
1980	-190,671
1981	103,797
1982	53,403
1983	107,822
1984	259,259
1985	191,233
1986	527,552
1987	469,648
1988	233,583
1989	NA
1990	NA
1991	NA
1992	NA
1993	NA
1994	NA
1995	NA
1996	NA
1997	NA
1998	NA
1999	NA
2000	NA
2001	NA
2002	NA
2003	NA
2004	NA
2005	NA
2006	NA
2007	NA

Table C6.11 Backing Data for Figure 4.5, continued

Data	Net Imports: Unwrought, Unalloyed (HTS 7601.10)	Net Imports: Unwrought Alloys (HTS 7601.20)	Primary Production	Net Imports of Plates, Sheet, Bars, Etc.	New Scrap Consumption at Primary Producers, Etc.
Source	U.S. ITC (2009)	U.S. ITC (2009)	USGS (2009)	USGS (2009)	USGS (2009)
Year					
1960	NA	NA	1,827,000	32,533	77,527
1961	NA	NA	1,727,000	44,821	85,481
1962	NA	NA	1,921,000	54,010	91,461
1963	NA	NA	2,098,000	37,466	114,206
1964	NA	NA	2,316,000	45,659	136,574
1965	NA	NA	2,498,000	59,160	182,258
1966	NA	NA	2,693,000	112,837	209,610
1967	NA	NA	2,966,000	53,361	218,149
1968	NA	NA	2,953,000	62,899	251,702
1969	NA	NA	3,441,000	55,047	308,065
1970	NA	NA	3,607,000	74,062	272,391
1971	NA	NA	3,561,000	66,218	288,393
1972	NA	NA	3,739,000	72,860	347,669
1973	NA	NA	4,109,000	53,659	391,145
1974	NA	NA	4,448,000	41,756	436,398
1975	NA	NA	3,519,000	55,451	428,794
1976	NA	NA	3,856,000	79,287	487,069
1977	NA	NA	4,118,000	68,447	463,721
1978	NA	NA	4,358,000	209,571	442,605
1979	NA	NA	4,557,000	182,349	470,810
1980	NA	NA	4,654,000	66,109	454,604
1981	NA	NA	4,489,000	128,913	543,498
1982	NA	NA	3,274,000	194,237	485,260
1983	NA	NA	3,353,000	335,975	544,785
1984	NA	NA	4,099,000	460,369	542,398
1985	NA	NA	3,500,000	423,825	534,550
1986	NA	NA	3,037,000	458,867	552,169
1987	NA	NA	3,343,000	415,705	688,963
1988	NA	NA	3,944,000	392,237	685,379
1989	-58,861	343,687	4,030,000	340,360	680,052
1990	-70,841	248,384	4,048,000	336,189	699,547
1991	-118,366	252,219	4,121,000	256,884	765,883

Data Source	Net Imports: Unwrought, Unalloyed (HTS 7601.10)	Net Imports: Unwrought Alloys (HTS 7601.20)	Primary Production	Net Imports of Plates, Sheet, Bars, Etc.	New Scrap Consumption at Primary Producers, Etc.
	U.S. ITC (2009)	U.S. ITC (2009)	USGS (2009)	USGS (2009)	USGS (2009)
1992	141,269	375,635	4,042,000	309,179	838,793
1993	825,940	605,327	3,695,000	401,000	940,617
1994	1,393,504	747,876	3,299,000	510,000	979,455
1995	978,877	566,062	3,375,000	631,000	1,028,500
1996	846,238	668,421	3,577,000	498,000	1,050,000
1997	1,025,308	708,319	3,603,000	566,000	1,240,000
1998	1,240,269	882,819	3,713,000	649,000	1,000,000
1999	1,219,920	1,187,720	3,779,000	735,000	1,070,000
2000	971,268	1,291,483	3,668,000	795,000	1,050,000
2001	1,339,257	1,005,116	2,637,000	683,000	917,000
2002	1,345,186	1,258,364	2,707,000	804,000	902,000
2003	1,587,851	1,138,539	2,703,000	820,000	941,000
2004	1,712,787	1,282,705	2,516,000	935,000	979,000
2005	1,956,784	1,377,913	2,481,000	1,188,000	992,000
2006	1,599,062	1,486,104	2,284,000	1,213,000	1,140,000
2007	1,399,210	1,183,334	2554000	1,069,000	1250000

Data	UBC Consumption
Source	USGS (2009)
Year	
1960	NA
1961	NA
1962	NA
1963	NA
1964	NA
1965	NA
1966	NA
1967	NA
1968	NA
1969	NA
1970	NA

Data	UBC Consumption
Source	USGS (2009)
1971	NA
1972	NA
1973	NA
1974	NA
1975	76,058
1976	98,707
1977	120,349
1978	151,651
1979	163,259
1980	271,470
1981	461,127
1982	511,404
1983	520,889
1984	542,780
1985	591,265
1986	524,385
1987	554,972
1988	667,524
1989	677,995
1990	886,657
1991	869,283
1992	961,198
1993	912,002
1994	949,550
1995	965,216
1996	920,563
1997	1,003,183
1998	916,545
1999	1,038,511
2000	877,376
2001	791,662
2002	758,520
2003	723,439
2004	784,703

Data	UBC Consumption
Source	USGS (2009)
2005	701,900
2006	700,600
2007	696,300

Table C6.12 Backing Data for Figure 4.5, continued

Data	Estimated Total Consumption of Primary	Index of Estimated Total Consumption of Primary	New Scrap Consumption at Secondary Smelters	Sec Smelter Old Scrap Consumption	All Others Old Scrap Consumption
Source	Own Calculation	Own Calculation	USGS (2009)	USGS (2009)	USGS (2009)
Year					
1960	1,826,263	100	240,838	80,204	1,933
1961	1,865,838	102	218,486	61,760	956
1962	2,111,504	116	293,554	76,002	2,910
1963	2,333,221	128	373,852	81,199	3,158
1964	2,544,724	139	359,635	91,256	3,751
1965	2,849,258	156	364,636	111,924	6,489
1966	3,132,241	172	438,273	96,798	3,205
1967	3,308,772	181	427,221	94,141	3,646
1968	3,464,149	190	527,317	103,335	7,685
1969	3,797,787	208	518,343	107,400	8,511
1970	3,852,858	211	433,006	113,985	9,506
1971	4,095,687	224	420,356	107,412	17,716
1972	4,390,589	240	469,427	106,813	30,924
1973	4,638,453	254	516,368	101,424	58,605
1974	5,024,894	275	401,193	113,345	64,308
1975	4,158,196	228	383,192	112,632	111,277
1976	4,685,283	257	469,937	144,124	138,229
1977	5,012,430	274	507,885	167,442	178,829
1978	5,424,835	297	530,285	187,387	201,597
1979	5,504,836	301	549,184	221,420	215,152
1980	5,255,511	288	484,762	260,423	261,759
1981	5,726,335	314	490,340	335,767	378,130

Data	Estimated Total Consumption of Primary	Index of Estimated Total Consumption of Primary	New Scrap Consumption at Secondary Smelters	Sec Smelter Old Scrap Consumption	All Others Old Scrap Consumption
Source	Own Calculation	Own Calculation	USGS (2009)	USGS (2009)	USGS (2009)
1982	4,518,303	247	457,421	285,337	487,731
1983	4,862,472	266	472,061	280,282	533,221
1984	5,903,806	323	459,333	272,891	551,471
1985	5,240,873	287	432,168	344,339	466,551
1986	5,099,973	279	473,346	268,524	503,608
1987	5,472,288	300	476,765	267,557	581,105
1988	5,922,723	324	410,048	419,771	653,146
1989	5,669,546	310	384,894	399,494	665,938
1990	5,899,552	323	377,018	588,493	862,389
1991	5,894,683	323	255,344	411,890	1,002,943
1992	6,292,439	345	353,948	589,885	1,166,916
1993	6,774,559	371	462,877	670,352	1,099,676
1994	7,131,509	390	704,842	443,363	1,195,659
1995	6,978,592	382	796,000	496,700	1,144,910
1996	6,891,801	377	885,000	542,000	1,170,000
1997	7,437,491	407	1,190,000	549,000	1,120,000
1998	7,518,814	412	1,350,000	724,000	910,000
1999	7,842,431	429	1,520,000	643,000	1,050,000
2000	7,361,644	403	1,370,000	582,000	918,000
2001	6,367,919	349	1,120,000	510,000	823,000
2002	6,516,706	357	1,170,000	506,000	785,000
2003	6,775,289	371	1,090,000	429,000	759,000
2004	6,927,490	379	1,150,000	470,000	778,000
2005	7,319,684	401	1,140,000	399,000	755,000
2006	6,936,662	380	1,550,000	498,000	796,000
2007	6,968,510	382	1350000	488,000	1,240,000

Table C6.13 Backing Data for Figure 4.5, continued

Data	Estimated Total Consumption of Primary	Index of Estimated Total Consumption of Primary	New Scrap Consumption at Secondary Smelters	Sec Smelter Old Scrap Consumption	All Others Old Scrap Consumption
Source	Own Calculation	Own Calculation	USGS (2009)	USGS (2009)	USGS (2009)

Data	Estimated Total Consumption of Primary	Index of Estimated Total Consumption of Primary	New Scrap Consumption at Secondary Smelters	Sec Smelter Old Scrap Consumption	All Others Old Scrap Consumption
Source	Own Calculation	Own Calculation	USGS (2009)	USGS (2009)	USGS (2009)
Year					
1960	1,826,263	100	240,838	80,204	1,933
1961	1,865,838	102	218,486	61,760	956
1962	2,111,504	116	293,554	76,002	2,910
1963	2,333,221	128	373,852	81,199	3,158
1964	2,544,724	139	359,635	91,256	3,751
1965	2,849,258	156	364,636	111,924	6,489
1966	3,132,241	172	438,273	96,798	3,205
1967	3,308,772	181	427,221	94,141	3,646
1968	3,464,149	190	527,317	103,335	7,685
1969	3,797,787	208	518,343	107,400	8,511
1970	3,852,858	211	433,006	113,985	9,506
1971	4,095,687	224	420,356	107,412	17,716
1972	4,390,589	240	469,427	106,813	30,924
1973	4,638,453	254	516,368	101,424	58,605
1974	5,024,894	275	401,193	113,345	64,308
1975	4,158,196	228	383,192	112,632	111,277
1976	4,685,283	257	469,937	144,124	138,229
1977	5,012,430	274	507,885	167,442	178,829
1978	5,424,835	297	530,285	187,387	201,597
1979	5,504,836	301	549,184	221,420	215,152
1980	5,255,511	288	484,762	260,423	261,759
1981	5,726,335	314	490,340	335,767	378,130
1982	4,518,303	247	457,421	285,337	487,731
1983	4,862,472	266	472,061	280,282	533,221
1984	5,903,806	323	459,333	272,891	551,471
1985	5,240,873	287	432,168	344,339	466,551
1986	5,099,973	279	473,346	268,524	503,608
1987	5,472,288	300	476,765	267,557	581,105
1988	5,922,723	324	410,048	419,771	653,146
1989	5,669,546	310	384,894	399,494	665,938
1990	5,899,552	323	377,018	588,493	862,389
1991	5,894,683	323	255,344	411,890	1,002,943
1992	6,292,439	345	353,948	589,885	1,166,916

Data Source	Estimated Total Consumption of Primary	Index of Estimated Total Consumption of Primary	New Scrap Consumption at Secondary Smelters	Sec Smelter Old Scrap Consumption	All Others Old Scrap Consumption
	Own Calculation	Own Calculation	USGS (2009)	USGS (2009)	USGS (2009)
1993	6,774,559	371	462,877	670,352	1,099,676
1994	7,131,509	390	704,842	443,363	1,195,659
1995	6,978,592	382	796,000	496,700	1,144,910
1996	6,891,801	377	885,000	542,000	1,170,000
1997	7,437,491	407	1,190,000	549,000	1,120,000
1998	7,518,814	412	1,350,000	724,000	910,000
1999	7,842,431	429	1,520,000	643,000	1,050,000
2000	7,361,644	403	1,370,000	582,000	918,000
2001	6,367,919	349	1,120,000	510,000	823,000
2002	6,516,706	357	1,170,000	506,000	785,000
2003	6,775,289	371	1,090,000	429,000	759,000
2004	6,927,490	379	1,150,000	470,000	778,000
2005	7,319,684	401	1,140,000	399,000	755,000
2006	6,936,662	380	1,550,000	498,000	796,000
2007	6,968,510	382	1350000	488,000	1,240,000

Table C6.9 Backing Data for Figure 4.5, continued

Data Source	Estimated Total Consumption of Secondary	Index of Estimated Total Consumption of Secondary
	Own Calculation	Own Calculation
Year		
1960	315,591	100
1961	336,499	107
1962	472,365	150
1963	601,756	191
1964	576,836	183
1965	666,786	211
1966	723,378	229
1967	672,115	213
1968	900,304	285
1969	752,909	239
1970	604,120	191
1971	766,309	243

Data	Estimated Total Consumption of Secondary	Index of Estimated Total Consumption of Secondary
Source	Own Calculation	Own Calculation
1972	877,531	278
1973	844,354	268
1974	753,003	239
1975	753,436	239
1976	971,799	308
1977	1,131,547	359
1978	1,228,224	389
1979	1,189,985	377
1980	1,075,697	341
1981	1,432,922	454
1982	1,429,468	453
1983	1,524,278	483
1984	1,646,794	522
1985	1,573,207	498
1986	1,856,948	588
1987	1,882,601	597
1988	1,876,571	595
1989	1,794,013	568
1990	2,076,284	658
1991	1,922,397	609
1992	2,486,384	788
1993	2,838,232	899
1994	3,091,740	980
1995	3,003,672	952
1996	3,265,421	1035
1997	3,567,319	1130
1998	3,866,819	1225
1999	4,400,720	1394
2000	4,161,483	1319
2001	3,458,116	1096
2002	3,719,364	1179
2003	3,416,539	1083
2004	3,680,705	1166
2005	3,671,913	1164
2006	4,330,104	1372
2007	4,261,334	1350

Table C6.10 Data for New Scrap Analysis

Data	Apparent Consumption	Secondary Production from New Scrap
Source	USGS (2009)	USGS (2009)
Year		
1946	478,000	170,000
1947	639,000	164,000
1948	753,000	173,000
1949	667,000	124,000
1950	871,000	152,000
1951	876,000	196,000
1952	1,020,000	212,000
1953	1,360,000	263,000
1954	987,000	224,000
1955	1,460,000	285,000
1956	1,740,000	300,000
1957	1,460,000	315,000
1958	1,260,000	249,000
1959	1,930,000	313,000
1960	1,590,000	311,000
1961	1,720,000	299,000
1962	2,050,000	377,000
1963	2,280,000	449,000
1964	2,240,000	494,000
1965	2,850,000	566,000
1966	3,430,000	635,000
1967	3,200,000	638,000
1968	3,480,000	740,000
1969	3,710,000	862,000
1970	3,400,000	728,000
1971	3,830,000	757,000
1972	4,470,000	795,000
1973	5,280,000	886,000
1974	4,920,000	887,000
1975	3,540,000	816,000
1976	4,610,000	963,000

Data Source	Apparent Consumption	Secondary Production from New Scrap
	USGS (2009)	USGS (2009)
1977	4,980,000	974,000
1978	5,480,000	996,000
1979	5,340,000	1,060,000
1980	4,590,000	960,000
1981	4,620,000	1,030,000
1982	4,380,000	884,000
1983	5,030,000	953,000
1984	5,240,000	935,000
1985	5,210,000	912,000
1986	5,140,000	989,000
1987	5,470,000	1,130,000
1988	5,370,000	1,080,000
1989	4,960,000	1,040,000
1990	5,260,000	1,030,000
1991	5,040,000	969,000
1992	5,720,000	1,140,000
1993	6,610,000	1,310,000
1994	6,880,000	1,580,000
1995	6,300,000	1,680,000
1996	6,610,000	1,730,000
1997	6,720,000	2,020,000
1998	7,090,000	1,950,000
1999	7,770,000	2,120,000
2000	7,530,000	2,080,000
2001	6,230,000	1,760,000
2002	6,320,000	1,750,000
2003	6,130,000	1,750,000
2004	6,060,000	1,870,000
2005	5,990,000	1,950,000
2006	5,980,000	2,300,000
2007	4,214,000	2,250,000

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APPENDIX D

SUPPLEMENTAL MATERIAL: CHAPTER 5

D1 Aluminum Prices and Price Ratios

Table D1.1 Nominal Primary, Secondary Alloy, and Scrap Aluminum Prices (U.S. Cents per Lb)

Date	Prime (LME)	Alloy (AMM)	Alloy (LME)	New Scrap Clips (Chicago)	Old Scrap Sheet and Cast (Chicago)
J-85	48.8	64	na	33.68	26.68
F-85	49.8	63.11	na	34.7	27.7
M-85	49.7	60.62	na	34.29	27.5
A-85	50.2	58.41	na	33.5	26.36
M-85	50.1	56.41	na	31.95	24.5
J-85	46.8	54.3	na	31.4	23.53
J-85	45.9	51.98	na	30.5	22.02
A-85	46.2	53.02	na	31.59	23.27
S-85	44.7	54.75	na	32.5	23.85
O-85	44	55.78	na	32.39	23.93
N-85	43.1	55.97	na	30.47	22.47
D-85	47.1	56.05	na	31.21	22.43
J-86	50.77	59.16	na	35.18	24.36
F-86	50.56	63.5	na	36.61	26.61
M-86	52.98	66.88	na	39.02	29.64
A-86	52.82	67.32	na	40.91	30.91
M-86	52.81	60.93	na	37.17	25.4
J-86	53.66	55.19	na	34.74	22.62
J-86	50.94	56.09	na	36.5	23
A-86	51.22	58.17	na	36.79	24.67
S-86	54.7	59.57	na	37.79	26.62
O-86	52.71	59.98	na	37.45	24.93
N-86	51.34	56.42	na	36.05	24.05
D-86	51.4	54.71	na	35.5	23.55
J-87	53.1	56.7	na	37.1	25.75
F-87	58.2	59.11	na	38.61	27.56
M-87	62	62.11	na	41.5	30
A-87	63.5	63.18	na	43.64	31.64
M-87	64	65.23	na	46.15	33.1
J-87	66.8	69.32	na	48.78	34.82
J-87	75	72.64	na	51.05	36.2
A-87	82.1	77.45	na	55.64	40.24
S-87	80.7	79.81	na	56.5	41.55
O-87	89.1	82.7	na	60.36	43.68
N-87	76.7	84.37	na	56.87	41.87
D-87	83.3	84.11	na	57.55	42.83
J-88	91.2	87.1	na	60.65	45.5
F-88	98.2	90.05	na	64.35	47.4
M-88	114.6	95.98	na	73.89	52.67
A-88	113.8	102.52	na	74.74	55.64
M-88	137	100.36	na	76.21	53.93

Date	Prime (LME)	Alloy (AMM)	Alloy (LME)	New Scrap Clips (Chicago)	Old Scrap Sheet and Cast (Chicago)
J-88	164.8	103.09	na	79.73	55
J-88	122.6	103.2	na	79.65	54.65
A-88	125.2	101.07	na	80.5	54.85
S-88	109.8	100.48	na	79.98	54.69
O-88	106.5	97.11	na	73.98	50.64
N-88	110.4	96.5	na	73.66	49.5
D-88	113.5	99.95	na	74.69	50.5
J-89	108.7	101.43	na	75.5	53.02
F-89	99	102.29	na	72.97	52.97
M-89	94	100.33	na	72.11	53.41
A-89	96.4	97.75	na	70.8	51.9
M-89	102.5	97.48	na	70.68	51.14
J-89	86.8	94.09	na	67.36	47.59
J-89	79.6	86.2	na	58.61	42.97
A-89	81.6	81.86	na	53.67	40.63
S-89	77.9	83.43	na	57.15	43.3
O-89	82.5	81.95	na	54	40.14
N-89	78.7	78.33	na	53	37.75
D-89	74.1	73.5	na	51.65	35.65
J-90	69.3	72.2	na	50.32	34.68
F-90	66	69.86	na	43.5	30.5
M-90	71.1	74.13	na	44.25	30.77
A-90	69.2	82.25	na	49	32.5
M-90	69.3	80.82	na	49	32.5
J-90	71	77.45	na	49	32.5
J-90	71.3	76.4	na	50.29	35.93
A-90	80.8	79.58	na	51.5	37.5
S-90	93.7	79.36	na	55.08	41.97
O-90	88.3	77.66	na	55.5	42.5
N-90	73.4	73.53	na	51.65	40.58
D-90	69.1	71.61	na	48.5	39
J-91	68.8	72.44	na	45	33.5
F-91	68.3	70.04	na	43	32
M-91	67.9	69.39	na	43	32
A-91	63.2	69.3	na	41.18	30.64
M-91	58.8	65.23	na	38.64	28.45
J-91	57.9	62	na	33.4	22.4
J-91	58.8	63.89	na	33	21.5
A-91	57.1	65.25	na	33	21.5
S-91	54.9	64.8	na	32.7	21.13
O-91	52.2	61.08	na	31	19
N-91	51.5	60.09	na	29.21	18.11
D-91	49.8	58.35	na	29	18
J-92	53.4	58.44	na	29.71	18.71
F-92	57.5	61.91	na	33.68	23
M-92	58.1	67.14	na	37.39	26.91
A-92	59.8	69.52	na	37.5	27

Date	Prime (LME)	Alloy (AMM)	Alloy (LME)	New Scrap Clips (Chicago)	Old Scrap Sheet and Cast (Chicago)
M-92	59.3	69.1	na	35.38	24.45
J-92	57.9	65.98	na	35	24
J-92	59.6	65.64	na	35	24
A-92	59.2	66.7	na	35	24
S-92	57.6	64.65	na	33.14	21.21
O-92	53.3	61.07	na	32	19.5
N-92	52.6	60.57	na	32	19.5
D-92	54.8	64.07	na	32	19.5
J-93	54.75	67.97	na	22	19.5
F-93	54.53	66.85	na	22	19.5
M-93	52.24	66.15	na	22	19.5
A-93	50.29	64.73	na	22	19.5
M-93	51.00	63.02	na	22	19.5
J-93	52.87	62.88	na	22	19.5
J-93	54.55	64.47	na	23.64	20.73
A-93	53.18	64.53	na	24	21
S-93	50.61	62.56	na	23.48	20.48
O-93	49.33	60.32	na	23	20
N-93	47.18	59.26	na	23	20
D-93	49.65	61.57	na	23	20
J-94	53.29	65.21	na	23.95	21.9
F-94	57.62	71.63	na	26.98	25.82
M-94	58.48	75.42	na	29.63	28.33
A-94	58.02	75.77	na	34.5	32.5
M-94	60.01	75.15	na	34.5	32.5
J-94	63.55	78.49	na	35.18	33.18
J-94	67.72	81.81	na	37.79	35.79
A-94	66.03	81.58	na	38.5	36.5
S-94	71.20	85.4	na	43.05	41.05
O-94	77.04	89.26	na	46.45	44.45
N-94	85.87	97.14	na	52.77	50.77
D-94	85.22	98.94	na	55.5	53.5
J-95	93.49	103.93	na	59.9	56.3
F-95	86.94	100.86	na	58.08	54.08
M-95	81.90	95.07	na	53.02	49.02
A-95	83.78	93.58	na	52.5	48.5
M-95	80.00	87.77	na	48.41	44.41
J-95	80.76	86.73	na	46.5	42.5
J-95	84.39	90.24	na	49.03	43.76
A-95	81.91	93.75	na	51.8	45.8
S-95	79.82	89.31	na	51.05	45.05
O-95	75.97	82.51	na	44.84	39.27
N-95	75.05	78.84	na	43	37.5
D-95	75.22	80.62	na	43	37.5
J-96	72.28	80.02	na	43	37.5
F-96	72.21	77.13	na	43	37.5
M-96	73.16	79.1	na	43	37.5

Date	Prime (LME)	Alloy (AMM)	Alloy (LME)	New Scrap Clips (Chicago)	Old Scrap Sheet and Cast (Chicago)
A-96	72.09	80.12	na	43	37.5
M-96	72.12	79.45	na	40	37.5
J-96	67.26	76.96	na	40	37.5
J-96	66.18	75.89	na	40	37.5
A-96	66.39	75.83	na	40	36.5
S-96	63.85	75.33	na	39.9	35.4
O-96	60.63	71.93	na	38	33.5
N-96	65.77	75.71	na	38	33.5
D-96	68.07	78.72	na	38	33.5
J-97	71.49	81.71	na	40.59	36.09
F-97	71.69	84.83	na	41	36.5
M-97	74.02	87.14	na	41	36.5
A-97	70.84	86.24	na	41	36.5
M-97	73.74	87.65	na	41	36.5
J-97	71.12	86.76	na	41	36.5
J-97	72.23	85.67	na	39	36.5
A-97	77.62	87.59	na	45.07	39.36
S-97	73.07	86.44	na	47.5	38.6
O-97	72.95	86.59	na	47.5	38.5
N-97	72.55	85.6	na	47.5	38.5
D-97	69.44	82.51	na	46.21	38.07
J-98	67.41	80.04	na	44.5	37.5
F-98	66.50	79.8	na	44.5	37.5
M-98	65.23	78.44	na	44.5	37.5
A-98	64.43	76.73	na	42.95	35.95
M-98	61.91	74.82	na	42.5	35.5
J-98	59.31	69.45	na	39.07	31.59
J-98	59.40	64.91	na	33	25.5
A-98	59.48	63.94	na	33	25.5
S-98	60.90	64.27	na	33	25.5
O-98	59.17	64.27	na	33	25.5
N-98	58.75	64.44	na	33	25.5
D-98	56.67	65.05	na	33	25.5
J-99	55.28	65.92	na	33	25.5
F-99	53.85	67.88	na	33	25.5
M-99	53.61	68.18	na	32.52	25.5
A-99	57.99	70.22	na	32	25.5
M-99	60.05	74.23	na	33.9	26.45
J-99	59.68	76.5	na	34	29.23
J-99	63.69	76.41	na	34.67	36.83
A-99	64.94	77.19	na	36	37.5
S-99	67.72	76.65	na	36	37.5
O-99	66.90	75.69	na	36	37.5
N-99	66.82	75.43	na	36	37.5
D-99	70.53	77.18	na	36	37.5
J-00	76.24	80.04	na	37.05	37.5
F-00	75.78	81.61	na	38	37.5

Date	Prime (LME)	Alloy (AMM)	Alloy (LME)	New Scrap Clips (Chicago)	Old Scrap Sheet and Cast (Chicago)
M-00	71.55	80.47	na	38	37.5
A-00	66.12	76.66	na	37.80	35.55
M-00	66.55	74	53.68	35.09	31.09
J-00	68.34	73.26	54.06	34.32	30.32
J-00	70.94	74.46	55.56	34.30	30.30
A-00	69.31	75.25	53.46	36.00	32.00
S-00	72.65	74.94	55.05	36.00	32.00
O-00	68.07	71.81	51.93	36.00	31.09
N-00	66.87	68.82	51.24	36.00	31.00
D-00	71.04	67.86	53.06	36.00	31.00
J-01	73.31	70.54	52.30	36.00	31.00
F-01	72.80	72.93	57.15	36.11	31.00
M-01	68.47	74.73	57.11	37.00	31.00
A-01	67.92	74.51	56.34	37.00	31.00
M-01	69.82	75.09	56.02	37.00	31.00
J-01	66.52	73.98	54.22	37.00	31.00
J-01	64.26	71.34	52.95	36.10	28.29
A-01	62.48	69.79	52.95	36.00	28.00
S-01	61.00	68.85	51.43	36.00	28.00
O-01	58.19	67.77	49.78	35.78	28.00
N-01	60.23	67.11	49.40	33.40	26.40
D-01	61.00	68.05	49.40	33.00	26.00
J-02	62.10	69.71	49.20	33.00	26.00
F-02	62.13	72.2	53.23	33.00	27.47
M-02	63.75	76.44	56.68	33.95	28.95
A-02	62.16	80.79	56.56	35.00	30.00
M-02	60.95	78.79	54.78	35.00	30.00
J-02	61.43	77.52	56.12	35.00	30.00
J-02	60.71	76.02	57.74	35.00	30.00
A-02	58.60	72.94	56.80	34.64	29.64
S-02	59.04	73.32	56.11	33.00	28.00
O-02	59.46	73.09	55.73	33.00	28.00
N-02	62.26	75.2	58.82	33.00	28.00
D-02	62.39	76.99	60.63	33.00	28.00
J-03	62.53	78.37	62.89	33.10	28.10
F-03	64.52	81.4	66.21	35.00	30.00
M-03	63.03	81.24	66.10	35.00	30.00
A-03	60.43	77.58	63.87	34.00	29.00
M-03	63.45	79.21	62.88	34.00	29.00
J-03	63.97	78.25	61.49	37.19	27.90
J-03	65.16	76.52	62.35	36.68	26.68
A-03	66.07	77.84	62.84	36.00	26.00
S-03	64.22	78.79	63.10	36.00	26.00
O-03	66.89	80.72	63.59	36.00	26.00
N-03	68.43	82.74	62.90	36.00	26.00
D-03	70.55	84.62	65.36	37.33	27.33
J-04	72.88	87.47	67.30	40.00	30.00

Date	Prime (LME)	Alloy (AMM)	Alloy (LME)	New Scrap Clips (Chicago)	Old Scrap Sheet and Cast (Chicago)
F-04	76.48	92.45	70.26	41.26	31.26
M-04	75.13	95.63	70.87	46.00	36.00
A-04	78.48	95.29	72.58	46.00	36.00
M-04	73.65	87.73	68.39	42.19	33.00
J-04	76.12	82.77	69.41	38.00	29.00
J-04	77.55	84.96	70.13	38.00	29.00
A-04	76.78	86.37	69.35	38.00	29.00
S-04	78.20	86.58	69.69	38.00	29.00
O-04	82.57	87.63	72.72	38.00	29.00
N-04	82.30	89.39	74.17	38.00	29.00
D-04	83.89	90.18	75.61	38.00	29.00
J-05	83.21	90.89	74.95	38.00	29.00
F-05	85.41	91.37	76.17	38.00	29.00
M-05	89.92	93.98	78.01	40.61	31.61
A-05	85.65	94.22	75.31	42.00	33.00
M-05	79.09	89.52	70.81	40.33	31.33
J-05	78.53	86.23	70.92	37.00	28.00
J-05	80.69	86.51	71.15	37.00	28.00
A-05	84.72	88.7	73.21	37.00	28.00
S-05	83.46	90.32	72.78	37.00	28.00
O-05	87.49	91.09	73.10	37.00	28.00
N-05	93.01	93.3	76.13	37.00	28.00
D-05	101.94	96.8	85.45	49.86	39.71
J-06	107.86	104.46	92.28	52.50	42.50
F-06	111.37	113.4	105.87	52.50	42.50
M-06	110.18	114.07	106.60	53.80	43.80
A-06	118.91	120.51	110.13	57.50	47.50
M-06	129.80	131.75	119.64	57.50	47.50
J-06	112.37	117.92	106.56	63.41	55.91
J-06	113.98	112.78	103.85	53.90	46.48
A-06	111.58	114.73	99.57	52.46	45.46
S-06	112.17	112.59	98.91	51.50	44.50
O-06	120.41	113.44	100.06	53.59	46.45
N-06	122.60	117.16	101.84	59.15	51.15
D-06	127.65	117.78	103.73	59.10	51.10
J-07	127.43	117.47	100.63	60.50	52.50
F-07	128.47	118.96	98.35	60.50	52.50
M-07	125.27	121.03	99.48	62.00	54.00
A-07	127.68	124.85	100.46	63.50	55.50
M-07	126.77	121.99	99.25	63.50	55.50
J-07	121.45	116.89	97.63	61.07	53.07
J-07	123.96	115.54	97.99	60.50	52.50
A-07	114.11	116.77	96.66	60.50	52.50
S-07	108.47	119.2	97.73	59.34	51.34
O-07	110.79	121.84	99.06	58.50	50.50
N-07	113.71	123.72	102.70	58.50	50.50
D-07	108.03	123.26	103.64	58.50	50.50

Date	Prime (LME)	Alloy (AMM)	Alloy (LME)	New Scrap Clips (Chicago)	Old Scrap Sheet and Cast (Chicago)
J-08	110.93	124.67	104.84	58.50	50.50
F-08	126.05	131.25	113.70	61.50	52.00
M-08	136.32	142.52	122.69	68.50	58.50
A-08	134.23	141.68	125.08	72.09	62.09
M-08	131.67	140.02	120.87	70.12	60.12
J-08	134.17	136.14	119.21	67.50	57.50
J-08	139.31	135.53	118.34	70.00	60.00
A-08	125.39	130.24	111.46	70.60	60.60
S-08	114.57	118.91	101.90	63.69	53.69
O-08	96.23	88.64	76.37	45.76	35.76
N-08	84.03	72.07	61.79	36.94	26.94
D-08	67.60	67.10	51.09	23.21	18.45
J-09	64.10	68.68	49.39	17.50	17.50
F-09	60.34	65.01	49.52	17.50	17.50
M-09	60.59	65.26	53.35	22.27	22.27
A-09	64.45	69.21	58.97	22.50	22.50
M-09	66.25	70.68	55.60	22.50	22.50
J-09	71.38	76.05	60.57	22.50	22.50
J-09	75.66	80.93	68.38	24.55	24.55
A-09	87.71	92.33	79.20	27.50	27.50
S-09	83.19	88.64	76.24	37.50	37.50
O-09	85.21	90.43	76.64	37.50	37.50
N-09	88.42	93.86	79.40	37.50	37.50
D-09	98.89	104.62	85.60	42.24	42.24
J-10	101.39	107.10	89.15	45.39	45.39
F-10	92.94	99.13	85.86	47.50	47.50
M-10	100.05	106.60	91.44	48.80	48.80

Table D1.2 Ratios of Primary (Pp), Secondary Alloy (Pa), Old Scrap (OS), and New Scrap (NS)

Date	Pa:Pp	Pa (LME):Pp	Pos:Pp	Pns:Pp
J-85	1.31	na	0.55	0.69
F-85	1.27	na	0.56	0.70
M-85	1.22	na	0.55	0.69
A-85	1.16	na	0.53	0.67
M-85	1.13	na	0.49	0.64
J-85	1.16	na	0.50	0.67
J-85	1.13	na	0.48	0.66
A-85	1.15	na	0.50	0.68
S-85	1.22	na	0.53	0.73
O-85	1.27	na	0.54	0.74
N-85	1.30	na	0.52	0.71
D-85	1.19	na	0.48	0.66
J-86	1.17	na	0.48	0.69

Date	Pa:Pp	Pa (LME):Pp	Pos:Pp	Pns:Pp
F-86	1.26	na	0.53	0.72
M-86	1.26	na	0.56	0.74
A-86	1.27	na	0.59	0.77
M-86	1.15	na	0.48	0.70
J-86	1.03	na	0.42	0.65
J-86	1.10	na	0.45	0.72
A-86	1.14	na	0.48	0.72
S-86	1.09	na	0.49	0.69
O-86	1.14	na	0.47	0.71
N-86	1.10	na	0.47	0.70
D-86	1.06	na	0.46	0.69
J-87	1.07	na	0.48	0.70
F-87	1.02	na	0.47	0.66
M-87	1.00	na	0.48	0.67
A-87	0.99	na	0.50	0.69
M-87	1.02	na	0.52	0.72
J-87	1.04	na	0.52	0.73
J-87	0.97	na	0.48	0.68
A-87	0.94	na	0.49	0.68
S-87	0.99	na	0.51	0.70
O-87	0.93	na	0.49	0.68
N-87	1.10	na	0.55	0.74
D-87	1.01	na	0.51	0.69
J-88	0.96	na	0.50	0.67
F-88	0.92	na	0.48	0.66
M-88	0.84	na	0.46	0.64
A-88	0.90	na	0.49	0.66
M-88	0.73	na	0.39	0.56
J-88	0.63	na	0.33	0.48
J-88	0.84	na	0.45	0.65
A-88	0.81	na	0.44	0.64
S-88	0.92	na	0.50	0.73
O-88	0.91	na	0.48	0.69
N-88	0.87	na	0.45	0.67
D-88	0.88	na	0.44	0.66
J-89	0.93	na	0.49	0.69
F-89	1.03	na	0.54	0.74
M-89	1.07	na	0.57	0.77
A-89	1.01	na	0.54	0.73
M-89	0.95	na	0.50	0.69
J-89	1.08	na	0.55	0.78
J-89	1.08	na	0.54	0.74
A-89	1.00	na	0.50	0.66
S-89	1.07	na	0.56	0.73
O-89	0.99	na	0.49	0.65
N-89	1.00	na	0.48	0.67
D-89	0.99	na	0.48	0.70
J-90	1.04	na	0.50	0.73

Date	Pa:Pp	Pa (LME):Pp	Pos:Pp	Pns:Pp
F-90	1.06	na	0.46	0.66
M-90	1.04	na	0.43	0.62
A-90	1.19	na	0.47	0.71
M-90	1.17	na	0.47	0.71
J-90	1.09	na	0.46	0.69
J-90	1.07	na	0.50	0.71
A-90	0.98	na	0.46	0.64
S-90	0.85	na	0.45	0.59
O-90	0.88	na	0.48	0.63
N-90	1.00	na	0.55	0.70
D-90	1.04	na	0.56	0.70
J-91	1.05	na	0.49	0.65
F-91	1.03	na	0.47	0.63
M-91	1.02	na	0.47	0.63
A-91	1.10	na	0.48	0.65
M-91	1.11	na	0.48	0.66
J-91	1.07	na	0.39	0.58
J-91	1.09	na	0.37	0.56
A-91	1.14	na	0.38	0.58
S-91	1.18	na	0.38	0.60
O-91	1.17	na	0.36	0.59
N-91	1.17	na	0.35	0.57
D-91	1.17	na	0.36	0.58
J-92	1.09	na	0.35	0.56
F-92	1.08	na	0.40	0.59
M-92	1.16	na	0.46	0.64
A-92	1.16	na	0.45	0.63
M-92	1.17	na	0.41	0.60
J-92	1.14	na	0.41	0.60
J-92	1.10	na	0.40	0.59
A-92	1.13	na	0.41	0.59
S-92	1.12	na	0.37	0.58
O-92	1.15	na	0.37	0.60
N-92	1.15	na	0.37	0.61
D-92	1.17	na	0.36	0.58
J-93	1.24	na	0.36	0.40
F-93	1.23	na	0.36	0.40
M-93	1.27	na	0.37	0.42
A-93	1.29	na	0.39	0.44
M-93	1.24	na	0.38	0.43
J-93	1.19	na	0.37	0.42
J-93	1.18	na	0.38	0.43
A-93	1.21	na	0.39	0.45
S-93	1.24	na	0.40	0.46
O-93	1.22	na	0.41	0.47
N-93	1.26	na	0.42	0.49
D-93	1.24	na	0.40	0.46
J-94	1.22	na	0.41	0.45

Date	Pa:Pp	Pa (LME):Pp	Pos:Pp	Pns:Pp
F-94	1.24	na	0.45	0.47
M-94	1.29	na	0.48	0.51
A-94	1.31	na	0.56	0.59
M-94	1.25	na	0.54	0.57
J-94	1.24	na	0.52	0.55
J-94	1.21	na	0.53	0.56
A-94	1.24	na	0.55	0.58
S-94	1.20	na	0.58	0.60
O-94	1.16	na	0.58	0.60
N-94	1.13	na	0.59	0.61
D-94	1.16	na	0.63	0.65
J-95	1.11	na	0.60	0.64
F-95	1.16	na	0.62	0.67
M-95	1.16	na	0.60	0.65
A-95	1.12	na	0.58	0.63
M-95	1.10	na	0.56	0.61
J-95	1.07	na	0.53	0.58
J-95	1.07	na	0.52	0.58
A-95	1.14	na	0.56	0.63
S-95	1.12	na	0.56	0.64
O-95	1.09	na	0.52	0.59
N-95	1.05	na	0.50	0.57
D-95	1.07	na	0.50	0.57
J-96	1.11	na	0.52	0.59
F-96	1.07	na	0.52	0.60
M-96	1.08	na	0.51	0.59
A-96	1.11	na	0.52	0.60
M-96	1.10	na	0.52	0.55
J-96	1.14	na	0.56	0.59
J-96	1.15	na	0.57	0.60
A-96	1.14	na	0.55	0.60
S-96	1.18	na	0.55	0.62
O-96	1.19	na	0.55	0.63
N-96	1.15	na	0.51	0.58
D-96	1.16	na	0.49	0.56
J-97	1.14	na	0.50	0.57
F-97	1.18	na	0.51	0.57
M-97	1.18	na	0.49	0.55
A-97	1.22	na	0.52	0.58
M-97	1.19	na	0.49	0.56
J-97	1.22	na	0.51	0.58
J-97	1.19	na	0.51	0.54
A-97	1.13	na	0.51	0.58
S-97	1.18	na	0.53	0.65
O-97	1.19	na	0.53	0.65
N-97	1.18	na	0.53	0.65
D-97	1.19	na	0.55	0.67
J-98	1.19	na	0.56	0.66

Date	Pa:Pp	Pa (LME):Pp	Pos:Pp	Pns:Pp
F-98	1.20	na	0.56	0.67
M-98	1.20	na	0.57	0.68
A-98	1.19	na	0.56	0.67
M-98	1.21	na	0.57	0.69
J-98	1.17	na	0.53	0.66
J-98	1.09	na	0.43	0.56
A-98	1.07	na	0.43	0.55
S-98	1.06	na	0.42	0.54
O-98	1.09	na	0.43	0.56
N-98	1.10	na	0.43	0.56
D-98	1.15	na	0.45	0.58
J-99	1.19	na	0.46	0.60
F-99	1.26	na	0.47	0.61
M-99	1.27	na	0.48	0.61
A-99	1.21	na	0.44	0.55
M-99	1.24	na	0.44	0.56
J-99	1.28	na	0.49	0.57
J-99	1.20	na	0.58	0.54
A-99	1.19	na	0.58	0.55
S-99	1.13	na	0.55	0.53
O-99	1.13	na	0.56	0.54
N-99	1.13	na	0.56	0.54
D-99	1.09	na	0.53	0.51
J-00	1.05	na	0.49	0.49
F-00	1.08	na	0.49	0.50
M-00	1.12	na	0.52	0.53
A-00	1.16	na	0.54	0.57
M-00	1.11	0.81	0.47	0.53
J-00	1.07	0.79	0.44	0.50
J-00	1.05	0.78	0.43	0.48
A-00	1.09	0.77	0.46	0.52
S-00	1.03	0.76	0.44	0.50
O-00	1.05	0.76	0.46	0.53
N-00	1.03	0.77	0.46	0.54
D-00	0.96	0.75	0.44	0.51
J-01	0.96	0.71	0.42	0.49
F-01	1.00	0.79	0.43	0.50
M-01	1.09	0.83	0.45	0.54
A-01	1.10	0.83	0.46	0.54
M-01	1.08	0.80	0.44	0.53
J-01	1.11	0.82	0.47	0.56
J-01	1.11	0.82	0.44	0.56
A-01	1.12	0.85	0.45	0.58
S-01	1.13	0.84	0.46	0.59
O-01	1.16	0.86	0.48	0.61
N-01	1.11	0.82	0.44	0.55
D-01	1.12	0.81	0.43	0.54
J-02	1.12	0.79	0.42	0.53

Date	Pa:Pp	Pa (LME):Pp	Pos:Pp	Pns:Pp
F-02	1.16	0.86	0.44	0.53
M-02	1.20	0.89	0.45	0.53
A-02	1.30	0.91	0.48	0.56
M-02	1.29	0.90	0.49	0.57
J-02	1.26	0.91	0.49	0.57
J-02	1.25	0.95	0.49	0.58
A-02	1.24	0.97	0.51	0.59
S-02	1.24	0.95	0.47	0.56
O-02	1.23	0.94	0.47	0.55
N-02	1.21	0.94	0.45	0.53
D-02	1.23	0.97	0.45	0.53
J-03	1.25	1.01	0.45	0.53
F-03	1.26	1.03	0.46	0.54
M-03	1.29	1.05	0.48	0.56
A-03	1.28	1.06	0.48	0.56
M-03	1.25	0.99	0.46	0.54
J-03	1.22	0.96	0.44	0.58
J-03	1.17	0.96	0.41	0.56
A-03	1.18	0.95	0.39	0.54
S-03	1.23	0.98	0.40	0.56
O-03	1.21	0.95	0.39	0.54
N-03	1.21	0.92	0.38	0.53
D-03	1.20	0.93	0.39	0.53
J-04	1.20	0.92	0.41	0.55
F-04	1.21	0.92	0.41	0.54
M-04	1.27	0.94	0.48	0.61
A-04	1.21	0.92	0.46	0.59
M-04	1.19	0.93	0.45	0.57
J-04	1.09	0.91	0.38	0.50
J-04	1.10	0.90	0.37	0.49
A-04	1.12	0.90	0.38	0.49
S-04	1.11	0.89	0.37	0.49
O-04	1.06	0.88	0.35	0.46
N-04	1.09	0.90	0.35	0.46
D-04	1.07	0.90	0.35	0.45
J-05	1.09	0.90	0.35	0.46
F-05	1.07	0.89	0.34	0.44
M-05	1.05	0.87	0.35	0.45
A-05	1.10	0.88	0.39	0.49
M-05	1.13	0.90	0.40	0.51
J-05	1.10	0.90	0.36	0.47
J-05	1.07	0.88	0.35	0.46
A-05	1.05	0.86	0.33	0.44
S-05	1.08	0.87	0.34	0.44
O-05	1.04	0.84	0.32	0.42
N-05	1.00	0.82	0.30	0.40
D-05	0.95	0.84	0.39	0.49
J-06	0.97	0.86	0.39	0.49

Date	Pa:Pp	Pa (LME):Pp	Pos:Pp	Pns:Pp
F-06	1.02	0.95	0.38	0.47
M-06	1.04	0.97	0.40	0.49
A-06	1.01	0.93	0.40	0.48
M-06	1.02	0.92	0.37	0.44
J-06	1.05	0.95	0.50	0.56
J-06	0.99	0.91	0.41	0.47
A-06	1.03	0.89	0.41	0.47
S-06	1.00	0.88	0.40	0.46
O-06	0.94	0.83	0.39	0.45
N-06	0.96	0.83	0.42	0.48
D-06	0.92	0.81	0.40	0.46
J-07	0.92	0.79	0.41	0.47
F-07	0.93	0.77	0.41	0.47
M-07	0.97	0.79	0.43	0.49
A-07	0.98	0.79	0.43	0.50
M-07	0.96	0.78	0.44	0.50
J-07	0.96	0.80	0.44	0.50
J-07	0.93	0.79	0.42	0.49
A-07	1.02	0.85	0.46	0.53
S-07	1.10	0.90	0.47	0.55
O-07	1.10	0.89	0.46	0.53
N-07	1.09	0.90	0.44	0.51
D-07	1.14	0.96	0.47	0.54
J-08	1.12	0.95	0.46	0.53
F-08	1.04	0.90	0.41	0.49
M-08	1.05	0.90	0.43	0.50
A-08	1.06	0.93	0.46	0.54
M-08	1.06	0.92	0.46	0.53
J-08	1.01	0.89	0.43	0.50
J-08	0.97	0.85	0.43	0.50
A-08	1.04	0.89	0.48	0.56
S-08	1.04	0.89	0.47	0.56
O-08	0.92	0.79	0.37	0.48
N-08	0.86	0.74	0.32	0.44
D-08	0.99	0.76	0.27	0.34
J-09	1.07	0.77	0.27	0.27
F-09	1.08	0.82	0.29	0.29
M-09	1.08	0.88	0.37	0.37
A-09	1.07	0.91	0.35	0.35
M-09	1.07	0.84	0.34	0.34
J-09	1.07	0.85	0.32	0.32
J-09	1.07	0.90	0.32	0.32
A-09	1.05	0.90	0.31	0.31
S-09	1.07	0.92	0.45	0.45
O-09	1.06	0.90	0.44	0.44
N-09	1.06	0.90	0.42	0.42
D-09	1.06	0.87	0.43	0.43
J-10	1.06	0.88	0.45	0.45

Date	Pa:Pp	Pa (LME):Pp	Pos:Pp	Pns:Pp
F-10	1.07	0.92	0.51	0.51
M-10	1.07	0.91	0.49	0.49

D2 Aluminum Beverage Can Case Study Methodology

The net trade of can stock is not included in the model due to a lack of sufficiently detailed export data. In 1990, 68,000 metric tons of can stock was imported (U.S. ITC 2009). By 1993 imports had decreased to roughly 13,300 tons and by 1996 less than 1,000 tons of can stock was imported (U.S. ITC 2009).

D2.1 Model Equations

The total mass of aluminum required annually in year t , T_t, is calculated as

$$T_t = \frac{S_t}{1-\delta_m}\left[\delta_l\left(\frac{1}{1-\delta_r}-1\right)+1\right], \qquad \text{(D2.1)}$$

where S_t is the mass of aluminum beverage can shipments in year t and δ_m, δ_r are the beverage can manufacturing and can stock rolling losses, respectively. The fraction of metal lost remelting the rolling scrap that is returned directly to the process is represented by the general melt loss parameter, δ_l.

The mass of secondary metal recovered is calculated by multiplying the sum of collected UBCs and estimated can manufacturing scrap by the UBC remelting loss. The amount of primary aluminum required as makeup is the difference between T_t and the mass of secondary metal.

The annual amount of UBCs consumed in year t, C_t, is estimated as

$$C_t = R_t + I_t + V_t, \qquad \text{(D2.2)}$$

Where R_t , I_t, and V_t are the annual amount of UBCs collected, net UBC imports, and inventory adjustment respectively.

The amount of secondary material recovered is calculated as

$$\left(C_t + \frac{S_t\delta_m}{1-\delta_m}\right)(1-\delta_s), \tag{D2.3}$$

where δ_s is the UBC melt loss. The difference between (D2.1) and (D2.2) represents the annual required mass of primary aluminum.,$P_{t,}$ or equivalently

$$P_t = \frac{S_t}{1-\delta_m}\left[\delta_l\left(\frac{1}{1-\delta_r}-1\right)+\delta_m(\delta_s-1)+1\right]+C_t\left(\delta_s-1\right) \tag{D2.4}$$

The EOLR approach assumes that the mass of secondary material is equal to the mass of material is not lost to the system. For aluminum beverage cans the mass of secondary material in the EOLR case is the same as in the recycled content (RC) case, which is represented by (2). The amount of primary aluminum required is equal to the mass of material lost with the EOLR approach in year t, L_t, which is calculated as

$$L_t = S_t - R_t + \left(C_t + \frac{S_t\delta_m}{1-\delta_m}\right)\delta_s + \frac{S_t\delta_l}{1-\delta_m}\left(\frac{1}{1-\delta_r}-1\right). \tag{D2.5}$$

The third group of terms represents the mass of aluminum lost to remelting rolling scrap. The mass of material lost is equivalent to the mass of primary aluminum that must be consumed as makeup to the system.

Total GHG emissions in year t using the RC approach are calculated as

$$P_t\mathrm{GHG}_{p,t} + \left(C_t + \frac{S_t\delta_m}{1-\delta_m}\right)(1-\delta_s)\mathrm{GHG}_s \tag{D2.6}$$

where $\mathrm{GHG}_{p,t}$ is the GHG intensity of primary ingot in year t and GHG_s is the GHG intensity of secondary ingot production.

Total GHG emissions in year *t* using the EOLR approach are calculated as

$$L_t \mathrm{GHG}_{p,t} + \left(C_t + \frac{S_t \delta_m}{1-\delta_m}\right)(1-\delta_s)\mathrm{GHG}_s \quad \text{(D2.7)}$$

The difference between the emissions calculated by the RC approach and the EOL recycling approach in year *t* is expressed by the equation

$$\left(C_t - R\right)_t GHG_p \quad \text{(D2.8)}$$

D2.2 Implications of Secondary Alloy Emission Factor on Model Results

Table D2.1 Model Results Using IAI (2000) Recycled Ingot Emission Factor, MMTCO2-eq

	RC			EOLR		
	GHG	GHG High	GHG Low	GHG	GHG High	GHG Low
1990	9.54	10.66	8.47	9.88	11.04	8.76
1991	9.33	10.07	8.58	9.86	10.64	9.07
1992	8.25	8.90	7.59	8.66	9.34	7.97
1993	8.63	9.35	7.91	9.12	9.88	8.36
1994	7.81	8.48	7.15	8.88	9.65	8.12
1995	8.25	8.90	7.60	9.60	10.36	8.83
1996	8.05	8.78	7.32	8.88	9.70	8.07
1997	7.24	7.96	6.51	8.43	9.28	7.58
1998	7.59	8.31	6.88	8.76	9.60	7.93
1999	7.00	7.83	6.16	8.03	9.00	7.06
2000	7.24	8.00	6.48	8.54	9.45	7.63

Table D2.2 Model Results Using EAA (2008) Recycled Ingot Emission Factor, MMTCO2-eq

	RC			EOLR		
	GHG	GHG High	GHG Low	GHG	GHG High	GHG Low
1990	9.54	10.66	8.47	9.88	11.04	8.76
1991	9.33	10.07	8.58	9.86	10.64	9.07
1992	8.25	8.90	7.59	8.66	9.34	7.97
1993	8.63	9.35	7.91	9.12	9.88	8.36
1994	7.81	8.48	7.15	8.88	9.65	8.12
1995	8.25	8.90	7.60	9.60	10.36	8.83
1996	8.05	8.78	7.32	8.88	9.70	8.07
1997	7.24	7.96	6.51	8.43	9.28	7.58
1998	7.59	8.31	6.88	8.76	9.60	7.93
1999	7.00	7.83	6.16	8.03	9.00	7.06

	RC			EOLR		
	GHG	GHG High	GHG Low	GHG	GHG High	GHG Low
2000	7.24	8.00	6.48	8.54	9.45	7.63

D3 Aluminum Engine Block Case Study

D3.1 Model Equations

The total mass of aluminum required to produce ingot *I* in kg for a fleet of 100,000 is

$$T = \frac{100{,}000 \times m}{\delta_c\left(1 - \delta_l - \theta_c\delta_l\right)} \tag{D3.1}$$

where m is the mass of aluminum in each engine block in kg, δ_c is the sand casting scrap rate, δ_l is the metal loss during scrap remelting, and

The RC approach requires estimates of the amounts of primary and secondary aluminum consumed to produce the mass of finished castings. The total mass of required secondary aluminum S in kg is

$$S = T\chi \tag{D3.2}$$

where T is as described in (D2.1), and χ is the fraction of secondary aluminum alloy.

The total required mass of primary aluminum P in kg is simply the difference between the total required aluminum, T, and the mass of secondary aluminum, S.

The EOL approach calculates the mass of primary aluminum required by the product system as the sum of all metal losses. For clarity and transparency each incident of metal loss has been described individually. The metal in kg lost during scrap recycling for secondary ingot production, L_S, is

$$L_S = \frac{S\delta_l}{1 - \delta_l}\ . \tag{D3.3}$$

The mass of aluminum lost during the remelt of casting scrap that is directly returned to the casting process, L_C, is

$$L_C = T\left(1-\theta_c\right)\delta_l \tag{D3.4}$$

In addition to the metal lost during secondary metal and engine casting production, losses occur as vehicles are retired and then shredded. The metal in kg lost during ELV collection in year i, $L_{ELV,i}$, is

$$L_{\mathrm{ELV},i} = D_i\left(1-\theta_{\mathrm{ELV}}\right) \tag{D3.4}$$

where D_i is the mass in kg of scrapped engine blocks in year i and θ_{ELV} is the ELV recycling rate.

The metal in kg lost during shredding in year i, $L_{r,i}$, is

$$L_{r,i} = \left(D_i - L_{ELV,i}\right)\left(1-\theta_r\right) \tag{D3.5}$$

where θ_r is the shredding and separation recovery rate.

The metal in kg lost during nonferrous separation in year i, $L_{\mathrm{n},i}$, is

$$L_{\mathrm{n},i} = \left(D_i - L_{\mathrm{ELV},i} - L_{r,i}\right)\left(1-\theta_{\mathrm{n}}\right) \tag{D3.6}$$

The total mass of aluminum losses during end-of-life operations, L_{EOL}, is equal to the sum of the losses in the collection, shredding, and nonferrous separation activities over the 30-year period from 1990 to 2020. This is represented in equation form as

$$L_{EOL} = \sum_{i}^{30}\left(L_{ELV,i} + L_{r,i} + L_{n,i}\right). \tag{D3.7}$$

Total GHG emissions are calculated using the RC approach by the equation

$$P \times \mathrm{GHG}_{p,1990} + S \times \mathrm{GHG}_s + (100{,}000m) \times \mathrm{GHG}_c + \sum_{i=0}^{30} \left(\left(D_i - L_{\mathrm{ELV},i}\right)\mathrm{GHG}_d + \left(D_i - L_{\mathrm{ELV},i} - L_{r,i}\right)\mathrm{GHG}_r + \left(D_i - L_{\mathrm{ELV},i} - L_{r,i} - L_{n,i}\right)\mathrm{GHG}_n \right) \tag{D3.8}$$

where $\mathrm{GHG}_{p,1990}$ is the GHG emission intensity of primary production in 1990, GHG_c is the emission intensity of shape casting, GHG_s is the emission intensity of secondary ingot production, and GHG emissions intensities of primary and secondary aluminum, shape casting, ELV dismantling, shredding, and separation are $\mathrm{GHG}_{p,1990}$, GHG_s, GHG_c, GHG_d, GHG_r, and GHG_n respectively. The GHG emission factor for secondary aluminum ingot production is assumed to be 0.506 kg CO_2-e/kg, (EAA 2008). The dismantling GHG emission factor of 0.000827 kg CO_2-e/kg, the shredding emission factor of 0.028 kg CO_2-e/kg, and the nonferrous separation emission factor of 0.023 kg CO_2-e/kg were developed based on energy data provided by Staudinger and Keoleian (2001) and electricity emissions factors found in the U.S. LCI Database (NREL 2009).

The EOLR approach calculates total GHG emissions as

$$\left(L_s + L_c\right)\mathrm{GHG}_{p,1990} + 100{,}000m \times \mathrm{GHG}_c + \sum_{i=0}^{30} \begin{pmatrix} \left(L_{\mathrm{ELV},i} + L_{r,i} + L_{n,i}\right)\mathrm{GHG}_{p,i} + \left(D_i - L_{\mathrm{ELV},i}\right)\mathrm{GHG}_d + \\ \left(D_i - L_{\mathrm{ELV},i} - L_{r,i}\right)\mathrm{GHG}_r + \left(D_i - L_{\mathrm{ELV},i} - L_{r,i} - L_{n,i}\right)\mathrm{GHG}_n \end{pmatrix} \tag{D3.9}$$

The difference between GHG emissions in kg CO_2e calculated by the RC and EOLR approaches in 1990 is summarized as

$$(P - \mathrm{E}_m) \times \mathrm{GHG}_{p,1990} + S \times \mathrm{GHG}_s - \sum_{i=1990}^{2020} \left((\mathrm{E}_{\mathrm{ELV},i} + \mathrm{E}_{r,i}) \times \mathrm{GHG}_{p,i} \right) \tag{D3.10}$$

The difference between GHG emissions in kg CO_2e calculated by the RC and EOLR approaches in year i is summarized as

$$- \sum_{i=1990}^{2020} ((E_{ELV,i} + E_{r,i}) \times GHG_{p,i}) \quad (D3.11)$$

D3.2 Implications of Secondary Alloy Emission Factor on Model Results

Table D3.2Model Results Using EAA (2008) Secondary Alloy Emission Factor

Allocation Approach	Retirement Distribution	TCF?	Total (metric tons CO2-eq)	Fraction from Initial Production	Fraction from End-of-Life Management
RC	Schmoyer (2001)	No	5,780	99.1%	0.9%
		Yes	10,270	99.5%	0.5%
	Lu (2006)	No	5,780	99.1%	0.9%
		Yes	10,280	99.5%	0.5%
EOL	Schmoyer (2001)	No	9,630	35.8%	64.2%
		Yes	12,340	50.1%	49.9%
	Lu (2006)	No	10,260	33.6%	66.4%
		Yes	12,960	47.5%	52.5%

Table D3.3 Model Results Using IAI (2000) Secondary Alloy Emission Factor

Allocation Approach	Retirement Distribution	TCF?	Total (metric tons CO2-eq)	Fraction from Initial Production	Fraction from End-of-Life Management
RC	Schmoyer (2001)	No	5,940	99.2%	0.8%
		Yes	10,570	99.5%	0.5%
	Lu (2006)	No	5,940	99.1%	0.9%
		Yes	10,570	99.5%	0.5%
EOL	Schmoyer (2001)	No	9,790	36.9%	63.1%
		Yes	12,630	49.0%	51.0%
	Lu (2006)	No	10,420	34.7%	65.3%
		Yes	13,260	48.6%	51.4%

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