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Resource Use and Water Implications of Material Consumption in Consumer Electronics

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Resource Use and Water Implications of Material Consumption in Consumer Electronics

by

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ABSTRACT

Rapid technological innovation has introduced a broad spectrum of materials in the consumer electronics sector. Consumption of these materials increases the demand for water and potentially discharge contaminants into the water resources across their life cycle, exacerbating freshwater scarcity and pollution. These water impacts have not yet been fully studied, as most literature on consumer electronics focuses on supply chain energy, carbon footprint, or end of life management. Evaluating water impacts requires data on material content, life cycle water consumption and emissions at spatial level, and availability of impact assessment models that connects life cycle data to water impacts. Data on these aspects are available at varied degrees for different materials used in the electronics.

This research created data on materials used in consumer electronics and studied implications on water resources for two major material categories - metals and plastics. Bill of materials (BOMs) data were created for 95 unique consumer electronic products that contain information on mass of major materials and components. Then, life cycle water impacts associated with extraction and production of metals found in consumer electronics are evaluated to identify material hotspots for future improvement. Water impacts were analyzed for individual metals and then for the representative metal profile of case study products (smartphones and laptop computers). Finally, profile of polymers and additives in the e-waste is created to understand linkage to water impacts as well to evaluate implications to establishing e-plastics circular systems.

Results indicate that, on the individual material level, precious metals have the highest water impacts in their supply chain. Water scarcity impact is mainly because of water consumed directly for mining operations and indirectly for energy production, and water degradation attributed to metal emissions during mine tailings management. The geographical region where metal production happens is also a contributing factor to water impacts, as water stress varies spatially. Therefore, sourcing metals from regions with lower water stress is an opportunity to reduce supply chain water impacts. At product level, precious metals have the highest contribution per smartphone, whereas aluminum has the highest contribution per laptop. Product design changes, such as use of recycled metal or using a low impact metal are observed to reduce water impacts. Further, e-waste shows a diverse mix of polymers and additives, including flame

retardants, pigments, and heavy metals that can potentially pollute water resources if released. As a result, transition to circular systems is important to keep the plastics from entering the environment. To enable this transition, multistakeholder engagement in the electronics sector is required to make an informed decisions in product design, policy planning and material recovery infrastructure.

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Chapter 1 Introduction

1. Material use in consumer electronics and related interactions with water resources

Rapid technological innovation has introduced a broad spectrum of materials to consumer electronics, including base metals such as steel and aluminum, precious metals such as gold and platinum, critical metals like indium and rare earth elements (REEs) (Cucchiella et al. 2015; Işıldar et al. 2018), hazardous metals such as lead or mercury (Chen et al. 2011; Kiddee, Naidu, and Wong 2013), and hard to recycle materials like polymers containing halogenated flame retardants (Friege 2012; Christian et al. 2014). These materials play an essential role in providing the form, finish, or functionality that consumers demand (Cucchiella et al. 2015; Ryen et al. 2014; Tansel 2017). For example, lighter and thinner touch-enabled flat panel technology used in phones, tablets, and TVs relies on scarce metals like indium (Boundy, Boyton, and Taylor 2017). Lithium-ion batteries are widely used to power consumer electronics because of their high energy density, which is provided by materials such as lithium and cobalt (Zubi et al. 2018). Due to rapid innovation cycles, declining product lifespans (Bakker et al., 2014), and introduction of new technologies, the demand for a diverse array of materials is expected to continually increase and evolve (Althaf, Babbitt, and Chen 2021).

While materials have transformed the consumer electronics industry, increasing demand has also led to new sustainability challenges. Consumer electronics literature has addressed supply chain security (Gaustad et al. 2018), physical resource availability (Olivetti et al. 2017), supply chain energy usage (Deng, Babbitt, and Williams 2011; Ryen, Babbitt, and Williams 2015; Socolof et al. 2001; Williams 2004; Yu, Williams, and Ju 2010), carbon footprint (Hischier and Baudin 2010; Huang, Weber, and Matthews 2009; Moberg et al. 2014; Ryen, Babbitt, and Williams 2015; Teehan and Kandlikar 2013), and improper end-of-life management impacts (Chen et al. 2011; Kiddee, Naidu, and Wong 2013). This body of research has indirectly addressed water impacts, but not fully evaluated the connection between related sustainability issues and water consumption or pollution. For example, the highest contributing factor towards carbon emissions in the electronics supply chain is electricity usage, which also consumes and degrades water resources during fossil fuel combustion (Mekonnen, Gerbens-Leenes, and Hoekstra 2015). Further, mining residues, or tailings, can release harmful compounds to water that exacerbate health impacts and contribute to social and political disruptions (Adonteng-Kissi and Adonteng-

Kissi 2017). Additionally, when e-plastics ends up in either uncontrolled landfills or elsewhere in the environment, microplastics (Chai et al. 2020; Labunska et al. 2013; Zhang et al. 2021), flame retardants (FRs) and other contaminants can potentially leach into water resources, degrading the water quality (Chen et al. 2012; Kiddee, Naidu, and Wong 2013). Considering these interactions between water and other sustainability issues, it is essential to analyze how the materials used in electronics contribute towards water impacts and to identify opportunities to reduce these impacts.

2. Methodologies available to evaluate water impacts

Freshwater scarcity and pollution are a rising global concern. Freshwater represents only 3% of total water on the earth's surface, of which only 1% is readily available for human use, and rest is locked in glaciers and ground water (Berger, Pfister, and Motoshita 2016). This uneven distribution of water has led to spatially and temporally variation in stress on water resources (see Figure 1.1). In addition to lack of water availability, pollution also contributes to stress on water resources. All human activities, including agricultural, industrial, and domestic result in release of pollutants degrading water quality. For instance, it is estimated that more than 80% of global wastewater is discharged back to water bodies without any treatment degrading water quality and causing damage to ecosystems (UNWWDR 2017). Furthermore, water stress is aggravated by population changes (Liyanage and Yamada 2017), climate change impacts (Gosling and Arnell 2016; Haddeland et al. 2014), and expansion of agricultural activities to accommodate growing population (Parris 2011). As a result, there is a mounting pressure on the businesses to implement more sustainable and innovative practices related to water use and pollution.

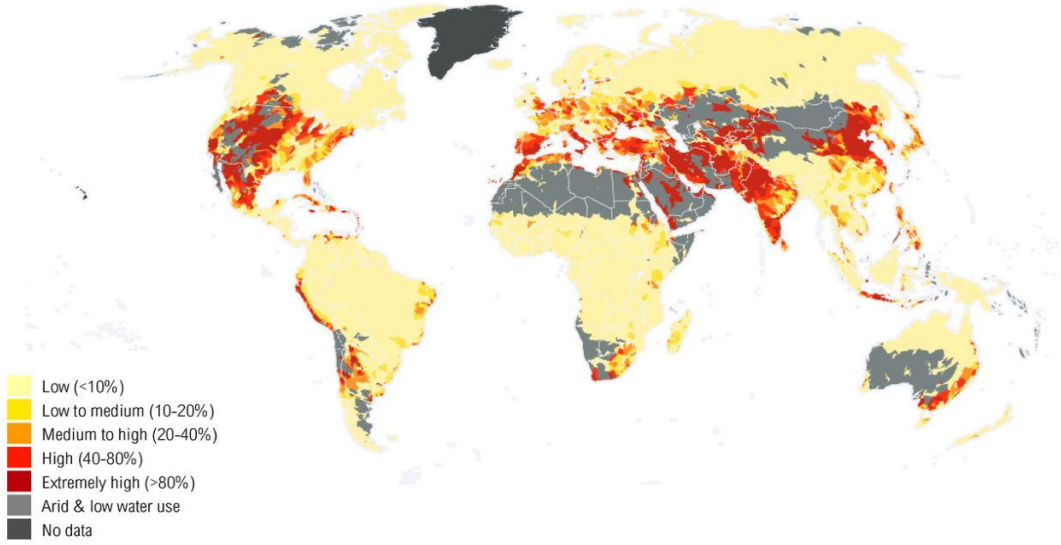

Figure 1.1 Water stress by country by World Resources Institute

Note: Source of the map is Gassert et al (2015). Water stress shown here is estimated using Aqueduct water tool by World Resources Institute. Water stress is represented in terms of total annual water withdrawals expressed as a percent of the total annual available flow.

The concept of water footprint was developed to address freshwater scarcity and pollution impacts by understanding the life cycle water use and emissions of any product or processes or businesses and its implications to the environment. Life cycle thinking provides a holistic view as it includes all the environmental interactions that happen across the product life cycle, therefore avoiding the shift in environmental burden. Further, life cycle analysis also provides information on hotspots, that is helpful to overall sustainability of products through changes in process, material, and design choices (European Commission 2010). Currently, two methodologies are available to evaluate water impacts from the life cycle perspective: 1) Water footprint network (WFN) and 2) ISO 14046 LCA water footprint.

The WFN methodology is based on the concept of "virtual water," which is defined as the water used in the production of products or services (Allan 1997). The WFN adapted and expanded the scope of virtual water definition to include global water use flows and redefined the water footprint as "the total volume of freshwater consumed over the entire supply chain of a product or service" (Hoekstra et al. 2012; Hoekstra and Chapagain 2006). The framework of WFN methodology includes four phases: goal and scope, water footprint accounting, sustainability assessment, and response formulation (see Figure 1.2). WFN mainly relies on multidimensional indicators that account for geographical and temporal specifications along with volume of water

consumed. These indicators include blue water, which accounts for ground and surface water; green water, which accounts for rainwater consumed; and gray water, which accounts for freshwater needed to dilute the pollution caused by the processes (Hoekstra et al. 2012).

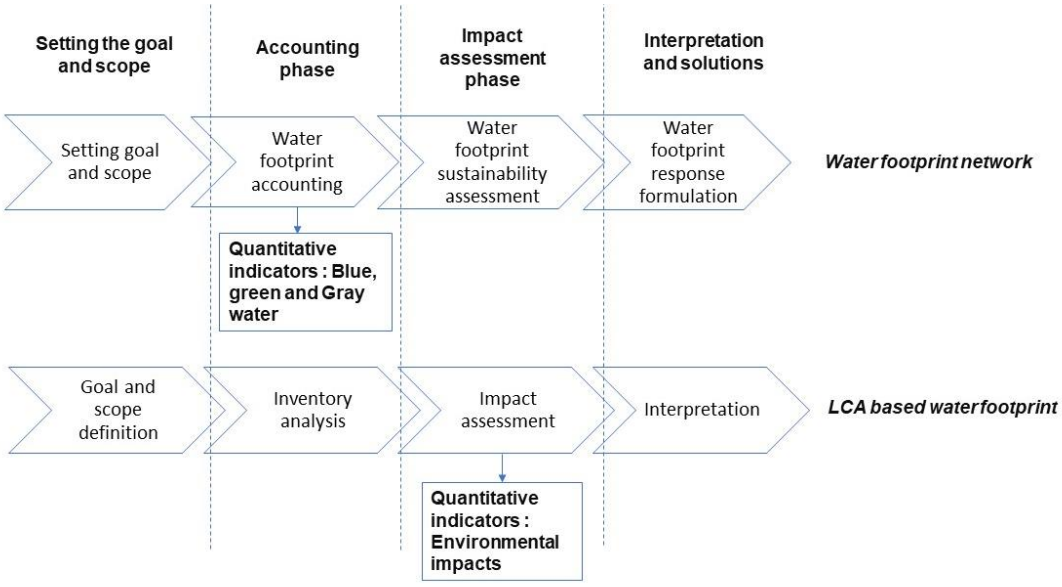

Figure 1.2 Framework of water footprint network and lifecycle assessment methodology
Note: figure has been adapted from Boulay, Hoekstra, and Vionnet (2013)

Increasing concerns on water use has led to the development of a framework by the life cycle assessment community to quantify cumulative resource inputs, emission releases, and resulting environmental impacts across a full material supply chain and/or product life cycle. These impacts can include the consumption of freshwater resources and the degradation of water quality due to pollutant releases. Measuring consumptive and degradative water impacts has been enabled by advancements in life cycle water impact assessment methods (e.g. (Boulay et al. 2018a; 2011; Pfister, Koehler, and Hellweg 2009; Ridoutt and Pfister 2010)). The ISO 14046 (ISO 2020) standard was developed to provide a framework (See Figure 1.2) similar to LCA principles, for analyzing water footprint encompassing of consumption, pollutant release, and attendant impacts (Kounina et al. 2013). In this context, water footprint is defined as a metric that aims to address the potential environmental damage and the deprivation caused by water usage.

While differences exist in the model development, application, and communication, both the methodologies ultimately drive solutions towards water conservation. For instance, WFN

methodology has been applied to primarily estimate volumetric water consumption of products such as food (Ercin, Aldaya, and Hoekstra 2012; Gerbens-Leenes, Mekonnen, and Hoekstra 2013; Page, Ridoutt, and Bellotti 2012; Ridoutt et al. 2010; Ruini et al. 2013), textile products (Chapagain et al. 2006; Chico, Aldaya, and Garrido 2013) or paper products (Schyns, Booij, and Hoekstra 2017; van Oel and Hoekstra 2012) for identifying activities that contribute the greatest towards water use. On the other hand, the ISO water footprint methodology has been applied to a wide range of products and sectors, including food (Hess et al. 2016; Manzardo et al. 2016; Silalertruksa et al. 2017), automobiles (Berger et al. 2012), metal production (Buxmann, Koehler, and Thylmann 2016), plant products (Musikavong and Gheewala 2016), and dairy products (Ridoutt and Hodges 2017) for evaluating sustainability solutions to reduce water use related impacts. These examples highlight the utility of both methodologies in supporting decision making to conserve water resources and the choice of the methodology is dependent on the goal of the research. As the main aim of this dissertation is to evaluate the water impacts, the study opted for life cycle-based water footprint assessment framework.

The ability to conduct water footprint assessment and create sustainable solutions for materials used in consumer electronics depends on the data availability on the following three aspects: 1) Material profile, which provides information on type and quantity of materials and where these materials are being used in the product, 2) Information on life cycle freshwater withdrawals, and emissions to water that impact water quantity and quality on a spatial scale at individual processes level (i.e., life cycle inventory (LCI) data), 3) Impact assessment models that link LCI data on water consumption and degradation to a cumulative water footprint indicator. In this dissertation, consumer electronics materials analyzed are metals and plastics, and information on the above three aspects are available at varied levels for these materials (see Table 1.1). For metals, data on material content is limited to specific case study products (Huisman et al. 2008; Hikwama 2005; Oguchi et al. 2011; Teehan and Kandlikar 2013). However, life cycle databases (e.g., ecoinvent, GaBi, and U.S. LCI databases) are available for metals that capture life cycle water consumption and emissions on a spatial and process scale (Classen et al. 2009). Further, established impact assessment models are available that explain the linkage between life cycle inventory data of metal supply chain and water impacts (Frischknecht et al. 2007). Therefore, the next step for metals is to create material content data for products and connect to the existing life cycle inventory data to conduct a complete water footprint assessment.

On the other hand, for plastics, minimal data exists on material content in products, which is outdated (Fisher et al. 2000) and is limited to studies specific to Europe (Beccagutti et al. 2016; Chen et al. 2012; Florea et al. 2011; Fontana et al. 2019; Maris et al. 2015; Martinho et al. 2012; Peeters et al. 2015; 2012; Schlummer et al. 2007; Strobl et al. 2021; Wagner et al. 2020). Also, existing LCI databases for plastics are based on generic averages and do not capture data at spatial and processes level (Hischier 2007). Furthermore, current impact assessment models do not completely establish the linkage between plastics use in electronics and water impacts (Frischknecht et al. 2007). Therefore, for plastics, the next step is to establish what these materials are and how they are used in electronics to understand the potential linkages to water resources.

Table 1.1 Research gap for consumer electronic materials

Information needed for water footprint	**Example units**	**Metals**	**Plastics**
Material Intensity per product	kg/product	Limited to specific case study products	Limited to European studies, Outdated U.S. studies
Life cycle inventory of water consumption and emissions to water per material	m^3 of water /kg of material, kg element /kg of material	Available with spatial resolution at process level	Minimal, limited to generic averages
Impact models that connect inventories to water impacts	m^3 eq./m^3, kgP eq. /kg P, $kgSO_2eq/kgSO_2$	Established linkage between metals supply chain and water impacts	Linkage between plastic and water impact is incomplete

3. Research objective and questions

This dissertation aims to answer the overarching question: What materials are used in consumer electronics and how do these materials link to impact on water resources? Chapter 2 addresses part of this research question by creating material and component profile data of consumer electronic products, which is essential to identify key materials and their quantity to conduct water footprint assessment. Chapter 3 then uses the data created on metals in Chapter 2, builds life cycle inventory data on water use and emissions, and utilize impact assessment models to assess water impacts associated with extraction and refining processes of metals. Further, Chapter 3 also identifies various opportunities to reduce the water impacts associated with metals

through product design and material sourcing strategies. Chapter 4 establishes the profile of polymers and additives used in consumer electronics to understand their potential linkage to impact on water resources. Chapter 4 also integrates the polymer profile data with published material flow analysis model to quantify national level e-waste flows and evaluate implications to establishing e-plastics circular system to tackle plastic pollution. Chapter 5 provides conclusions on this research along with limitations, and future research opportunities. The novelty of this research is that it presents for the first time, a comprehensive and transparent database on product material and component profile, freshwater scarcity and quality impacts associated with metals supply chain, profile of polymers and additives in the e-waste exploring the implications to water resources as well as establishing e-plastics circular systems in the U.S. to tackle plastic pollution. This information can guide various stakeholders in the industry in improving the overall sustainability of consumer electronics through product design, policy planning and material recovery.

Three research questions are posed to address the overarching objective, which are detailed below

Research Question 1: What are the materials and components in consumer electronics that may contribute to water impacts?

Approach: Create a comprehensive and transparent database on commonly used consumer electronic products by disassembly, physical characterization methods and published literature resources. Identify the key materials and their quantities that are needed to conduct water footprint assessment.

Research Question 2: What are the freshwater scarcity and pollution impacts associated with metals used in consumer electronics and opportunities to reduce these impacts?

Approach: Use life cycle water footprint assessment method to evaluate supply chain water impacts of metals to identify “hotspots” for future improvement and explore the opportunities to reduce water these impacts through supply chain diversification and product design changes.

Research Question 3: What is the current profile of polymers and additives used in consumer electronics? What is the potential link between these materials and impacts on water resources?

Approach: Create a profile on polymer types and additives used in consumer electronics by physical identification, and spectroscopic techniques (FTIR and XRF) to establish linkages to water impacts. Evaluate implications to implementing circular systems as a solution to address plastic pollution.

Chapter 2 Disassembly-based bill of materials data for consumer electronic products

(Note: This chapter is an adaption from a previously published journal article "Babbitt, C.W., Madaka, H., Althaf, S. et al. Disassembly-based bill of materials data for consumer electronic products. Sci Data 7, 251 (2020). https://doi.org /10.1038/s41597-020-0573-9)

1. Introduction

To model life cycle inventory data and carry out water footprint assessment, the first step is to establish the bill of materials. These BOMs includes data on type and quantity of materials, and components in which these materials are being used. This information will help us to identify the key materials to prioritize impact assessment as well as respective quantities in a product.

BOM data for electronics products are rarely disclosed by manufacturers. While studies do exist in the literature, sources available vary widely in transparency and reproducibility, as data are presented in various formats and levels of detail. For instance, high level data are available on specific to case study products, such as CRT TVs (Huisman et al. 2008) and monitors (Hikwama 2005), printers (Oguchi et al. 2011), and desktops (Oguchi et al. 2011). Only one study was found that have detailed BOM data for a wide range of products of different model years, product designs, and functional attributes (Teehan and Kandlikar 2013).

Therefore, this research was carried out to collect, verify, and disseminate BOM data that describe the major materials and components contained in common consumer electronic products. The primary goal was creating a transparent database for a wide cross-section of technologies and time periods that could be used by other researchers studying sustainable solutions for consumer electronics. Thus, the study focused on empirical data, obtained by extensive product disassembly and physical material characterization, and organized into a reproducible framework. Recognizing that consumer electronics will continue to evolve in the future, this data set can be updated following this framework as new products enter the market and as other researchers publish studies containing BOM data. To this end, the study also evaluated existing examples of BOM data available in the open literature, which were found to vary widely in quality and reproducibility. Select literature values were also included to supplement the empirical BOM data.

2. Methodology

This study estimated the average bill of materials for 25 common categories of consumer electronics products using a combination of empirical analysis via product disassembly and physical material identification and measurement and external validation via literature benchmarking. Product categories (Table 2.1) were selected for study based on high ownership rates in U.S. households and prevalence in the electronic waste stream (Althaf, Babbitt, and Chen 2019). Within the 25 product categories analyzed, a total of 95 individual products were disassembled, spanning a wide array of model years, product designs, and functional attributes (Table 2.1). These products were primarily obtained opportunistically or by request from donation events and e-waste recycling firms, although some were purchased as used devices from online resellers.

Table 2.1 List of 25 product categories analyzed

Product category	Data points from lab (products disassembled)	Years covered by lab data	Data points from literature	Years covered by literature data
Basic mobile phone	9	1998 -2010	0	-
Blu-ray player	3	2006-2012	0	-
CRT monitor	0	-	6	1990*
CRT TV	0	-	3	1987*
Desktop – integrated	1	2011	0	-
Desktop – traditional	1	2009	9	1985-2010
Digital camcorder	1	1998	2	Unknown
Digital camera	8	2002-2010	2	Unknown
Drones	4	2013-2016	0	-
DVD player	3	2004 -2005	4	Unknown
E-reader	2	2010-2014	2	2001-2010
Fitness tracker	6	2012-2014	0	-
Gaming console	3	2005-2006	3	Unknown
Laptop	16	1999- 2011	0	-
LCD monitor	2	2006-2008	10	2009*
LCD TV	1	2009	12	2002-2008
LED TV	1	2016	1	2011
LED monitor	2	2014-2016	0	-
MP3 player	5	2004-2010	1	2009
Netbook	3	1998-2008	1	2009
Non-smart thermostat	2	2011- 2015	0	-
Smart thermostat	2	2011-2015	0	-
Plasma TV	0	-	8	2002*
Printer	5	1999-2009	5	2001*
Smartphone	12	2004-2015	0	-
Tablet	2	2011-2014	1	2009
VCR	1	1990	6	1986-2002

Note: * Denotes products from published sources that have incomplete or uncertain information regarding production date. The year stated is the best approximation by those studies or by these authors based on model details or other specifications given.

2.1 Collecting lab-scale bill of materials data via disassembly

A standard disassembly procedure was designed based on examples of BOMs in the literature (Teehan and Kandlikar 2013; Huisman et al. 2008; Huisman 2003; Kozak and Keolelan 2003; Stobbe 2007; Socolof et al. 2001; Hikwama 2005; Oguchi et al. 2011; Chancerel and Rotter 2009; Peeters et al. 2012; Lee and Hsi 2002; Streicher-Porte et al. 2007; Townsend et al. 2004; "California Department of Toxic Substances Control. Determination of Regulated Elements in Discarded Laptop Computers, LCD Monitors, Plasma TVs and LCD TVs" 2004) and followed to ensure consistent data collection across multiple researchers who contributed to the disassembly dataset. The process of disassembly started by recording the mass of the full product assembly. The full weight included product power cords if they were affixed to the product (as opposed to detachable). Subsequently, the product was disassembled to its major assemblies, which were assigned a unique number and description. The number and organization of unique assemblies varied by product, depending on the complexity of the product's design and the logical way in which its internal components could be grouped.

For example, a tablet (Figure 2.1) was disassembled into five assemblies: battery (lithium-ion battery cells and associated connectors), motherboard (includes PCB), display (includes flat panel glass, cover glass, display bezel, PCBs, plastic films, and other connectors), casing (back cover including camera lens), and interior parts (includes small PCBs, and miscellaneous metal and plastic parts). Screws and other small parts from the same major assembly were grouped and weighed together. On the other hand, smartphones were observed to have more streamlined designs that could be described within two assemblies: main body (includes motherboard, interior parts, and battery) and display (includes flat panel glass, cover glass, plastic films, bezel, and other connectors).

Figure 2.1 Example of product disassembly for a tablet (Samsung Galaxy Tab 4 SM-T530, 2014), illustrating the disaggregation achieved via lab disassembly and physical identification and measurement of representative assemblies (underlined terms), components, and materials.

Each of the major assemblies was weighed and then disassembled as far as possible with physical separation techniques (hand and power tools). Ideally, disassembly led to parts that were comprised of a single type of material, which could be classified as copper, steel, aluminum, other metals (typically magnesium), glass, or plastics (Figure 2.1). These classifications were made based on visual inspection, physical properties, manufacturer labels, and recycling codes. Metal identification was verified using a Delta handheld XRF analyzer (Model DP-2000CC, >99% accuracy for Fe and Al and 95% accuracy for Mg, determined by repeated measurements using a reference alloy with known composition). For example, metals were first tested for ferrous content using a magnet. If magnetic properties were not observed, the metal is assumed to be either stainless steel or aluminum, and then verified with XRF. Copper was primarily identified based on visual inspection (e.g., copper wiring), and magnesium was identified using manufacturer label (parts stamped with a label indicating “Mg”) and verified with XRF. The small fraction of material that could not be classified into these material types, including paper films, rubber, adhesives, or epoxies, was classified as “others.”

The disassembly process also resulted in components that were composites of multiple materials that were partially or totally inseparable by physical means alone. For example, LCD display

modules could be separated to the point where some materials were individually identifiable, such as the display frame (plastic or steel), the polarizer and optical films (plastic and paper/others), and in some cases a tempered glass cover (other glass). However, the flat panel glass itself is a composite made up of multiple layers and materials, including a glass substrate, liquid crystal layer, transparent electrode, and other films, which were not further separable. Components like lithium-ion batteries and printed circuit boards (PCBs) themselves contain many of the same materials reported in the BOM, such as aluminum, copper, steel, and plastic, as well as other elements, including gold, silver, cobalt, and lithium, all of which would only be separable by chemical or thermal techniques that are outside the scope of this study. Thus, the total mass of the component, at a point where it was no longer separable by physical disassembly, was recorded and reported in the BOM. As a result, the total amounts of individual materials in the BOM only represent the content of those materials present in a distinct, separable form in the product. The reported mass of components may include additional amounts of those materials and other elements that are not reported here but that can be estimated by connecting this study with literature that has reported elemental concentrations, such as the mass of precious metals contained in PCBs (Wang and Gaustad 2012) or the mass of indium contained in flat panel display glass (Boundy, Boyton, and Taylor 2017).

All of the above mentioned mass measurements were collected using three balances, which were selected according to the size and weight of the part or material being weighed: 50 kg capacity (Acculab bench scale, model SVI-50C with 5 g resolution), 30 kg capacity (Measuretek high precision counting scale, model EHC-CF-30, with 1 g resolution), and 200 g capacity (Fisher Science compact balance, model CLF201, with 0.1 g resolution). The final mass of all the assemblies, and their respective sub-assemblies, components, and materials were compiled into a BOM for each product.

2.2 Collecting literature bill of materials data

Because some BOM data already exist in the open literature, available sources were collected and assessed for potential to include in the BOM datasets (Table 2.1). One challenge was that literature BOM data are often presented in varied formats, according to the purpose of the study for which the material data were collected. Therefore, selection of literature sources (Teehan and

Kandlikar 2013; Huisman et al. 2008; Huisman 2003; Kozak and Keolelan 2003; Stobbe 2007; "California Department of Toxic Substances Control. Determination of Regulated Elements in Discarded Laptop Computers, LCD Monitors, Plasma TVs and LCD TVs" 2004; Socolof et al. 2001; Hikwama 2005; Oguchi et al. 2011; Chancerel and Rotter 2009; Peeters et al. 2012; Lee and Hsi 2002; Streicher-Porte et al. 2007; Townsend et al. 2004) to include alongside empirical data was based on three parameters: traceability, level of detail, and category consistency. Traceability refers to the degree of transparency in an article's methodology with respect to how product disassembly and BOM construction were carried out, or in other words, the ability to trace reported material composition data back to methods as they were explained in the paper. Level of detail refers to the degree of disaggregation in the reported data, ranging from studies that only report final cumulative mass percent (low detail) to detailed component-level disassembly data (high detail). Finally, category consistency refers to the degree of similarity between the material categories considered in this study and those reported by the published sources. For example, some literature BOMs report "metals" content as opposed to breaking this down into specific types of metals (steel, aluminum, copper). Each parameter is rated as high, medium, or low depending on the published source (Table 2.2).

Based on this assessment, one of three scenarios was typically observed, which determined how the literature data were treated and whether they were ultimately included in the final average BOM values (Table 2.4).

Scenario 1: Literature reported a transparent product disassembly methodology, fully detailed bill of materials with major component assemblies and subassemblies. For example, (Teehan and Kandlikar 2013) manually disassembled fourteen different products following a methodology similar to that used in this work. Complete BOM were reported, including information on model number and year. In cases like this, the literature data could be directly aligned to the primary BOM data sets with no or minimal adjustments (e.g., aggregating material compositions at a product level).

Scenario 2: Literature provided a transparent product disassembly methodology, but the reported BOM are only partially detailed or reported in a different format, and thus required processing for consistency with the primary disassembly data. For example, a study by the California Department of Toxic Substances ("California Department of Toxic Substances Control.

Determination of Regulated Elements in Discarded Laptop Computers, LCD Monitors, Plasma TVs and LCD TVs" 2004) also used direct disassembly of 19 products to find composition of major component assemblies.

Table 2.2 Assessment of literature BOM data sources

Reference	Traceability	Level of detail	Category consistency
AEHA reported in Oguchi et al (2011)	low	low	low
California Department of Toxic Substances Control. Determination of Regulated Elements in Discarded Laptop Computers, LCD Monitors, Plasma TVs and LCD TVs (2004)	medium	medium	medium
Chancerel and Rotter (2009)	medium	medium	medium
Hikwama (2005)	high	high	high
Huisman (2003) via Huisman et al. (2008)	medium	medium	high
Huisman et al. (2008)	medium	medium	high
JEITA reported in Oguchi et al. (2011)	low	low	low
JOGMEC reported Oguchi et al. (2011)	low	low	low
Kozak and Keolelan (2003)	high	high	high
Lee and Hsi (2002)	medium	medium	high
MoE reported in Oguchi et al.(2011)	low	low	low
Oguchi et al. (2011)	low	low	low
Peeters et al. (2012)	low	low	medium
Socolof et al. (2001)	medium	medium	high
Streicher-Porte et al. (2007)	medium	medium	medium
Stobbe (2007)	high	high	high
Teehan and Kandlikar (2013)	high	high	high
Tohoku Bureau of ETI via Oguchi et al. (2011)	low	low	low
Townsend et al. (2004)	low	low	medium

Based on the goals of that study, only the mass of major components (PCBs, LCD panels, and fluorescent bulbs within LCD lighting) were detailed, and no distinctions were made between types of metals present in the products. To align these data with the BOM dataset, some minor processing was required, such as disaggregating the "total metal" category into specific metal categories (aluminum, copper, steel and other metals) according to the percentages observed for similar products in the primary disassembly data. This assumption was based on empirical observation of consistent relative contributions of specific metals within most product categories.

Scenario 3: In studies where reporting a BOM is not the primary goal of the research, material data may be presented without a full explanation of methods or compositional breakdown. For example, Oguchi et al. (2011) provided a comprehensive study on the characteristics of end-of-life electronics as a potential source for metals recovery. Because the study goal was quantifying

the metal content present in electronic products, there was less focus on other materials, such as plastics or glass. As a result, the published material data do not sum to 100% of the product mass. In these cases, the partial data are listed in the BOM datasets with the missing mass percentage composition assigned to the "other" category. Because these data have a fundamentally different structure, they cannot be compared directly to the primary disassembly results and are not included in final average mass compositions reported.

Subsequent to these determinations, literature values that reflected scenarios one and two above were included in determining average material compositions for each product category as follows

$$\text{Average material composition} = \frac{\sum_{i=1}^{n} PD_i + \sum_{j=1}^{N} LD_j}{n+N}$$

Where PD_i is the material composition for each product "*i*" disassembled in the lab, and *n* is the number of products disassembled in that product category. LD_j is the material composition for each product "*j*" taken from literature, and *N* is the number of products taken from literature for the product category. The final calculated averages are shown in the data record described below and the summary BOM (Online-only Table 1).

3. Results and Discussion

The BOM datasets are available at figshare (Babbitt et al. 2019). These data records are compiled in two Excel workbooks containing BOM data collected and organized at different levels of aggregation, corresponding to the ways in which researchers might need to access this information. First, the "Disassembly Detail" workbook provides resolved material and component data at the level of each major assembly and subassembly. Each worksheet represents a single product category, and most categories contain detailed data for multiple product samples (Table 2.1). An example of these results is provided here (Table 2.3) for the tablet pictured in Figure 2.1, demonstrating how the disassembly and material identification processes were translated into an assembly-level BOM

Table 2.3 Example of assembly-level BOM data for a tablet (Samsung Galaxy Tab 4 SM-T530, 2014), illustrating how data are presented in the Disassembly Detail" workbook. Mass data are in grams.

Material	Material and mass (g) breakdown by component:					Total material mass
	Casing	Display	Battery	Interior parts	Motherboard	
Aluminum						
Copper				2.2		2.2
Steel		20	0.3	6.1		26.4
Plastic	65	43.5		41.3		149.8
Li-ion battery			125			125
PCB		7.3		2.4	26.2	35.9
Flat panel glass		60				60
CRT glass						
Other glass		90				90
Other metals						
Others	1	1.4		0.2		2.6
Total component mass	66	222.2	125.3	52.2	26.2	491.9

Second, the "Product Bill of Materials" workbook provides total mass and mass percent of each separable material and component for all products studied and a mean, maximum, and minimum mass (g) and mass percent (%) for each product category calculated using the lab data points.

The workbook also contains literature values, which were collected, evaluated, and processed according to the methods section reported above. If available, assembly-level literature BOM data are included, however, it was more common to find published data presented as mass percentages for the product as a whole. The qualitative analysis of data from published literature is indicated next to each data point. An example of these results is provided here (Table 2.5) for the tablet category, which included two disassembly-based data points and one high-quality literature data point, all of which were reflected in the product category average BOM. A summary table containing the product-level average BOM values is shown in Table 2.4

Table 2.4 Average bill of materials of consumer electronics

	Mass in percentage										
Product category	**Al**	**Cu**	**Fe**	**Plastic**	**Li-ion Battery**	**PCB**	**Flat Panel Glass**	**CRT glass**	**Other glass**	**Other metals**	**Others**
Basic mobile phones	5.6	2.1	1.0	34.0	24.9	18.2	8.1		--	--	6.0
Blu ray disc player	0.2	3.5	59.8	17.5		17.6			0.4	--	1.0
CRT monitor	1.0	4.2	9.3	16.0		7.5		59.3	--	1.6	1.2
CRT TV	0.4	3.4	5.4	16.9		5.7		64.2	--	1.7	2.4
DVD players	1.1	4.1	48.6	33.2		12.8			--	--	0.2
Traditional / Integrated desktop	8.7	3.9	52.2	20.9		9.9	1.0		1.2	0.9	1.2
Digital camcorders /cameras	12.0	1.3	15.5	40.6	2.4	15.6	4.1		1.7	--	6.9
Drones	0.7	2.5	10.8	50.8	6.5	14.2	0.8		--	--	13.6
E-readers	6.1	0.4	4.6	32.1	20.7	12.1	16.2		5.3	2.20	0.3
Fitness trackers	3.2	1.2	18.7	18.1	7.2	6.1	2.9		--	--	42.5
Gaming consoles	9.1	2.2	31.3	34.5	0.10	16.6			--	0.04	6.2
Laptops	15.4	1.8	11.5	28.3	14.0	12.4	8.2		--	5.8	2.4
LCD monitors	6.2	5.3	35.8	28.4		6.2	17.9		--	--	0.2
LCD TV	2.5	0.9	42.6	28.1		5.8	12.9		--	4.7	2.5
LED monitors/ LED TV	13.6	0.02	27.9	39.8		4.1	14.2		--	--	0.3
MP3 player	25.5	0.7	13.8	15.4	11.6	14.4	7.7		9.4	--	1.5
Netbook	10.0	1.4	5.5	29.1	17.7	16.1	11.3		2.2	4.3	2.5
Printer	0.2	0.6	30.0	60.8		3.1	0.1		3.8	--	1.4
Smart phone	9.4	1.2	6.3	23.2	22.6	14.0	8.9		7.9	2.5	4.1
Smart thermostats	1.9	0.2	8.9	54.1	1.7	21.6	5.4		--	--	6.3
Tablet	9.3	0.4	4.1	19.5	22.8	6.6	14.8		21.8	--	0.6
VCR	--	1.5	64.6	18.6		15.3			--	--	--

Note: Averages include disassembly-based lab data and high-quality literature data. Cells left blank to indicate that material or component does not apply to the product. For example, CRT glass is only found in CRT TVs and monitors and is not expected in other product categories. Cells containing "—" indicate that material was not directly detected or quantified, although it may be present in the product. For example, the total reported mass of a

PCB (component) would likely include individual materials present in that component (aluminum, copper, gold) that are not detectable or separable by physical disassembly alone. Likewise, stated values for each material reflect only the mass of that material when it is separable, identifiable, and quantifiable by physical disassembly and not additional amounts of that material that may be contained in composite components (i.e., materials are not double counted).

Table 2.5 Example of product-level BOM data for the tablet category, illustrating how data are presented in the Product Bill of Materials workbook.

Material categories	**This study**				**Literature** (Teehan and Kandlikar 2013)		**Average mass %**
	Samsung (2011)		**Samsung (2014)**		**Apple iPad (2009)**		
	Mass (g)	**Mass %**	**Mass (g)**	**Mass %**	**Mass (g)**	**Mass %**	
Aluminum	45.0	8.0	--	--	137	20.0	9.3
Copper	3.9	0.7	2.2	0.45	1.1	0.16	0.4
Steel	28.1	5.0	26.4	5.4	12.5	1.8	4.1
Plastic	127	22.7	150	30.5	36.4	5.3	19.5
Li-ion battery	135	24.1	125	25.4	129	18.8	22.8
PCB	54.4	9.7	35.9	7.3	20.1	2.9	6.6
Flat panel glass	55.0	9.8	60.0	12.2	154	22.5	14.8
CRT glass							
Other glass	110	19.6	90.0	18.3	188	27.4	21.8
Other metals	--	--	--	--	--	--	--
Others	1.5	0.27	2.6	0.53	7.5	1.1	0.6
Total mass (g)	**560**		**492**		**686**		

Note: Cells shaded gray indicate that the specified material is not applicable to this product. Cells with "--" indicate that the specified material was not detected by physical separation of the product.

In Table 2.4, Table 2.5, and in the Product Bill of Materials workbook, ‘zero’ values could result for different reasons, which were conveyed by different cell formatting. Cells shaded gray indicate that a material is ‘zero’ because it is not applicable to the product. For example, CRT glass is found in CRT TVs and monitors, and lithium-ion batteries are found in mobile products, but these components are not expected to be present in other product categories. Cells containing '--' indicate that a material was not detected by the stated disassembly and material identification process, but we cannot rule out the potential that it is present in the product within a composite component or in an undetectable concentration (e.g., as an additive, alloy, tramp element, or contaminant, etc.). For example, the total reported mass of a PCB (component) would likely include individual materials present in that component (including, for example, aluminium and copper) that are not detectable or separable by physical disassembly alone. However, stated values for each material reflect only the mass of that material when it is separable and

quantifiable by physical disassembly, and do not include additional quantities potentially contained in non-separable components (i.e., materials are not double counted).

4. Technical Validation

Data were validated using quality controls within the study (internal validation) and by best available benchmarks to product market information and literature values (external validation). One aspect of validation was evaluating if the disassembly and material identification methods were implemented without errors or variations that may introduce uncertainty to the results. In part, such uncertainty was mitigated by using a standard procedure and instruments (balances, XRF) with sufficient resolution and accuracy for the size and type of measurements made (see instrumental specifications in the Methods section). This uncertainty was also assessed by identifying data points that could be re-evaluated using multiple estimations. Specifically, the total product mass was determined prior to disassembly (for most products) and then re-estimated by summing the masses of individual materials and components after disassembly. Variability between these two estimates would point to small parts or materials lost to disassembly or inaccuracies in instrumentation. Data in the "Uncertainty Analysis" workbook, also posted to the figshare repository, demonstrates that the percent difference between these two mass measurements was about 0.5% on average, with a maximum of 2.5% for a single product.

To validate these measurements against external references, product mass estimated as the post-disassembly sum of material and components was also compared to reported weights from manufacturers, where such information could be obtained for the same make, model, and year product as studied in the lab. These comparisons showed about 1% difference on average, with a maximum of about 10% difference between values. It appeared that the few products with greater differences may be due to exclusion of the power cord in the BOM mass. Because many of the products disassembled were obtained from the e-waste stream, peripheral items like cords were not consistently available, and so they were excluded unless they were affixed to the product. Thus, final mass values may underestimate total mass in cases where a detachable power cord is sold with a product but not captured in the BOM. Other small discrepancies may represent uncertainty associated with disassembling products that may have been customized or upgraded after purchase, which would influence the final weight. However, these cases were few, and the majority of mass estimates were very close to available product specifications,

providing additional confidence that products disassembled in the lab represent realistic models of the product, as described by the brand or third party verified websites.

The above approaches to validation are limited to total product mass, as no comparable internal measurement was available for repeating estimates on individual material identification or mass. However, disassembly and material identification data could be validated against literature sources if a comparable product were available. From the dataset, two products were identified as being very similar to both our lab data and a high quality literature study (Teehan and Kandlikar 2013): an Amazon Kindle from 2010 and an Apple iPod Touch 8 Gb from 2008/2009. The products were close but not exact matches, as the Kindle described in the literature was a third-generation model of the original design and the one disassembled was a first iteration of a slightly altered design. The iPods were identical in make and model but were potentially manufactured in different years. The lab study identified the iPod to be from 2008, based on the date stamped on the case; the literature only reported it as 'circa 2009'. However, these are the most similar options available to provide BOM validation.

Side-by-side comparisons of the BOMs for both products are included in the Uncertainty Analysis workbook. For the Kindle, total product mass reported in both BOMs differed by only 1.7g (0.8%) while mass reported for specific assemblies varied by +/-5 g or less, typically due to small differences in how parts were assigned (e.g., assigning screws to 'interior parts' vs. 'casing' assemblies). The mass contribution by specific materials and components were also highly consistent, barring one exception, where this study found an approximately 20g internal backplate to be steel (verified by magnetic properties and XRF) and the literature study assigned it as aluminum based on the lack of magnetic properties. The discrepancy is likely due to small design or manufacturing differences between the two models. For the iPod, total product mass reported in both BOMs differed by only 1.6 g (1.4%) while mass reported for assemblies was +/-1 g or less. The mass contribution by specific materials was extremely close between the two BOMs, with the biggest variability (3.8 g) stemming from this study assigning the plastic frame surrounding the flat panel screen to the "plastics" category, while the literature study included it in the LCD display category.

While this detailed level of comparison was not possible for all products, as no other model and year overlap was found, the two examples provided show high agreement, indicating that the

methods of disassembly and material identification were robust. However, it should be noted that the applicability of reported BOM findings to studies involving current electronic products will depend on the similarity of product designs and the extent to which technology has evolved over time. Many of the products included in this data set are older models, currently being discarded. As such, they are good representations of materials and components now found in the e-waste stream, but not necessarily generalizable to new technologies being manufactured and sold currently. For relatively well-established technologies, the overall material composition has been shown to remain relatively constant over time, particularly once a specific design and form factor is established in the market (Kasulaitis et al. 2015). For emerging technologies, materials are not yet well understood and will require additional study and BOM characterization. However, the framework for disassembly, material identification, and measurement presented here can be adapted to collect additional data for new products as they become available.

5. Conclusion

This chapter created BOM data for 95 unique consumer electronics products capturing the material and component profile. The novelty of this chapter is that it provides a transparent and reproducible database for a wide cross-section of technologies representing various designs and time periods. The BOM data are essential as it provides material content per product, which is the first step in conducting water footprint assessment. These BOM data are also helpful for various stakeholders, including, recyclers, policymakers for conducting research and creating solutions to improve various sustainability aspects of consumer electronics. Findings of this chapter indicate that metals and plastics are the major contributors towards the mass of materials present in consumer electronics, suggesting that these materials will have a major impact on water resources. Now that material profile on metals used in consumer electronics is established, next chapter dives specifically into life cycle of metals understand water use and building life cycle databases to conduct water footprint assessment.

Chapter 3 Opportunities for reducing the supply chain water footprint of metals used in consumer electronics

(Note: This chapter is an adaption from a previously published journal article "Madaka, H., Babbitt, C.W., Ryen, E.G., 2022. Opportunities for reducing the supply chain water footprint of metals used in consumer electronics. Resources, Conservation and Recycling 176, 105926. https://doi.org/10.1016/j.resconrec.2021.105926")

1. Introduction

Chapter 2 established the BOM data of consumer electronic products capturing data on material content. These material data can then be used to identify specific water inputs for impact assessment. BOM data showed that metals dominate the material profile of consumer electronics, as a result, metals are expected to have a major impact on water resources. Next step is to review the life cycle of metals in order to understand where and how water consumption and degradation impact occurs to build inventory data. Then, this inventory data can be connected to the impact assessment models to quantify freshwater scarcity and pollution impacts of extraction and production processes of metals.

Water is a critical input in metal extraction and refining process for activities such as grinding, floatation, gravity concentration, medium separation, and hydrometallurgical processes (Gunson et al. 2012; Haggard, Sheridan, and Harding 2015; Northey et al. 2016; 2014). Water usage in these processes depends on various factors, such as mining site, type of ore being processed, processing techniques, and the local climate (Glaister and Mudd 2010; Gunson 2013; Haggard, Sheridan, and Harding 2015; Northey et al. 2016; 2014). For instance, pyrometallurgical processing of copper ores consumes 91 m^3 of groundwater per tonne of copper, whereas hydrometallurgical processing consumes about 70 m^3 per tonne (Northey et al. 2014). Water is also used in auxiliary activities such as dust suppression, cooling, washing equipment, and human consumption in the mining sites (Northey et al. 2016). Furthermore, mining is an energy-intense process, and water is required at multiple steps in the energy life cycle: from drilling and extracting fossil fuels to boilers, cooling towers, and emission control systems at thermoelectric power plants (Mekonnen, Gerbens-Leenes, and Hoekstra 2015). In the U.S., for example, thermoelectric power plants are responsible for over 40% of all U.S. water withdrawals, 3% of which is ultimately lost and not returned to surface water systems (Dieter et al. 2018).

Not only do metals used in the electronics sector deplete local watersheds, but they also result in downstream water degradation. Acid main drainage, release of heavy metals and processing chemicals, or improper management of mine tailings may result in potentially harmful emissions to surface and groundwater systems (Gunson 2013; 2012; Northey et al. 2016). An analysis of water samples surrounding the gold mines in the Lower Pra Basin of Ghana suggest that the release of heavy metals, including cadmium, mercury, and copper from gold mining processes, is responsible for polluting ground water resources (Dorleku, Nukpezah, and Carboo 2018). Further, metal extraction and processing often take place in water-scarce regions, putting pressure on local water resources. For instance, in the past decade, China, one of the major producers of various consumer technology metals (U.S. Geological Survey 2020), has experienced severe water scarcity driven by rapid industrialization (Wang, Zhong, and Iceland 2017).

Potential water impacts from mining are compounded by increasing population (Liyanage and Yamada 2017), expansion of agricultural activities (Parris 2011), and climate change impacts such as altered weather patterns, droughts, and flooding (Gosling and Arnell 2016; Haddeland et al. 2014). These external factors often lead to resistance to mining projects from local communities. One example is seen in Peru, a water stressed country in South America. The Conga mine in the Cajamarca region, Peru, had been planned for the production of gold and copper, which are widely used in consumer electronics; however, the project was shut down due to public opposition (Jamasmie 2016). This resistance was mainly due to planned open pit mining, which risked endangering lakes and wetlands in the region that were the main source of water for the public (Jamasmie 2016). Addressing these sustainability issues and preserving water resources will require a greater understanding of how water-intense sectors, like mining and manufacturing in the consumer electronics sector, contribute to water quantity and quality risks across different regions.

Given the social, economic, and environmental concerns surrounding water impacts, a growing body of research has begun quantifying the water footprint of consumer electronics. Case studies have been carried out to quantify the life cycle volumetric water consumption for personal computers (Alafifi 2010), printed circuit boards (Alcaraz Ochoa et al. 2019), lithium-ion batteries (Gong et al. 2018), and semiconductors (Cooper and Pafumi 2010). When considering

the full product life cycle, material extraction contributes about 10% to total volumetric water consumption (Alafifi 2010), but may actually be responsible for disproportionately high regional water stress impacts because mining and refining processes often take place in water stressed regions, rather than regions where water is abundant (Cooper et al. 2011; Frost and Hua 2017). To our knowledge, the spatial variability in water stress impact for consumer electronics materials has not yet been analyzed. Further, materials with low volumetric water consumption could ultimately have higher degradative impacts to water quality. Studies have shown a link between material supply chains and impacts associated with water systems, including eutrophication, ecotoxicity, and acidification, in case studies on televisions (Song et al. 2012), desktop computers (Duan et al. 2009), mobile phones (Moberg et al. 2014), and RAM components (Liu, Lin, and Lewis 2010). By combining the analysis of water consumption and degradation impacts, there is an opportunity to analyze consumer electronics materials more holistically and then identify strategies to minimize supply chain impacts.

Therefore, this research applies life cycle-based water footprint methodology to assess the potential freshwater quantity and quality impacts associated with the extraction and production of metals used in consumer electronics. This study is carried out at three levels: first analyzing water impacts for specific metals individually, then for the metals contained in two representative case study products, and finally for the electronics sector in aggregate. The goal is to identify material "hotspots" and then analyze scenarios under which impacts can be reduced, including supply chain shifts, use of recycled content, and material substitution. The insights provided by this research are intended to guide product designers and manufacturers towards reducing environmental impacts of producing consumer electronics.

2. Methodology

The study analyzed water footprint of metal supply chains in the electronics sector, following life cycle assessment (LCA) methods. These methods are carried out in four steps: the definition of study goal and scope, a life cycle inventory to quantify water inputs and emissions, life cycle impact assessment to quantify the resulting potential for environmental damage, and interpretation according to study goal. Each of these steps is detailed further in the following sections.

2.1 Goal and scope

The goal of this research is to assess potential freshwater quantity and quality impacts of metals used in consumer electronics. The study is intended to inform multiple stakeholders who might use results in different ways, including product design, manufacturing, supply chain management, policy development, and materials recycling. Because these audiences would require information at varied scales, the results are ultimately normalized according to three functional units: 1) Per kg of metal produced; 2) Per product; and 3) Per the global electronics sector. The system boundary reflects a cradle-to-gate approach, because materials studied find varied use in a wide array of electronic products. Thus, component manufacturing and assembly, product use, and end of life phase are not included in the scope of the study, although this introduces potential limitations that are discussed later. This study specifically focuses on metals because they account for the major share of a typical electronic product's mass composition (Babbitt et al. 2020) and because metal mining and production takes place in many geographical regions, and as a result, variability exists in water consumption and discharges. The system boundary (Figure 3.1) includes the physical transformation of metal ores into mineral concentrates (mining and concentration), further transformation into mineral products and intermediates (purification), and subsequent conversion into the final metal or alloy (refining). Input flows are the freshwater withdrawals associated directly with metals processing and indirectly with energy generation and upstream chemical and material processing. Output flows are the pollutant emissions that are directly linked to potential water quality degradation through available life cycle impact assessment methods (see Section 2.3).

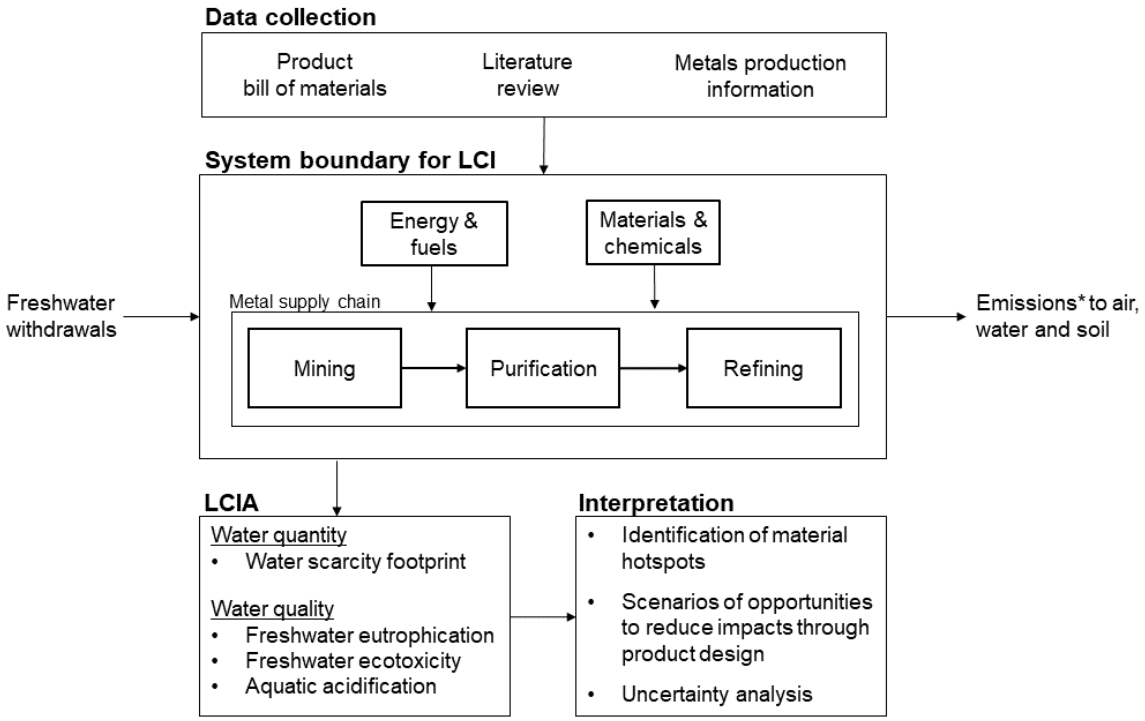

Figure 3.1 Life cycle framework adopted in this study to quantify water consumption and degradative impacts of consumer electronic material supply chains. *The specific emissions and impacts analyzed by the study are detailed further in the Section on Life Cycle Inventory.

2.2 Life cycle inventory

The life cycle inventory was developed in two phases: first, consumer electronic product bills of materials (BOMs) were used to determine specific metals that are of importance to the consumer electronics sector broadly and that are found within specific case study products; and second, production process data were compiled to model water inputs and emissions associated with the processing steps and upstream inputs required to produce all of the identified metals.

2.2.1 Individual materials

Common materials used in consumer electronics were first identified using BOM data collected in Chapter 2 (also Babbitt et al. (2020)) , and then augmented with data that detailed composition of complex components such as printed circuit boards (PCB), display units, and batteries (Buechler et al. 2020; Cucchiella et al. 2015; Işıldar et al. 2018; Wang and Gaustad 2012). Consumer electronics metals identified were categorized as base metals, precious metals, critical

metals, rare earth elements (REEs), and hazardous metals (Table 3.1). Due to lack of LCI data, a limited set of materials including vanadium and certain REEs (those other than lanthanum, praseodymium, and neodymium) were excluded.

Table 3.1 Classification of consumer electronic metals considered in the scope of the study

Base	Precious	Critical	Rare earth elements	Hazardous
Steel	Gold	Manganese	Lanthanum	Lead
Aluminum	Silver	Gallium	Praseodymium	Mercury
Copper	Platinum	Indium	Neodymium	Chromium
Nickel	Palladium	Cobalt		Cadmium
Magnesium	Rhodium	Lithium		
Zinc		Tantalum		
Titanium		Tin		
		Antimony		
		Barium		
		Tellurium		

2.2.2 Case study products

To evaluate how individual material impacts contribute at the product scale, the smartphone and laptop were used as case studies. These devices have high ownership rates and prevalence in the e-waste stream (Althaf, Babbitt, and Chen 2021; 2019). BOM data collected were used to quantify the mass of specific materials for multiple product models representing different brands and manufacturing years (Table A1). BOMs included bulk materials such as steel, aluminum, and copper, as well as complex components, including PCBs, batteries, and displays (Babbitt et al. 2020). The mass of individual metals present in those components was determined from a compilation of literature (Bizzo, Figueiredo, and de Andrade 2014; Boundy, Boyton, and Taylor 2017; Guo et al. 2011; Hagelüken 2006; Oguchi et al. 2011; Szalatkiewicz 2014; Vats and Singh 2015; Wang and Gaustad 2012; Yamane et al. 2011; Yazıcı et al. 2010) compiled by Althaf, Babbitt, and Chen (2021), Tan et al. (2017), Sahan et al. (2019) and Buechler et al. (2020). The data on individual metals are compiled in Tables A2-A9. The complete summary of product material composition is provided in Table A10.

2.2.3 Global electronics sector

To evaluate water impacts of material consumption more broadly, data were also collected on the total global use of metals in the electronics sector. Current global metal production data for the electronics sector were collected from U.S. Geological Survey (2020) and Graedel et al. (2015), and summarized in Table A11. For broader context, these data represent the entire electronics sector, which includes consumer electronics, electronic appliances, and other electric and electronic equipment, including as electronic components found in other products (e.g., the motor of an electric vehicle).

2.2.4 Production processes

Material data were linked to water consumption and emission release using mining and production process data in SimaPro v8.5 using the ecoinvent v3.5 database ("Allocation, cut-off by classification" system model). Ecoinvent data use generic water flows that are allocated to different countries based on the best available knowledge of the location of the unit processes (Classen et al. 2009). The baseline case of material production was modelled according to the average global production market mixes with primary content alone (no recycled content). See Table A12 for the list of process blocks used from ecoinvent, and Figure. A1 for an example process block modeled. Scenario analyses described in Section 2.4.1 were used to model variability around these initial assumptions, including geographic variability and production from secondary sources. While the LCI data used include a wide range of resource inputs and emission releases, this study specifically focused on those with a direct linkage to water impacts discussed in Section 2.3. Specifically, the flows quantified were the volumetric consumption of freshwater to each of the production processes and, following ISO 14040 methods, the release of only those emissions to land, air and water that potentially impact water quality

2.3 Life cycle impact assessment

Water consumption and quality impacts were first evaluated per kilogram of each metal produced and then scaled to the per product and per sector functional units according to the amount of each material contained in a smartphone or laptop and globally used in the electronics industry, respectively. Characterization of these impacts are described further in the sections below.

2.3.1 Water scarcity footprint

Water consumption impacts were quantified by the Water Scarcity Footprint (WSF), which represents the product of the inventory flow (volume of water consumed in m^3 per functional unit) and the water stress characterization factors (Eq. 1).

$$Water\ scarcity\ footprint\ (WSF) = Water\ consumption\ (m^3/functional\ unit)\ \times Water\ stress\ characterization\ factors\ (m^3\ equivalents\ /\ m^3 consumed\ ) \quad \text{Eq. 1}$$

Two water stress characterization factors were used: Pfister et al. (2009) and AWARE (Boulay et al. 2018a). The Pfister et al. (2009) water stress characterization factors used here (m^3 eq./m^3 consumed) is based on the ratio of total annual freshwater withdrawals to the hydrological availability of freshwater resources at a country level (Pfister, Koehler, and Hellweg 2009). The AWARE method water stress characterization factors used here (m^3 eq. /m^3 consumed) is based on the inverse of the AMD (availability minus demand), which indicates the relative available water remaining per area in a watershed (i.e., determining the water availability minus the demand of humans and environmental water requirements) aggregated to a country and annual resolution (Boulay et al. 2018a).

These methods were used because they provide country-level characterization factors that allow for an assessment of geographic variability in metal sourcing. The WSF was initially based on the assumption that consumer electronic materials are sourced according to the global average production mix. However, water availability and withdrawals vary spatially, and assuming an average global production mix might under- or over-estimate the impact. To capture this geographic variability, we analyzed WSF across the wide range of producer countries from which electronics metals are known to be sourced (U.S. Geological Survey 2020). The water stress characterization factors of each producer country was estimated again using the AWARE method (Boulay et al. 2018a) and presented for an equivalent volume of water consumption (Table A14). While water scarcity can vary significantly within a country itself, capturing water impacts at this detailed level was not possible due to lack of ecoinvent LCI data on water consumption for processes specific to individual mining sites.

2.3.2 Water quality impacts

Human activities influence water quality in varied ways, from the direct release of contaminants that degrade water resources to the long-term climate impacts of water temperature and aquatic organism health (Khatri and Tyagi 2015). Here, we specifically focus on the impacts to freshwater quality that are directly linked to chemical emissions resulting from metals production. Specifically, we analyzed freshwater ecotoxicity and eutrophication (using ReCiPe midpoint (H) characterization factors; Huijbregts, 2016), and aquatic acidification (Impact 2002+ midpoint characterization factors; Jolliet et al., 2003). While geographic variability in water quality impacts can still be studied through the differences in underlying country-specific LCI data, the impact characterization factors used here represent an average value that does not vary by location.

2.4 Interpretation

Interpretation of results was first carried out to identify material "hotspots" -- those metals contributing the greatest water consumption and degradative impacts per metal, per product, and per sector. Then, scenario analysis was used to evaluate potential opportunities to reduce the impacts of representative material hotspots, as described below. Results were also interpreted through the lens of potential variability and uncertainty in modeling choices and the extent to which these factors may influence the ultimate water impact results.

2.4.1 Scenario analysis

Scenarios were created around potential strategies that might be applied during the design and manufacturing stage for consumer electronic products to reduce water impacts. The three scenarios, described below, were aimed at addressing materials and processes that the baseline results showed to have a significant water impact. While the nature of these impacts is described in the Results section, we note an example here to clarify our approach: in the product case studies, gold and aluminum were identified as water impact hotspots for the smartphone and laptop, respectively. Therefore, scenarios examined how model results might change according to strategies specifically applied for these materials, namely, sourcing materials from alternative supply chains (in the case of gold), substituting with a lower impact material (in the case of

aluminum), or increasing the recycled content (for both elements). The three scenarios are described below.

Alternate supply chains:

This scenario examined the overall impact reduction potential that may be achieved by specifying that gold be obtained from supply chains in regions with lower water stress. The WSF is governed by both volumetric water consumption and the country-specific water stress index. Given available LCI data, it is not possible to estimate how water consumption for mining and metals processing would change due to supply chain shifts, especially for countries not yet modeled in ecoinvent. Thus, the water consumed for extraction and production of 1 kg of gold is assumed to be constant regardless of country. But the water stress associated with that water consumption could be varied according to country-specific AWARE characterization factors. Thus, WSF per kg of gold was calculated as if it were produced solely from one of three countries with low water stress risks: Canada, Russia, and Brazil. These countries currently contribute to the global production of gold and have available resources that could meet demand from the electronics sector (U.S. Geological Survey 2020). The country-specific results for gold, scaled according to mass of gold per smartphone, were combined with the previously calculated contributions of the other elements present in the smartphone (as modeled for the baseline) (Table A24 and Table A25). Results were interpreted on a per smartphone basis and provide a theoretical upper bound on the extent to which a single material supply chain can influence the overall WSF of materials contained within a consumer electronics product.

Material substitution:

This scenario examined the potential impact reduction potential that may be achieved by material substitution during product design. The baseline laptop BOM data showed aluminum to be a common casing material, but other materials are also feasible for this application, including plastic and magnesium (Babbitt et al. 2020). A representative model (i.e., 14-inch laptop) containing 444 g of aluminum in the casing was used as the reference. An equivalent laptop of the same size but with an alternative casing material would contain 336 g of acrylonitrile butadiene styrene (ABS) or 388 g of magnesium, according to example product BOMs (Table A26 and Table A27) and due to different properties of the materials. Other sources of aluminum

identified in the BOM (wiring, battery, hard drive, etc.) were not changed, as these are not easily substituted during product design, a common issue for many electronics materials that provide unique properties and functionality (Babbitt et al. 2021). Results were interpreted on a per laptop basis, relative to the degree of improvement over the reference all-aluminum case.

Material recovery in a circular economy:

Our baseline results captured impacts associated with primary production of metals, representing a worst-case scenario. Circular economy aspirations will require pathways to recycle materials, even for demanding, high-purity applications like electronics. Here, we modelled use of recycled content aluminum and gold recovered from e-waste (Bakas, Herczeg, and Blikra Vea 2016; Bigum, Brogaard, and Christensen 2012) following open loop allocation, cut-off principles (refer to Appendix A section 1.5). According to these principles, the original material extraction and production of a material is allocated to the primary user (i.e., first life cycle), but the processes required to collect, recover and purify a material from the electronics scrap was allocated to the second life cycle (Nicholson et al. 2009). Figure. A2 shows the extended scope of LCI carried out for e-waste recycling processes and the materials recovered. Industry-specific recovery rates were collected from (Bigum, Brogaard, and Christensen 2012; Caffarey 2012; Hagelüken 2008) to estimate water scarcity footprint of 1 kg of a recycled material. Then, WSF of gold and aluminum from both primary and secondary material supply is estimated. Ranges of secondary supply that can theoretically be specified for electronics applications were based on realistic values and optimistic upper bounds of potential recycled content: 45-90% for aluminum and 35-70% for gold (Ashby 2012; Graedel et al. 2011).

2.4.2 Uncertainty analysis

This study relied on a comprehensive database of electronic product BOMs (Babbitt et al. 2020), and aggregate data on material production processes (ecoinvent). Uncertainties in results may stem from data limitations around elemental composition of specific materials contained in composite components like lithium-ion batteries or printed circuit boards. The elemental concentration of a metal in such a component depends on the type, manufacturer, and age of the product, and such estimates are additionally confounded by variability in the empirical methods and instrumentation used to detect and quantify elemental concentrations (Guo et al. 2011; Sethurajan et al. 2019; Yang, Liu, and Yang 2011). For example, the literature sources used to

estimate elemental concentration in a typical smartphone PCB reported gold content ranging from 0.003 to 0.18 % by mass (Oguchi et al. 2011; Sahan et al. 2019; Tan et al. 2017; Vats and Singh 2015) (Table A3 and Table A41). The influence of variability in material concentration on baseline results was assessed again using the example of gold content in a smartphone. Multiple data points from literature were compiled (Table A3) to establish maximum, minimum, and average values of gold content in a smartphone (Table A41). The baseline WSF per smartphone was then re-analyzed across these ranges.

3. Results and Discussion

3.1 Baseline material-level impacts

Results show that when analyzing individual materials, precious metals (rhodium, platinum, gold, palladium) have the highest water scarcity footprint per kg, particularly in comparison to that of base metals (steel, copper, aluminum) (Figure 3.2a). The WSF results calculated with Pfister et al. (2009) water stress characterization factors range from 0.007 m^3 eq. for 1 kg of steel to 93.1 m^3 eq. for 1 kg of rhodium, whereas WSF calculated with AWARE water stress characterization factors ranges from 0.53 to 7518 m^3 eq. for the same metals, which represent the lowest and highest material level impacts. While the absolute values of WSF vary between approaches, the relative rankings of metals in terms of water scarcity impact is the same for both methods. Further, the metals with highest WSF results were also found to represent the largest ecological risks due to pollutants released from metal extraction processes. Specifically, precious metals have the highest impacts per kg for all three water quality impacts quantified: freshwater ecotoxicity, freshwater eutrophication, and aquatic acidification (Figure 3.2b).

Precious metals are typically found in low concentrations in ores, and as such often require higher quantities of water and energy to extract and refine (Calvo et al., 2016; Mudd, 2008). These impacts are anticipated to continue to grow with increased demand for scarce elements in electronics, as resource requirements for extraction and production processes increase with declining ore grade (Calvo et al. 2016; Miranda 2010; Norgate and Jahanshahi 2010). Another factor that influences baseline WSF results is the underlying LCI data used to model water consumption (detailed further using gold as an example in Table A13). Ecoinvent processes for metals are typically constructed using economic allocation methods to partition impacts for

materials created from joint production processes (Classen et al. 2009; Frischknecht et al. 2007). Because precious metals have high economic value, they typically are allocated a greater degree of impacts (Classen et al. 2009), although this is an area that requires future methodological study (Bustamante et al. 2016).

a)

Materials	Pfister et al. (m^3 eq)	AWARE (m^3 eq)
Steel		
Aluminium		
Copper		
Nickel		
Magnesium		
Titanium		
Zinc		
Gold		
Silver		
Palladium		
Platinum		
Rhodium		
Manganese		
Tellurium		
Barite		
Cobalt		
Antimony		
Tin		
Lithium		
Indium		
Gallium		
Tantalum		
Cadmium		
Chromium		
Lead		
Mercury		
Lanthanum		
Praseodymium		
Neodymium		

b)

Materials	Freshwater eutrophication (kg P eq)	Freshwater ecotoxicity (kg 1,4 DCB eq)	Aquatic acidification (kg SO_2 eq)
Steel			
Aluminium			
Copper			
Nickel			
Magnesium			
Titanium			
Zinc			
Gold			
Silver			
Palladium			
Platinum			
Rhodium			
Manganese			
Tellurium			
Barite			
Cobalt			
Antimony			
Tin			
Lithium			
Indium			
Gallium			
Tantalum			
Cadmium			
Chromium			
Lead			
Mercury			
Lanthanum			
Praseodymium			
Neodymium			

Figure 3.2. Heat map showing the comparison of material hotspots identified for a) water scarcity footprint (m3 eq) calculated per kilogram of each metal using Pfister and AWARE impact assessment methods; and b) water quality impacts per kilogram of each metal: Freshwater ecotoxicity potential (kg 1,4 DCB eq), Freshwater eutrophication potential (kg P eq) and Aquatic acidification potential (kgSO_2 eq).

Note: The color scale is based on the relative percentile (90^{th}, 50^{th} and 10^{th}) to which each metal belongs according to its ranking within the impact category (numeric values are provided in the Supplemental Information file). For example, metals in the 90^{th} percentile range are those with the highest impacts and are shown in the darkest color.

However, the underlying processes that contribute to material level WSF results vary by metal, according to the processes in which the greatest amounts of water are consumed. For example, 46% of the WSF for gold is attributed to water consumed directly for mining and production activities, and 36% to indirect water consumption associated with energy production and consumption associated with mining and production activities (Figure 3.3). Fuel and electricity consumption occur widely across the mining and production process, including opening mine pits, crushing, grinding of ores, refining metals, and for general plant operation. Upstream water impacts of producing chemicals and materials needed for mining and process operation contributed 16% of the total WSF for gold, with the remaining 2% associated with other activities (Figure 3.3). On the other hand, aluminum, which is an energy intensive material, owes 84% of its WSF to indirect water consumption associated with energy generation, with only 6% of the impact attributed to water consumed directly for mining and production activities, 6% to upstream water impacts of materials and chemicals, and the remaining 3% to other activities. Conversely, cobalt, a critical metal with moderate water scarcity impacts (in the 50th percentile range in Figure 3.2), has 53% of WSF attributed to upstream water impacts of producing chemicals and materials, 29% due to direct water consumed during mining and production, with the remaining fractions associated with energy production (12%) and other activities (6%) (Figure 3.3).

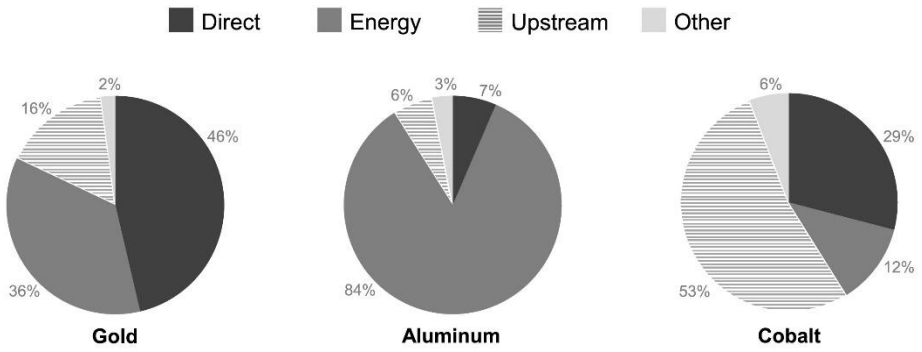

Figure 3.3 Relative contribution to material level WSF results for specific metals, as estimated using AWARE (m^3 eq/kg of each material).
Note: Contributions are disaggregated into four categories: direct (process) water use, water associated with energy conversion and fuel extraction, upstream water inputs (associated with producing chemicals or materials required for the metal's production), and all other types of water consumption.

For water quality impacts, process contribution analysis identified pollutant releases during the treatment of sulfidic mine tailings as being the primary cause of freshwater ecotoxicity and eutrophication impacts (98% of the total impacts for gold, Figures. A5 and A6). For example, eutrophication impact of gold in this study is mainly attributed to the presence of phosphorous in chemicals used to control environmental damage of mine tailings. Water quality impacts are also influenced by fossil fuel extraction and combustion to produce the energy needed for mining processes. These upstream energy systems are a contributing factor for acidification impact across all elements and for the eutrophication and ecotoxicity impact across base metals including copper, nickel, lead, and manganese. Acidification impacts are also attributed to the blasting process used to open mines, which releases nitrogen oxide, carbon monoxide, and ammonia into the environment (this process represents 43% of water quality impacts for gold, see Figure. A7). Results do not include any impact on water quality from the particulates of rock created during blasting, as they are omitted from the LCI database, suggesting a potential opportunity for future study and data collection.

3.2 Geographic variability in baseline material results

The baseline WSF results are also influenced by the global average production mix of metals, due to the spatial variations in water consumption in different mining operations, as modeled by the available inventory data, and regional water demand and availability, as captured by country-specific water stress characterization factors (see Table A13 for further comparisons of these underlying factors). While geographic variation in inventory flows cannot be evaluated extensively due to LCI data limitations, we can analyze variability in water stress. To this end, Figure 3.4 shows the "relative" WSF for 1 m^3 of water consumed for each producer country, thus holding volumetric water consumption constant and allowing for direct comparisons of country-level water stress based on the AWARE characterization factors (Boulay et al. 2018a). The variability shown in Figure 3.4 accounts for freshwater availability within a region and the marginal demand of human and aquatic ecosystems on those water resources. This variability in WSF provides an opportunity to leverage alternate material sourcing decisions that can potentially reduce life cycle water impacts, an opportunity that is explored through scenario analysis in Section 4.1.

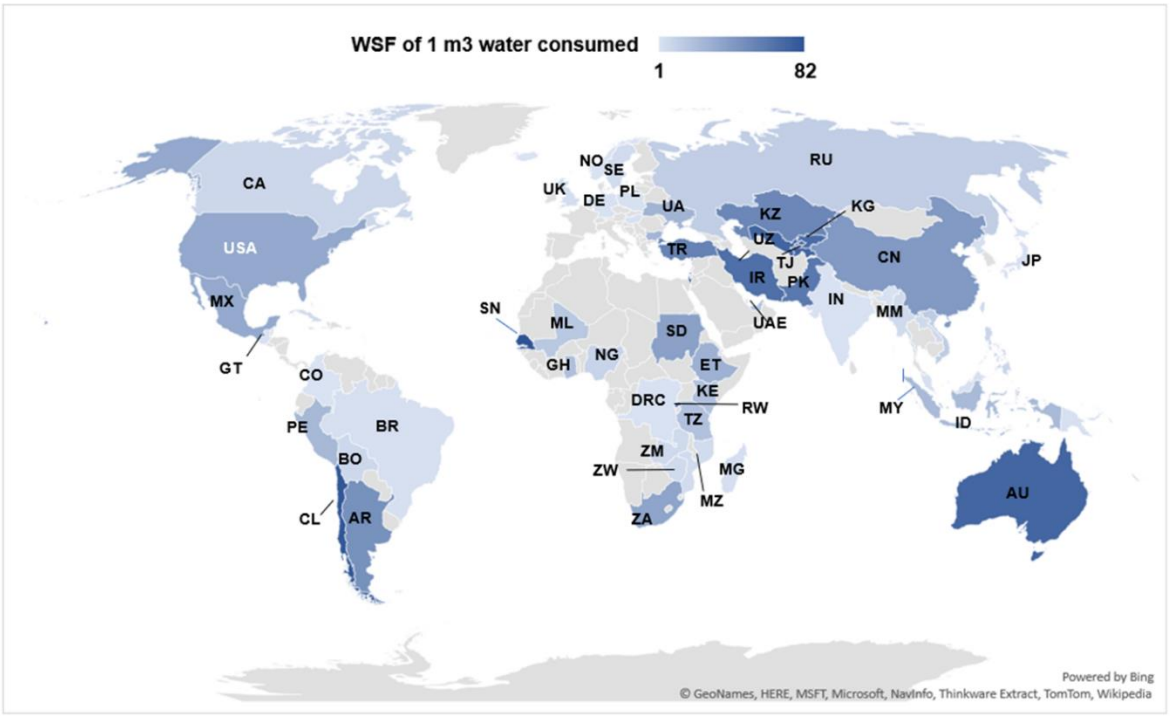

Figure 3.4 Relative WSF (m^3 eq/1 m^3 water consumed) for countries known to produce electronic materials.

Note: Color intensity corresponds to relative magnitude of impacts, where darker shades suggest greater water stress risks. Here, the shading associated with a value of "1" represents the world average WSF, and all darker shades reflect the factor by which each country analyzed is relatively greater than the world average. For example, the country corresponding to the value of "82" has the highest relative impact of all analyzed (82 times greater than world average). Countries shown in gray were not analyzed.

However, these factors do not account for any additional developments in infrastructure that may be required to make water widely available to all its users. As a result, some countries, such as the Democratic Republic of the Congo (DRC), are observed to have low WSF (Figure 3.4), but in fact lack the water infrastructure needed to expand mining in this region, due to under-investment in water systems and conflict-related destruction (Partow 2011). Sourcing metals from the DRC also faces other barriers, such as social and geopolitical risks from the mining industry in this region (Althaf and Babbitt 2021), underscoring the importance of considering broader tradeoffs in supply chain decisions. When changing a supply chain to water abundant countries is not feasible, companies can invest in infrastructure to improve water quality treatment in mining and manufacturing areas. This treatment infrastructure would improve quality and also help tackle water scarcity in a region, thereby reducing overall water impacts of products and benefiting local communities (Damania et al. 2019). Scenario analysis on alternate

supply chains are further explored below to understand if regional variability may be leveraged to reduce the impacts of materials in consumer electronics.

3.3 Baseline product-level impacts

Results discussed so far focus on the water impacts per kg of each material. However, materials are used in widely varying amounts within common electronics products. On a per product level, material hotspots (Figure 3.5) had both similarities and distinctions from those identified above, depending on the product considered. In the case of a smartphone, precious metals gold and palladium collectively contributed nearly 45% of the total WSF (0.25 m^3 eq. for all metals in the product), followed by aluminum (28%), and lithium (6%). For the laptop, on the other hand, base metals aluminum and copper collectively contributed almost 45% of the total WSF (4.23 m^3 eq. unit for all metals), followed by palladium and gold collectively (31%), and cobalt (10%). The WSF of metals per product is influenced by both the magnitude of water consumption per material and the mass of that material contained in the product. For example, base metals have a small WSF impact per kg, but they contribute almost 45% of the WSF impact of a laptop because they comprise a significant fraction (>25%) of this product's mass (Figure 3.6). On the other hand, precious metals account for almost 45% of the WSF of a smartphone because the magnitude of impact per kg (see Table A22, and Figure A8) offsets the relatively low mass (0.4% of an average smartphone). Precious metals were also observed to be the dominant contributors for all three water quality metrics, for both the products analyzed (Figure 3.5). As material trends in consumer electronics products continue to evolve, the anticipated hotspots will likely vary in the future. For instance, the overall concentration of gold in electronics is declining with changes to product design (Althaf, Babbitt, and Chen 2021; Kasulaitis, Babbitt, and Krock 2019). As a result, the contribution of gold to WSF and other environmental impacts may similarly change in the future.

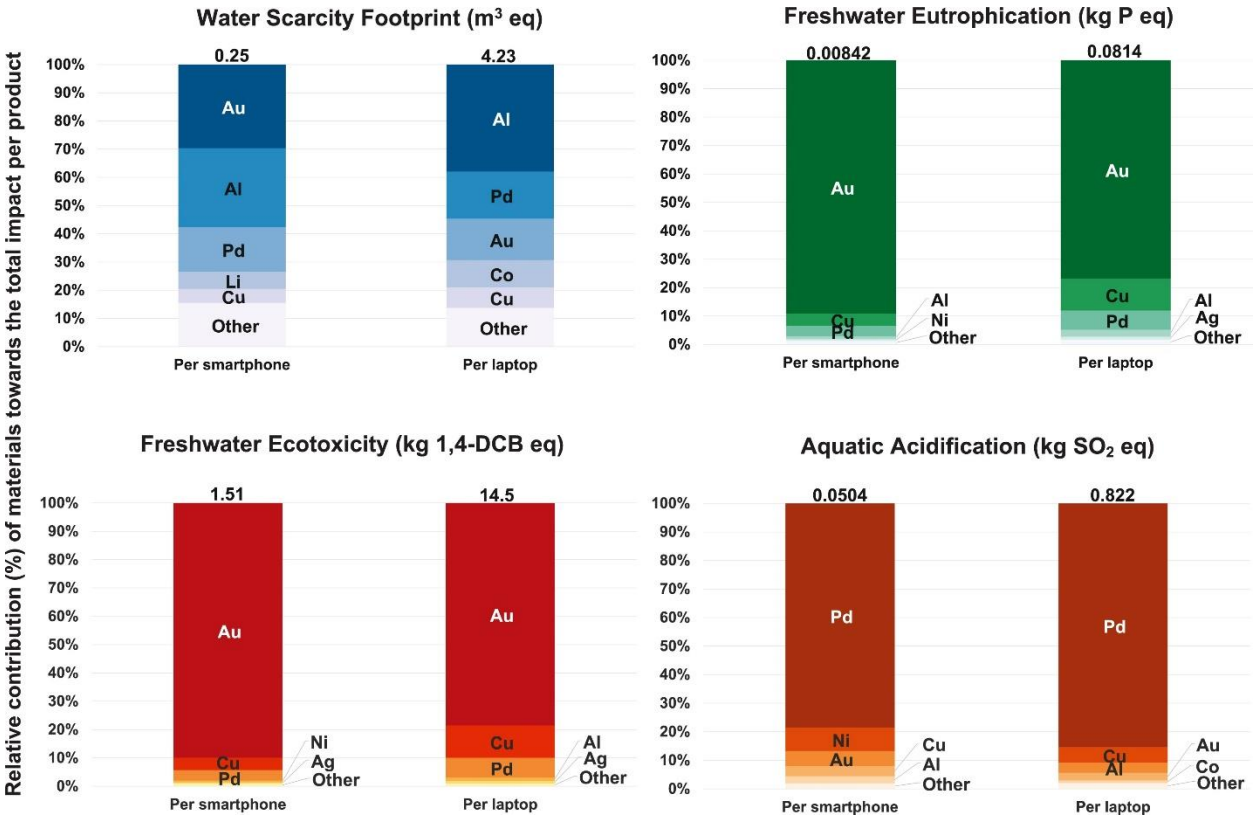

Figure 3.5 Relative contribution of metals per smartphone and per laptop towards the total water scarcity footprint (m^3 eq), freshwater ecotoxicity (kg 1,4 DCB eq), freshwater eutrophication (kg P eq), and aquatic acidification (kg SO2 eq) for all metals per product.

Note: The value on top of each bar shows the impact for all metals cumulatively in a product. The relative magnitude of a laptop is higher than a smartphone due to overall mass of a product. Note that for all the impacts, the contribution of the top five metals is shown, and the rest of metals are aggregated as "Other." The color intensity corresponds to the relative degree to which each metal contributes to the total on a ranked basis.

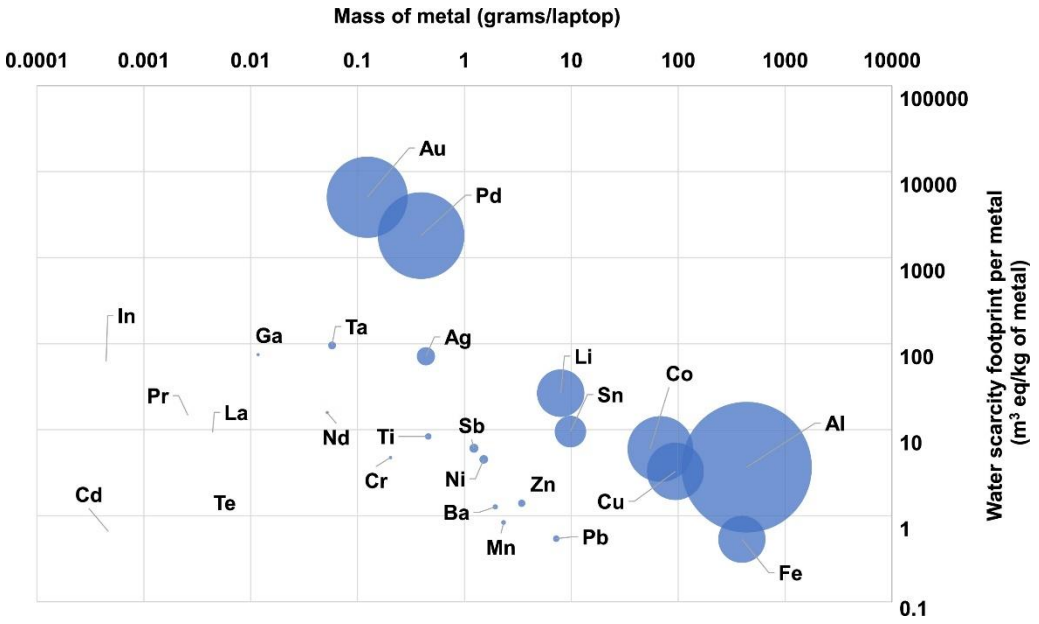

Figure 3.6 Comparison of material concentration in a laptop (g/laptop) and WSF for each metal (m3/per kg metal).
Note: Both axes are presented on a log scale. The circle size corresponds to the material's overall contribution to the total WSF for all laptop materials (from 1 to 100%). A similar analysis is provided for the smartphone in the Supplemental Information file.

Beyond the materials discussed thus far, other metals to note are lithium and cobalt, which are also observed to be key contributors towards WSF of a smartphone and laptop, respectively (Figure 3.6 and Figure A8). These elements are used in the electrodes of lithium-ion batteries found in mobile devices, and their use is expected to rapidly grow due to both the continued adoption of electronics and the projected demand for electric vehicle batteries (Fu et al. 2020). Both metals present unique challenges to water scarcity impacts. Globally, the majority of lithium is currently produced via brine extraction (Liu, Zhao, and Ghahreman 2019) and primarily extracted in Chile, in particular Salar de Atacama, which holds almost 30% of world's lithium resources (Agusdinata et al. 2018). However, this region is already one of the driest places on earth, and increasing demand for lithium is leading to overexploitation of water resources (Liu and Agusdinata 2020). Further, brine extraction may emit pollutants from evaporation pools into the water supply, leading to increased water quality impacts and exacerbating tensions between the local communities and mining companies (Liu and Agusdinata 2020). On the other hand, more than 60% of world's cobalt is sourced from the DRC, where concerns have grown about the interaction of mining activities and socio-political

vulnerabilities (Rustad, Østby, and Nordås 2016). Added impacts from water extraction and pollutions may aggravate existing social concerns over cobalt extraction in vulnerable regions (Kemp et al. 2010; Olivetti et al. 2017). Increased demand for cobalt and lithium should be met in ways that reduce both water scarcity impacts and interconnected impacts to society.

3.4 Baseline sector-level impacts

Finally, to provide broader context on water impacts from electronics materials, WSF and quality metrics were also estimated relative to the current background situation of global metal use in the electronics sector. In the case of WSF, base metals (Fe, Al) contribute nearly 80% at the sector level (see Figure. A4), primarily because they are used in larger products and appliances (Işıldar et al. 2018; Morf et al. 2007), which are not distinguishable from consumer electronic devices in the underlying data (U.S. Geological Survey 2020; Graedel et al. 2015). For water quality impacts acidification, base metals again contributed >75% of the cumulative impact for acidification, but for eutrophication and ecotoxicity, trends show that gold, along with base metals, are materials of concern (Figure. A4), largely due to the emissions released during mining and mine tailings management. The variation in results across the scales of analyses suggests that a wide array of strategies will be needed to alleviate water impacts, including product design by strategies discussed in the following section, lifespan extension for components containing elements with high individual impacts (Babbitt et al. 2021; Cordella, Alfieri, and Sanfelix 2021), product remanufacturing (Ardente et al. 2018; Nasr et al. 2018; Quariguasi-Frota-Neto and Bloemhof 2012), and increased commodity recycling (Hertwich et al. 2019; Menikpura, Santo, and Hotta 2014). However, any evaluation of such solutions would require an expanded analysis beyond the estimation presented here, as they would likely involve large-scale sectoral shifts involving ‘non-marginal’ changes in material use and the corresponding demand for water resources in different basins or countries (Pfister et al. 2017).

4. Interpretation and Scenario Analysis

Results presented thus far have been interpreted and compared internally, by identifying relative material hotspots for different water resource impacts and different scales (per material, product, and sector). These results can also be interpreted using the context provided by other studies, although much of the existing literature has focused on carbon and energy footprints, rather than

water. But, even using studies on other metrics as a point of reference, we do find parallels in results. For example, this study demonstrated that precious metals have the largest impacts to WSF and water quality at the material level, a finding that is mirrored in studies on global warming potential (GWP) and cumulative energy demand (CED) of material production (Nuss and Eckelman, 2014). The similarity in findings can be traced to the fossil fuel-based energy used during material extraction and refining, which contributes to climate, energy, and water impacts. This interconnection demonstrates the importance of an energy-water nexus perspective on material supply chains (Elshkaki 2019) and highlights the importance of transitioning to energy sources that have low carbon and water impacts.

At the product level, the ranking of material hotspots from WSF results are generally comparable to materials of concerned identified from analysis of carbon and energy impacts for a smartphone (Yu, Williams, and Ju 2010; Ercan 2013) and a laptop (Deng, Babbitt, and Williams 2011). While there are small differences in the absolute ranking of materials when evaluating different environmental metrics, these can also be attributed to variability and evolution in product design and material choice (Kasulaitis et al. 2015). Water quality impacts have also been quantified as a part of broader LCA studies for both laptops (Ciroth and Franze 2011; Grzesik-Wojtysiak and Kukliński 2013) and smartphones (Moberg et al. 2014). However, the approach in this study disaggregates impacts to determine specific material contributions, whereas past work typically treats material impacts collectively, since the studies' scopes often include the full product life cycle. This difference represents a key opportunity for future study, to extend results presented here to understand how material impacts – and the solutions to reduce them – relate to water impacts during product use and end-of-life management.

To further interpret results, three scenarios were analyzed to determine how findings may vary according to material supply chain and design changes, particularly in light of potential solutions for reducing the material and product impacts observed here. These scenarios focus on strategies that could be employed during the design and manufacturing stage: sourcing materials from lower WSF countries via alternate supply chains, using recycled materials to displace primary production, and substituting high-impact materials with alternatives expected to have lower water impacts. These scenarios were explored for two case study metals, gold and aluminum, which were identified as hotspots in the results described above.

4.1 Alternate supply chains

This scenario examined the potential to reduce water scarcity by sourcing materials from different regions, thus leveraging the geographic variability in water scarcity impact discussed in section 3.2 (Figure 3.4). Here, gold is used as a case study, due to its wide production geography and high WSF and quality impacts. When gold is modeled as being produced solely from Canada, Russia, or Brazil, the net WSF of materials per smartphone is reduced by 19-28% (Figure 3.7). These results reflect a change in gold supply chains alone, while other materials are held constant (modeled according to their global production mix as calculated in the Baseline scenario). Optimizing the production mix of other materials may theoretically reduce the estimated WSF further, but benefits are likely to be less significant, because other materials contribute less to WSF results and are characterized by lower supply chain variability. Note, however, that this scenario may also serve as a theoretical maximum of impact reductions, since supply chains are unlikely to so dramatically shift to a single country unless significant economic benefits overcome logistical barriers (Yokoi et al. 2021).

Further, limiting material sourcing to the one or a few of the lowest impact countries may create tradeoffs in economic and social aspects, such as job loss in producer countries or reduced supply chain diversity, which can leave material resources vulnerable to geopolitical disruptions (Althaf and Babbitt 2021). Alternatively, these results can also be interpreted as a call for a multi-stakeholder engagement to invest in sustainable water management systems, which can help build resilience for the future. Further, the electronics industry should integrate water into their business strategy, develop suitable metrics to evaluate water risks, engage in addressing shared water risks by raising awareness in their supply chain, and contribute to policy development that supports investments in water management solutions (Hoekstra 2014; Morgan 2018; Stuckenberg and Contento 2018). While these integrated strategies may take longer to bear results than others discussed below, they may ultimately be more scalable that fundamentally altering material extraction infrastructure, which faces bottlenecks to scale-up, particularly for critical materials (Olivetti et al. 2017).

4.2 Material substitution

Compared to other strategies, product design changes may offer more immediate potential for reducing water impacts. One such opportunity is substituting lower impact materials for those

hotspot materials used in large quantities in products, such as the aluminum used in laptop casing. Aluminum was observed as a hotspot in the laptop in our baseline scenario, but bill of material data (Babbitt et al. 2020) show that functionally equivalent casings can also be made with other materials, including plastic (typically acrylonitrile butadiene styrene (ABS)) and magnesium. Substituting either of these materials for aluminum casing results in at least a 17% reduction in the overall impact of materials in a laptop (Figure 3.8). This decrease in impact is driven in part by a reduction in net mass: for the same 14-inch laptop, magnesium casing weighs 13% less than the aluminum casing, and plastic casing weighs 24% less (Table A26).

This decrease is also attributed to lower WSF per kg for the alternate materials. The WSF of aluminum is 34% greater than that of ABS and 48% greater than that of magnesium. Both aluminum and magnesium are energy intensive processes, however, process contribution analysis showed that for aluminum, nearly 84% of the total impact is attributed to electricity production, which only contributed 66% to the net WSF of magnesium (Figure 3.3 and Figure. A9). These differences can be traced to the energy grid mix associated with global material production. For aluminum, the majority of electricity input is sourced from hydropower, while fossil fuel sources are used more in magnesium production (Classen et al. 2009). While fossil sources certainly create greater water quality impacts, further investigation into the underlying inventory data found that hydropower electricity generation for aluminum was mainly modelled on power stations with reservoirs having significantly higher evaporative losses when compared to the water losses associated with the same amount of energy produced from coal combustion (see Figure. A10), a dichotomy that was also shown in previous studies (Mekonnen, Gerbens-Leenes, and Hoekstra 2015; Zhao and Liu 2015). However, in addition to water footprint of energy, climate change impacts should be assessed collectively, as aluminum industry switched to hydropower to mitigate climate change (Yu et al. 2021). Further, given that climate change is one of the major factor influencing the water availability in a region (Gosling and Arnell 2016; Haddeland et al. 2014), future studies should develop impact assessment models that can capture this phenomenon to evaluate tradeoffs between hydropower and fossil fuel sources. Additionally, diversifying the renewable energy portfolio by investing more in solar and wind that have lower water consumption than hydropower (Mekonnen, Gerbens-Leenes, and Hoekstra 2015) can also be a potential option to reduce both climate change and water impacts.

While material substitution may offer more immediate reduction potential, it also introduces new tradeoffs that should be considered further. Substituting plastic for aluminum may reduce the upstream WSF impact of materials in a laptop, but it introduces the need for chemical additives such as brominated flame retardants, which may pollute water resources if emitted to the environment at product end-of-life (Chen et al. 2012). Further, material choices may have cascading impacts into other life cycle stages. For example, the high energy footprint of aluminum means that it is a costly metal to produce from primary resources alone (Sverdrup, Ragnarsdottir, and Koca 2015), creating an economic motivation for its recycling that is evidenced by global average recycling rate of 76% (Bloxsome 2020). Plastic, on the other hand, has higher costs to recycle and sees limited recovery in the current market (Di et al. 2021; Sahajwalla and Gaikwad 2018), which may negate initial WSF savings. Further, consumer attachment to electronics, and thus their willingness to repair and extend the product lifespan, is driven by both functional and emotional durability, both of which are thought to be enhanced by durable metallic casings (Lobos and Babbitt 2013). While the water footprint of lifespan extension is not evaluated here, it may see benefits from reduced resource extraction and waste generation as well as impacts from added energy – and likely water consumption – required over a longer use phase. For some materials, substitution is not effective or practical, as seen in the case for gold. While copper or silver are lower WSF alternatives, neither provides equivalent quality or functionality as gold wiring in printed circuit boards (Gan and Hashim 2015; Graedel et al. 2015).

4.3 Material recycling

The baseline results capture WSF impacts associated with primary material content, but material recovery in a closed-loop circular economy presents tremendous opportunity to reduce environmental impacts of material supply chains. This opportunity is explored for both gold and aluminum. For gold, increasing the secondary content from 0% to 35% (realistic estimate of average recycled content) results in a 10% reduction of the overall WSF of materials in a smartphone (Figure 3.7). Further increasing the recycled content of gold to 50% or 70% reduces WSF proportionally (by 15% or 20%, respectively), but would require overcoming critical barriers now facing the electronics recycling industry. Such challenges include products that are not designed for disassembly or recycling (Tansel 2017), low recycling rates (Forti et al. 2020),

and low collection rates of electronics after consumer use (Shittu, Williams, and Shaw 2021; Tesfaye et al. 2017).

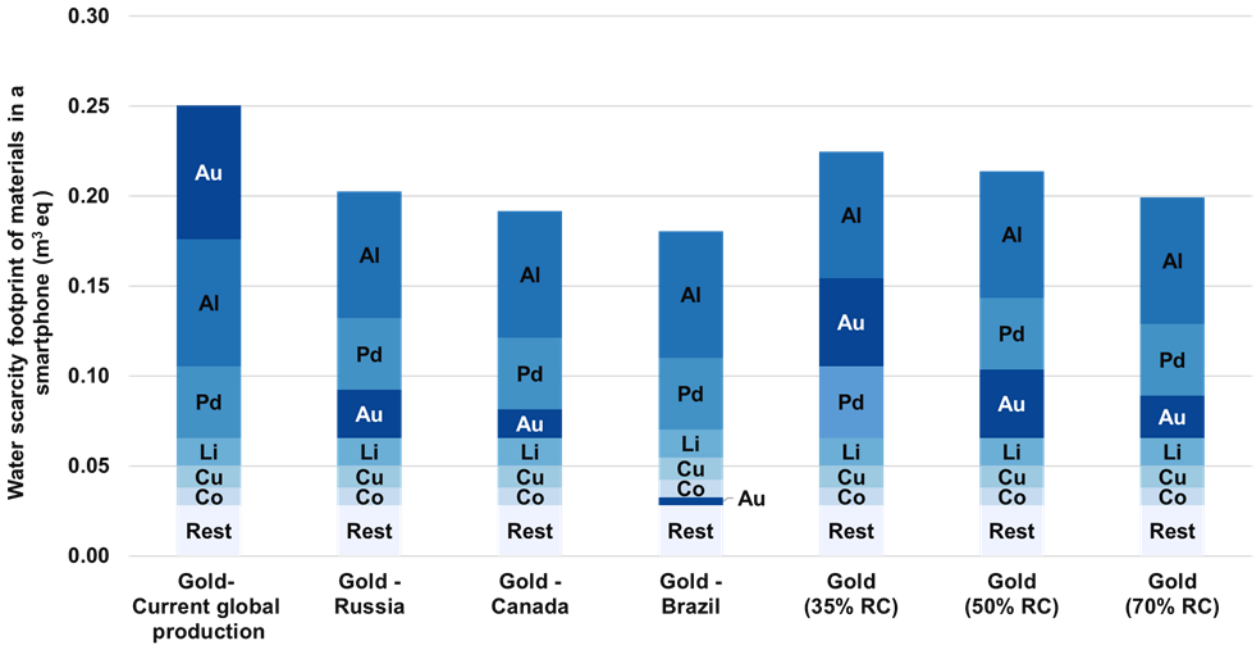

Figure 3.7 Potential opportunities for reducing WSF (m^3 eq) impact per smartphone: alternate supply chains and material recycling scenarios applied to gold as a representative case.
Note: The contribution of gold is represented in the darkest blue color. The six elements with the highest individual contributions are shown in each bar, with "Rest" including the contribution of all other metals in the smartphone.

For aluminum, increasing the recycled content of aluminum in laptop casing to 45% (current estimate of average global recycled content) reduces the overall WSF of materials in a laptop by 11% (Figure 3.8). Doubling this recycled content (to 90%) can reduce the total WSF of metals in laptop by up to 22%. While this target is ambitious, current products on the market achieve this level and suggest broader feasibility (e.g., Macbook Air laptop, Apple, 2020). While the benefits of using recycled content aluminum are lower than those for substituting aluminum with plastic or magnesium, it may offer a more realistic pathway, as aluminum is a highly recycled material with a well-established recovery infrastructure. On the other hand, combining material substitution and recycling can also result in greater benefits. For instance, replacing the aluminum casing with magnesium containing up to 90% recycled content could theoretically reduce the overall WSF of materials in a laptop by 30% (Table A40).

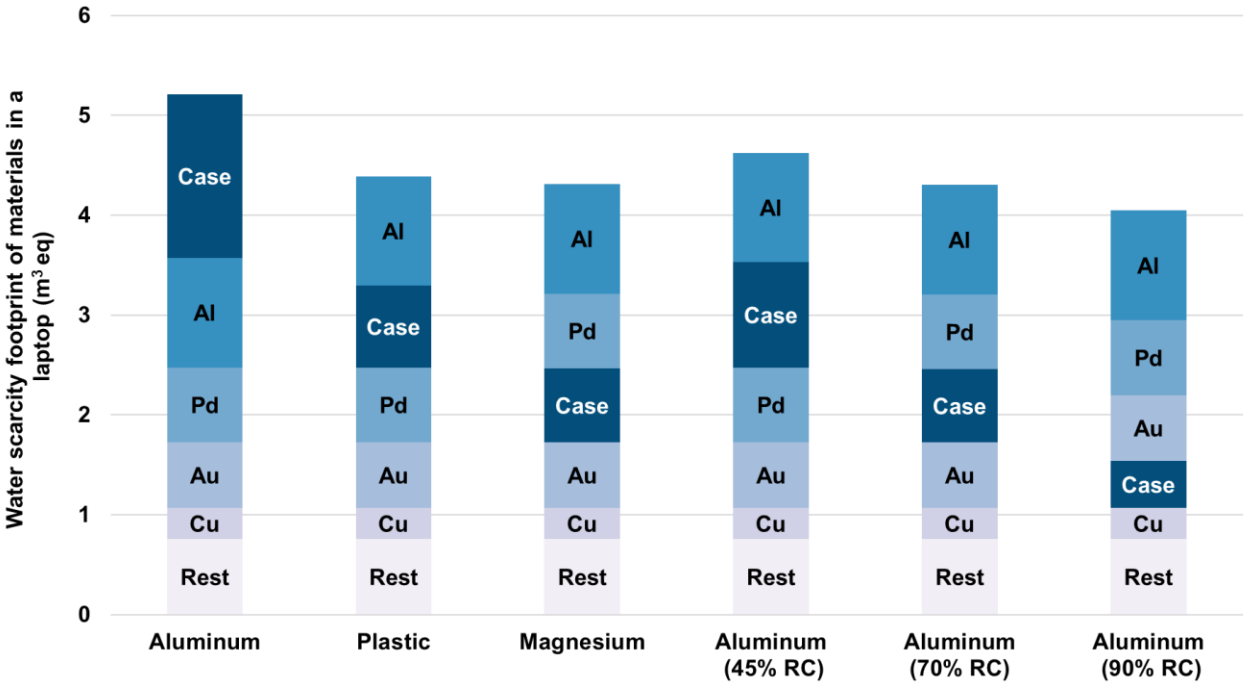

Figure 3.8 Potential opportunities for reducing WSF (m^3 eq) impact per laptop: material substitution and material recycling scenarios applied to aluminum used as a laptop casing material.

Note: The contribution of the select casing material is represented in the darkest blue color, and the specific "Case" material analyzed corresponds with the x-axis labels. Other non-casing uses of aluminum in the laptop are designated with "Al." The five materials with the highest individual contributions are shown in each bar, with "Rest" including the contribution of all other metals in the laptop.

A preliminary analysis was also conducted with available LCI data to estimate if scenarios discussed above provide similar benefits in reducing water quality impacts. Using 35% recycled content of gold in a smartphone reduces both freshwater ecotoxicity and eutrophication by 31%, and doubling this recycled content (70%) reduces the impacts by nearly 63% (Table A33 and Table A34). For aluminum, increasing recycled content of aluminum in a laptop casing only provides about 3% savings for both ecotoxicity and eutrophication, since this material did not have significant impacts in these categories in the baseline model (Table A39). However, all of these estimates should serve as a theoretical maximum of the extent to which impacts can be reduced by recycling, because recycling rate is an optimistic proxy for recycled content and because available LCI data only included pollutant emissions from upstream processes and energy generation, not directly from the recycling process itself. Substituting aluminum in a laptop casing by ABS reduces the total impact of impact of metals in a laptop by only 1 to 2% (Table A38). On the other hand, using magnesium instead of aluminum in a laptop slightly

increases ecotoxicity and eutrophication impacts by the same magnitude, due to the added emissions from fossil energy sources used in magnesium production, as discussed above (Table A38). Scenario analysis underscores the importance of systems-level analysis to identify potential tradeoffs that may arise across environmental impact categories.

4.4 Uncertainty analysis

While this study used comprehensive BOM data, potential sources of uncertainty include inconsistent data on elemental composition of select materials and variability in product design. To capture this variability, minimum, maximum, and average amounts of gold present in a smartphone were estimated by collecting data points from multiple literature sources (Table A41 and Table A42). In our Baseline product-level analysis, using average mass concentration of gold per smartphone, gold contributes 0.0746 m^3 eq (30% of total WSF of smartphone materials). At the highest mass concentration, this contribution more than doubles to 0.171 m^3 eq (49% of total WSF of smartphone materials), while at the lowest mass concentration, gold only contributes 0.003 m^3 eq (2% of total WSF of smartphone materials) (Table A42). For high gold content, the key material hotspots (gold and aluminum) for a smartphone remain the same as in our baseline scenario. However, at the lowest concentration, other materials (palladium, lithium, copper) become dominant contributors to the total WSF. This variability underscores the need to comprehensively analyze material hotspots for electronics, particularly capturing evolution over time, which may demonstrate the shifting role of precious metals as they become more dispersed and diluted across products (Kasulaitis, Babbitt, and Krock 2019).

More broadly, evaluating the water footprint of metal commodities has other general limitations associated with LCA modeling, LCI databases, and impact assessment methods. For instance, volumetric withdrawals and consumption of freshwater for extraction and production processes vary from region to region. The ecoinvent data do not fully capture this variability for metal production processes due to lack of publicly available data on water withdrawals and discharges. In such cases, water consumption per unit output of material is assumed to be the same in all regions where the metal is extracted. Even within the same country, water stress can vary seasonally and between catchments, but this level of disaggregation is not currently captured in LCI data. Such limitations are inevitable, but equally applicable to all materials analyzed here, so the associated uncertainties are not expected to change the underlying findings on material

hotspots. Other key uncertainties requiring future study are the extension of material-focused findings to the water footprint of a full electronic product life cycle. Use and end-of-life management will require additional water inputs and create different emissions that degrade water quality, but these are also subject to immense variability that are beyond the scope of this study, including evolving product lifespans (Babbitt et al. 2009) and product consumption and discard scenarios (Althaf, Babbitt, and Chen 2021; Ryen, Babbitt, and Williams 2015) .

5 Conclusion

This chapter applied BOM data collected in Chapter 2 to identify life cycle water inventory databases and models, and for the first time, evaluated freshwater scarcity and pollution impacts associated with metals used in consumer electronics. First, the water impact assessment was carried out for individual metals, and then for the representative material profile of smartphones and laptop computers to identify hotspots. Results indicate that, on the individual material level, precious metals have the highest water impacts, because of water consumed directly for mining operations and indirectly for energy production, and water degradation attributed to metal emissions during mine tailings management. At the product level, water impacts were influenced by both material intensity and water impacts on per kilogram basis. For instance, precious metals have the highest contribution per smartphone despite being present in relatively low concentrations due to higher impacts on per kilogram of metal produced. On the other hand, aluminum has the highest contribution per laptop because of higher concentration in a product though impact on per kilogram of metal is relatively low. The analysis also suggests that water impacts can be reduced by 19% when metals are sourced from low water scarcity regions and 20% by increasing use of recycled content in a product. Additionally, engagement from stakeholders across the electronics industry is needed to establish sustainable water management solutions, such as supplier engagement to promote water use efficiency, use of low-carbon and low-water energy resources.

This chapter advances the knowledge on water footprint of metals used in electronics and solutions to address these impacts, which was made possible due to BOM data created in Chapter 2 and availability of life cycle inventory data and impact assessment models for mining and refining processes of metals. However, for plastics, it is difficult to conduct this kind of water impact assessment due to the lack of detailed data on these materials and water inventory as well

as impact assessment models. The next chapter address this gap by first establishing data on what polymers are used in electronics and understand how these materials can potentially impact water resources.

Chapter 4 Profile of e-plastics in the U.S. and their potential implications to water resources

1. Introduction

Chapter 2 established BOM data for consumer electronics, and applying this data, Chapter 3 conducted a water footprint assessment of metals using existing life cycle inventory databases and impact assessment models. In Chapter 2, Plastics are also observed to be another significant contributor towards the total mass of electronics. However, there is lack of understanding of what these materials are and how they are used in electronics. As a result, to understand water impacts of plastics, the first step is to establish profile of polymers use in electronics.

Plastic use and pollution are growing environmental issues in the United States. In the last decade, plastics consumption and production has seen an exponential increase due to low production costs and the unique properties they provide (e.g., low weight, chemical resistance, flexibility) (Kan 2021). However, this increase in consumption comes at a cost to the water resources. Plastics production consumes water for extraction, manufacturing, and processing of nonrenewable sources, such as oil, natural gas, coal (Korol et al. 2019). Water is also required to produce energy, which is another critical input for plastics production and processing (Mekonnen, Gerbens-Leenes, and Hoekstra 2015; Korol et al. 2019). Furthermore, plastic waste often ends up in either uncontrolled landfills or elsewhere in the environment. From here, plastics can eventually enter freshwater sources via wastewater flows or tides potentially contaminating water resources (Jambeck et al. 2015; Lau et al. 2020; Lebreton and Andrady 2019; Schnurr et al. 2018; Ogunola, Onada, and Falaye 2018; Ford et al. 2022; Simul Bhuyan et al. 2021; Schwarz et al. 2019).

Recognizing these issues, studies have started quantifying plastic flows to evaluate opportunities for developing effective waste management policies. Research specific to the U.S. focused on plastic flows by a single polymer type (Chaudhari et al. 2021; Kan 2021; Kuczenski and Geyer 2010; Liang et al. 2021; Smith, Takkellapati, and Riegerix 2022) or by end uses (Di et al. 2021; Heller, Mazor, and Keoleian 2020). While these studies have been beneficial to understand the flows on single use plastics (e.g., Poly vinyl chloride (PVC), Polyethylene terephthalate (PET)) used in major sectors, such as packaging and consumer goods, they do not explore plastics use in

other significant sectors, such as consumer electronics. Though plastic waste from the electronics sector represents less than 1% of the total plastic waste fraction (Di et al. 2021), the contribution of electronics sector towards plastic pollution could be higher. This disproportionate impact is because of the nature of plastics used in electronics, both in terms of the diversity of plastics used (Martinho et al. 2012) and the type of materials contained in plastics that may cause elevated ecological impacts if released (Singh, Duan, and Tang 2020).

Consumer electronics are observed to contain more than 10 different types of plastics (Martinho et al. 2012). Some commonly used plastics found in the electronic products, include acrylonitrile-butadiene-styrene (ABS), high impact polystyrene (HIPS), polycarbonate (PC), polypropylene (PP), polystyrene (PS), and blends of polycarbonate and ABS (PC/ABS) (Maris et al. 2015). Additionally, additives, such as flame-retardants (FRs) are incorporated into these plastics to meet the Underwriters Laboratory (UL) flammability standard. There are different types of FRs used in electronics, including halogenated (e.g., brominated FRs (BFRs)), organophosphorus, inorganic and nitrogen based (Haarman, Magalini, and Courtois 2020; D. Yu et al. 2017). Most used in consumer electronics are BFRs, of which legacy ones (e.g., Polybrominated Diphenyl Ethers (PBDEs)) have been restricted by international regulations such as European Union (EU) Restriction of Hazardous Substances Directive 2002/95/EC (ROHS), EU Registration, Evaluation, and Authorization of Chemicals (REACH) and the United Nations Stockholm convention, due to environmental risks and human health concerns (Hennebert and Filella 2018; Wagner and Schlummer 2020). In 2019, Europe expanded the scope of BFRs and imposed a ban on use of any halogenated flame retardants in electronic displays enclosures and stands (European Union 2019). Along with these restricted FRs, other additives, which are also hazardous are used in electronics to provide color, appearance, and strength to the plastics. Some examples include pigments (e.g., Titanium dioxide, Zinc oxide) (Alassali et al. 2019), and various stabilizers or plasticizers (e.g., cadmium, Antimony) (Dimitrakakis et al. 2009).

When electronics are not properly managed at end of life, these plastics and additives can potentially leach into the environment damaging ecosystems and human health (Lyche et al. 2015; Singh, Duan, and Tang 2020).There are numerous pathways through which plastics and additives can potentially migrate to freshwater systems, including emissions from incineration

and recycling activities or leaching from landfills. For instance, studies have shown that burning of plastics containing flame retardants results in the spreading of FRs, toxic furans and dioxins emissions in the environment (Tue et al. 2016; Zhang, Buekens, and Li 2016). Presence of FRs (Choi, Lee, and Osako 2009; Morin et al. 2017), antimony trioxide (Zhan et al. 2020), and heavy metals, such as lead, mercury, chromium are observed in the leachates of landfills with e-waste (Singh, Duan, and Tang 2020). Furthermore, presence of FRs was observed in soil (Cristale et al. 2019; Matsukami et al. 2015), ambient air (Labunska et al. 2013), and river sediments (Li et al. 2019; Zhou et al. 2017) in the areas surrounding e-waste handling and processing facilities. Not just FRs, but presence of microplastics were also observed in the soil around the e-waste facilities (Chai et al. 2020; Labunska et al. 2013; Zhang et al. 2021). As a result, a robust end of life management policies is needed to tackle pollution from the electronics sector.

To keep the plastics from making their way into the environment, there is a need for the U.S. to transition to a more circular economy. Circular systems can keep the plastics in a closed loop system, rather than being discarded after use, ensuring that these plastics are managed in a socially and environmentally aware manner (Fellner and Brunner 2022). This responsible end of life management is enabled by the transition to green product design, the creation of recycling markets and the development of material recovery technologies. Additionally, there is a need for developing effective policies driving multiple stakeholder engagement in the electronics sector to conserve and recover resources (Bocken et al. 2016; Gaustad et al. 2018; Connor et al. 2016).

However, closing the loop for e-plastics in the United States is far from achievable with existing policies and infrastructure. As per conversations with recyclers, it is understood that some e-waste processing facilities would target to sort plastics from housing components of larger products, such as flat panel displays, CRTs, desktops. In many cases, remaining components and other low grade electronic products are sent to automated shredder, which produces a mixture of plastics, metals, glass and other contaminants (Elliott 2017; Riise 2020). Further, some facilities may process this shredder residue to separate plastics and metals fraction, of which plastics fraction in mostly is used for energy recovery (Kang and Schoenung 2005; Kreibe 2012). In some instances, plastic fraction that was separated from products are sold to downstream buyers in developing countries (e.g., China) where manual labor is cheap for sorting these materials to recycle (Elliott 2017; Schumacher 2016). However, in 2017, China introduced “National Sword

Policy" that banned import of scrap plastics, with strict contamination benchmarks (Brooks, Wang, and Jambeck 2018). Furthermore, the Basel Convention has passed a new amendment to regulate the transboundary movements of plastic waste (Wingfield and Lim 2022). Added to these, differences in product scope under state level e-waste policies and lack of national level legislation in the U.S. makes it economically non-viable for plastics recycling domestically. As a result, U.S. is left scrambling for solutions to manage plastic waste from the electronics.

While circular economy may offer a sustainable and safe way to handle plastics from electronics, lack of data on e-waste polymer composition is a barrier for implementation of closed loop systems. Research has evaluated the polymers and additives composition in e-waste (Beccagutti et al. 2016; Y. Chen et al. 2012; Florea et al. 2011; Fontana et al. 2019; Maris et al. 2015; Martinho et al. 2012; Peeters et al. 2015; 2012; Schlummer et al. 2007; Strobl et al. 2021; Wagner et al. 2020). However, these studies are related to Europe where the product scope of e-waste is different along with consumption patterns and regulatory requirements. A U.S.-specific study with detailed analysis of plastic composition by product category and total e-waste streams was conducted in the 2000s (Fisher et al. 2000). However, polymer and additives profile has shifted since then due to technological innovation, design, and policy changes. For example, the electronics industry has shifted towards using more halogen free organophosphorus FRs due to restrictions on selected BFRs (e.g., PBDEs) (Li et al. 2019; Riise 2020). At the same time, manufacturers have switched to PC/ABS, which enables the use of halogen free FRs, rather than using HIPS or ABS to phase out BFRs (Li et al. 2019; Riise 2020). A more recent study characterized the e-plastics profile in electronic shredder residue collected from the U.S. e-waste recycling facility, which indicated the presence of ABS and HIPS that can be recycled (Anderson, Yu, and Chen 2022). Our study aims to expand the scope and include a wide range of products that would capture a complete profile of e-plastics along with shredded residue. This information can be of value for stakeholders in the electronics industry to understand the water impacts as well as potential for implementing circular systems.

This research aims to provide a profile of polymers and additives contained in common consumer electronic products to understand potential linkages to water impacts. In addition, we estimated the flows entering the e-waste stream to understand implications to circular systems planning in order to manage plastic pollution. The study was carried out in two phases: First, the

profile of polymers and additives in each product category are characterized to estimate average composition. Then, this average composition on polymers is paired with national level e-waste flows (Althaf, Babbitt, and Chen 2021) to estimate the polymer flows and additives entering the waste stream in the United States. This study provides the most recent and transparent dataset on polymer use in consumer electronics in the United States, providing a strong scientific base for understanding the potential water impacts, and implications to circular system planning for this stream of materials.

2. Methodology

The study characterized the current use of polymers and additives in the electronics sector to interpret their potential link to impact on water resources. The methods are carried out in four steps: collection of polymer samples, experimental analysis to characterize polymers and additives profile, estimating baseline national level e-waste flows, and uncertainty analysis to capture variability in the polymer composition (see Figure 4.1). Each of these steps is detailed further in the following sections.

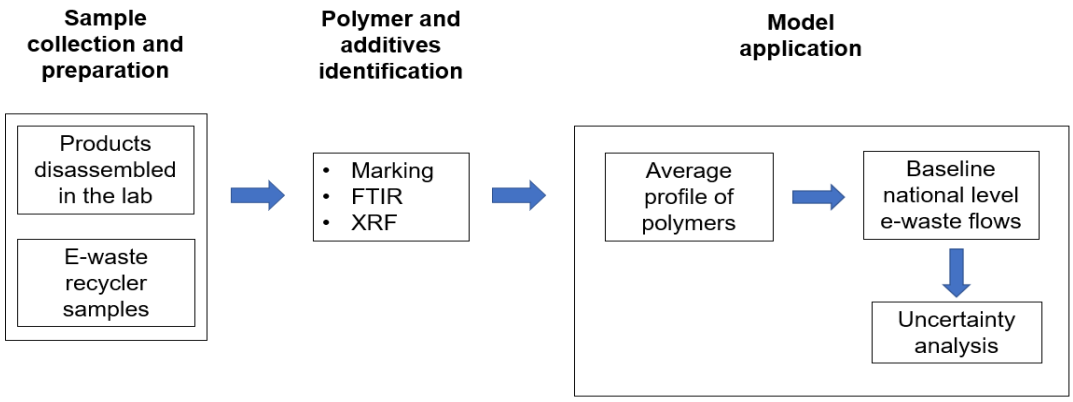

Figure 4.1 Methodological flow followed in this study

2.1 Sample collection and preparation

The study evaluated the polymer samples from the most common categories of consumer electronic products sold to U.S. households, which includes TVs, monitors, computers, audio-visual products, and mobile devices. A representative sample of plastics was collected that represents what might be entering end of life management after use in the U.S. households. This sample included plastic parts taken from products disassembled in the lab (as discussed in

Chapter 2) and provided by a major e-waste recycler in the northeast, USA. The collected samples were characterized based on their material composition, the presence or absence of key contaminants (particularly flame retardants) and their use in common electronic products.

2.1.1 Polymers from product disassembly in the lab

A total of 532 polymer samples representing multiple product categories were obtained from the disassembled products in the lab for BOM data creation in Chapter 2 (also Babbitt et al.(2020)). The products from which the polymer parts were collected represent a diverse range of models, product designs and functional attributes.

2.1.2 Polymers from an e-waste recycling facility

Two different types of samples were collected from a major e-waste facility located in the northeastern U.S. in fall of 2020. The first sample contained large, readily identifiable plastic parts (n=99) that were directly taken from desktop computers, flat panel displays (FPDs), and cathode ray tube (CRT) displays. The parts primarily included exterior casing and acrylic screens (in the case of FPDs). The second sample contained approximately 4912 g of shredded, mixed polymers of unknown origin. These samples were part of the output stream coming from a 5000-ton shredder that the e-waste facility reported was used to process small, low-grade electronic products that they determined were not economically valuable enough to warrant manual disassembly. Each piece was approximately 1 – 2 inch in dimensions. To obtain a representative sample that could be reasonably analyzed in subsequent analysis, five grab samples were taken from the shredded plastics. Each grab sample contained approximately 40-60 individual plastic pieces, for a total sample size of n=230.

2.2 Polymer identification

A sequence of steps was designed to identify and verify the polymer composition of the total plastic sample (including those obtained both from lab-based disassembly and the e-waste recycler). First, polymer samples were examined visually for a marking provided by the manufacturer. A marking generally includes information on the polymer type, flame retardant classification, and any other additives added to the polymer. To confirm visual identification and to further identify polymers in parts with no marking, Fourier transformed infrared spectroscopy

(FTIR) (Agilent technologies) was used, as described in past literature (Schlummer et al. 2007; Stenvall et al. 2013; Strobl et al. 2021; Wagner et al. 2020). Of the total polymer samples characterized, 49% of the total polymer samples were marked with sufficient information to determine polymer type. Of these total samples with marking, 23% were selected for verification using FTIR to ensure that polymer type was labeled correctly. More than 98% of these verified samples were observed to have accurate marking. In the absence of a marking, polymer type was identified using FTIR alone. A small fraction of plastic products (representing less than 6% of the total mass of all samples in a product) could not be analyzed using either method, primarily due to their small size and lack of marking. Spectra collected from FTIR were analyzed in the Know-it-All software. To validate the accuracy of the FTIR instrument and know it all software output, roughly 4% of total polymer samples were analyzed using a different IR instrument (Shimadzu technologies) and Lab solutions IR software. This testing process confirmed that polymer identification process was similar in nearly 60% of the samples verified (see Table B9). A small number of cases were observed, especially with polymer blends and ABS samples, when polymer type in the marking did not match with the spectra software output. These discrepancies could be attributed to limitations of spectrum software, additives, and also to possible plastic degradation (Maris et al. 2015). These conflicts were resolved by applying broader methodology that accounts for the plastic marking, spectrum, software output, and polymer properties. This broader methodology was further applied to fill any data gaps, including identifying any unlabeled polymers. Refer to Appendix B section 1 and section 2 for detailed discussion on resolving conflicts and filling data gaps.

For all the polymer samples analyzed, mass data were recorded using one of the three balances depending on the component mass and size: 50 kg capacity (Acculab bench scale, model SVI-50C with 5 g resolution), 30 kg capacity (Measuretek high precision counting scale, model EHC-CF-30, with 1 g resolution), and 200 g capacity (Fisher Science compact balance, model CLF201, with 0.1 g resolution). Along with mass, color and if possible, the component (e.g., casing, display) to which the sample belonged to is recorded as well.

2.3 Identification of flame retardants and other contaminants

Additives, such as FRs added to polymers to improve flammability (Haarman, Magalini, and Courtois 2020; Yu et al. 2017), are also identified. For each polymer sample with a visible

marking, the FR classification code was recorded if present. Out of total polymer samples characterized, only 34 samples were marked with information on flame retardant classification, of which four were labeled as FR (17) and remaining as FR (40). Of these 34 samples, 88% of the samples with marking were selected to be further verified using a Delta handheld X-ray fluorescence (XRF) analyzer (Model DP-2000CC). While XRF cannot identify a specific type of FR, it provides the relative concentration of bromine, antimony, chlorine among other metals it detects in a polymer. The bromine and antimony concentration together were used as a proxy indicator to identify the presence of halogenated flame retardant. In the past literature, these elements were observed to be present in higher concentrations in brominated flame retardants (Aldrian, Ledersteger, and Pomberger 2015; Wagner et al. 2020). For polymer samples that had no marking, XRF was used again. Nearly 63% (n=516) of total polymers characterized that had no marking were tested for BFRs. Other elements may also be present e-waste plastic samples, due to their use in plastic formulations and polymerization catalysts (Alassali et al. 2019; Dimitrakakis et al. 2009), or due to carryover contamination from previous e-waste recycling processes (Mao et al. 2020b; 2020a). Thus, the concentration of other elements of interest, including chromium, chlorine, lead, mercury, titanium was also captured in this analysis.

2.4 Baseline polymer flows entering e-waste stream

The data collected as described above were then combined with national level e-waste estimates to understand implications to establishing e-plastics circular systems to keep plastics out of water resources. National level e-waste estimates were obtained from the output data from a published material flow analysis (MFA) study developed by Althaf, Babbitt, and Chen (2021). These data represent the household discards of consumer electronic products that are commonly regulated within the U.S. and thus would be expected to be managed at an e-waste facility (similar to the one from which samples were obtained). The national level data report total end-of-life flows of individual products and major material categories contained in those products, including total plastics. Total plastic flows were then disaggregated into specific polymers using the compositional data collected as described above.

This disaggregation is achieved by multiplying total plastic mass outflow in each product category by the average polymer composition of that product category. Average composition is a challenging parameter, given the variability in product designs and their change over time. The

approach here was to first estimate an average composition to estimate baseline polymer flows and test the sensitivity of results to variations that reflected the actual variability observed in polymer samples analyzed from disassembled TVs in the lab. The average profile of polymers for each product category is further described in Appendix B section 3.

2.5 Uncertainty analysis

This study relied on empirical data on polymer composition in electronic products and material flow analysis data. However, uncertainties in results may stem from data around polymer composition data. Polymer composition in a product is highly dependent on the model, age of the product and regulatory requirements. For example, the data from disassembled TVs showed that the primary polymer used in the exterior casing was strongly dependent on the model and age of the product. For example, early displays primarily contained HIPS, which was later replaced by ABS, but as the electronics industry shifted toward bromine-free flame retardants, the casing material similarly shifted to polymers that were best suited for these material additives, namely PC/ABS blends (Riise 2020). Given the significant mass represented by display products, this example was specifically investigated to determine how variability in casing polymers might change the overall plastics flow. Data points representing specific TV models with casing components made of ABS, HIPS, and HIPS plus PC/ABS were identified in the disassembled products (see Table B10). For each of these scenarios, the final polymer flows entering the e-waste stream was estimated and compared to our baseline scenario to understand how trends changed (see Table B11).

3. Results and Discussion

3.1 Average profile of polymers in the e-waste

The results first demonstrate the wide variability in polymers contained in e-waste, as determined by characterization of polymer samples (n= 861) from disassembled products and from components and shredder residue from an e-waste recycler. In total, 23 different types of polymers were observed (see Table B2 for the number and type of polymers observed in each product category). Most unique polymer types (n=11) are observed in printers, which also contain the most plastic (on average 60% by mass) compared to other product categories (see Table 4.1). Other larger products, such as CRT displays, FPDs, and desktops contain plastics up

to 30% of the total mass (see Table 4.1), are observed to contain up to 7 unique types of polymers. Smaller electronics, including media and gaming and laptops that contain on average 30% of plastics have 9 unique polymer types. Mobile products, including phones, tablets contained only 3 unique polymers. This heterogeneity of polymers across product categories makes it challenging to segregate and recover plastics at end-of-life (Wagner et al. 2019). This inability would hinder circular systems transition that could help keep the plastics out of water resources.

The most recurring types of polymers observed in our analysis include acrylic, acrylonitrile–butadiene–styrene (ABS), high impact polystyrene (HIPS), blends of polycarbonate and ABS (PC/ABS), polycarbonate (PC). ABS and HIPS streams are largely found in casing components of products including TVs, CRT Displays and printers (see Table 4.2), whereas PC and PC/ABS polymer blend are mainly found in components such as casings, keyboards, bezels of laptops, desktops, phones, and tablets. Acrylic is used to make backlight to illuminate liquid crystal displays and is found in TVs, monitors, laptops, and tablets. Of all the samples characterized, a small portion (less than 5% of total samples) of plastic types are categorized as "others". While these polymers are present in relatively low concentrations, a wide variability in the plastic types is observed (see Table 4.3). The breakdown of these polymer types includes polymer blends (e.g., ABS/PMMA, ABS/PET, PC/PTFE, PS/PPE, PS/PPO), halogenated polymers (e.g., PVC), thermosets (e.g., PUR), valuable plastics (e.g., PP, PE, PET, PA), and other engineered plastics (e.g., POM, PBT, PPO, PPE).

Table 4.1. Average percentage mass of plastics per product category

Product category	**Average Mass of a product (g)**	**Average mass of plastic in a product (g)**	**% Contribution of plastic to total mass of product**
TV	28,625	8,407	29%
Monitor	4,482	1,329	30%
CRT Display	20,491	2,975	15%
Desktop	8,692	1,876	22%
Printer	6,138	3,741	61%
Laptop	2,706	772	29%
Phones	127	35	27%
Media consoles	3,140	845	27%
Tablets	433	105	24%

Note: Data has been taken from BOM data from Chapter 2 (also Babbitt et al.(2020)).

Table 4.2 Relative contribution of polymer types observed in characterized samples for each product category. Values are presented in percent by mass

Polymers	TV	Monitor	CRT Display	Desktop	Printer	Laptop	Phone	Media console	Tablet	Shredder residue
Acrylic	16%	34%				18%			19%	
ABS	26%	41%	44%	56%	51%	6.1%		52%		90%
PC/ABS	10%	4.2%		25%	0.45%	60%	17%			1.1%
PC	11%	4.3%	4.8%	7.9%	2.8%	12%	74%	12%	72%	5.3%
HIPS	33%	2.6%	48%	0.22%	27%			19%		0.65%
Unknown	1.9%	0.41%	3.1%		9.3%	2.7%	10%	9.0%	5.8%	
Others	2.1%	13%	0.7%	11%	9.0%	1.4%		7.2%	3.0%	2.8%

Note: "Unknown" indicates the polymers were not identified. Media consoles include DVD players, gaming consoles, and blu ray disc players. E-readers are included in Tablet category

Many of the polymer blends observed in this analysis are used to either improve aesthetic feel, increase thermal or mechanical resistance, or meet recycled content targets in a product. One such example is blend of ABS/PMMA, which is used to provide glossiness, scratch resistance and good color depth to a component (Szabo et al. 2011). In our analysis, ABS/PMMA blend is mainly found in polymer samples analyzed from external casing components, which requires to have good aesthetic properties as they play a role in consumer decision making. Another blend, ABS/PET could be attributed to manufacturers initiatives to facilitate use of recycled plastics in a product. In the recent years, brands have been taking initiatives (e.g., NextWave Plastics) to keep plastics out of oceans. As a result, brands have started to recycle ocean bound plastics, which is mainly PET, and integrate into products (Thakkar 2021; Lear 2019). For example, recently, as a part of these initiatives, HP launched a computer notebook that contains 50% recycled plastic in bezel and speaker enclosures (Peters 2019; Thakkar 2021). While these initiatives are promoting the use of recycled plastics in the electronics sector , new blends are being introduced that cannot be easily recycled with existing technologies (Maris et al. 2018). As a result, these plastics could either end up in landfills (Mortula et al. 2021) or incineration facilities, releasing microplastics into the environment that could potentially migrate to water resources (Chai et al. 2020; Labunska et al. 2013; Zhang et al. 2021).

Table 4.3. Relative breakdown of polymers categorized as "Other" in the total samples characterized in this study

Polymers	Percent by mass
Blend of ABS and polymethyl methacrylate (ABS/PMMA)	40%
Blend of ABS and polytetrafluoroethylene (ABS/PTFE)	2.1%
Blend of ABS and polyethylene terephthalate (ABS/PET)	11%
Blend of PC and polytetrafluoroethylene (PC/PTFE)	3.5%
Blend of polystyrene and polyphenylene ether (PS/PPE)	7.7%
Blend of polystyrene and polyphenylene oxide (PS/PPO)	0.55%
Polyoxymethylene (POM)	3.4%
Polyphenylene oxide (PPO)	12%
Polypropylene (PP)	6.9%
Polyethylene terephthalate (PET)	1.6%
Polyphenylene ether (PPE)	1.3%
Polyvinyl chloride (PVC)	1.5%
Polyamide (PA)	0.35%
Polyethylene (PE)	0.18%
Polyurethane (PUR)	2.4%
Polybutylene terephthalate (PBT)	4.9%
Poly tere-phthalimines (PPTP)	0.045%
Polyphthalamide (PPA)	0.33%

3.2 Flame retardants and other contaminants

Results also suggest the presence of brominated flame retardants in the polymer samples from e-waste, as determined by presence of FR (17) code on polymers and XRF analysis of polymer samples. The concentrations of bromine (Br) and antimony (Sb) in XRF analysis are used as a proxy to identify brominated flame retardants (Aldrian, Ledersteger, and Pomberger 2015; Wagner et al. 2020). The concentration ranges of bromine and antimony show a bimodal distribution, with many parts with close to zero or low concentrations and a small number of values with high concentrations (see Figure 4.2). These extreme ranges are an indication that these elements are not only observed as intentionally added additives as BFRs, but also as carryover contamination from the past management of products (Jandric et al. 2020; Turner and Filella 2017; Li et al. 2020). XRF analysis also show the presence of chlorine (Cl) element, which typically indicate the presence of a halogenated polymer, poly vinyl chloride in the e-waste literature (Shaw and Turner 2019; Turner 2018) (Figure 4.2).

In the polymer samples that are analyzed by XRF, the highest concentrations of Br, Sb, and Cl are found in ABS and HIPS polymers used in CRTs and display samples from the e-waste recycler, and in some of the older models of products disassembled in the lab. These CRT products and polymers are no longer widely used, which is why there are fewer values of high concentrations, suggesting that regulations on BFRs use might be working. This decrease in BFR also suggest that the electronics industry is shifting to using more of phosphorous and mineral FRs in newer electronic products. This shift is evident in our analysis, as we observed presence of halogen free organic phosphorus compounds (ISO 2021), determined by FR(40) code in marking of the plastic samples. These FRs are mainly observed in PC/ABS blend and PC polymer samples from enclosures of smaller electronics including laptops, phones, and tablets.

E-plastics are also observed to contain other heavy metals including chromium, mercury, lead in low levels indicating a possibility of carry over contamination from previous product life (Mao et al. 2020b; 2020a) (Figure 4.2). Additionally, titanium is identified as well, which indicates the presence of titanium oxide, a pigment added to the plastics to achieve whiteness, brightness, and opacity (Alassali et al. 2019). The higher levels of titanium are mainly observed in the white ABS plastics samples, and as the shade of the color moves towards black, the concentration levels are relatively low.

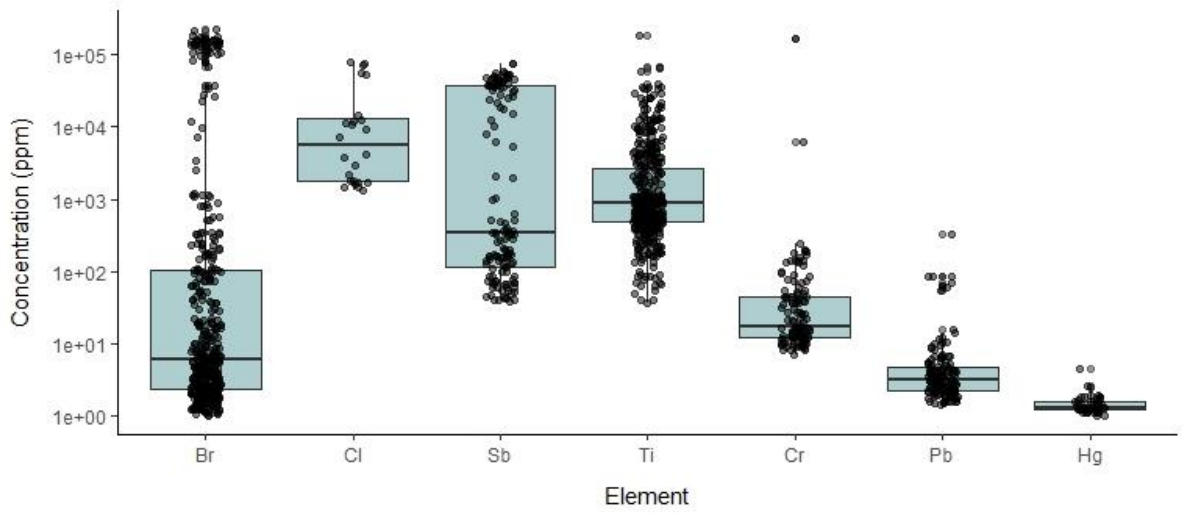

Figure 4.2 Concentration ranges of elements of interest identified in XRF analysis.
Note: concentrations values on y-axis are expressed in log value. Br – Bromine, Cl- Chlorine, Sb-Antimony, Ti – Titanium, Cr- Chromium, Pb – lead, and Hg – mercury

3.3 Baseline national level e-waste flows

We estimated baseline national level e-waste flows using a published MFA model (Althaf, Babbitt, and Chen, 2021) that accounts for product consumption patterns, lifespan, and average polymer composition data. These flows are evaluated to understand the implications of establishing e-plastics circular systems to keep plastics out of the environment. Results indicate that the e-waste stream is mainly dominated by ABS and HIPS polymers, followed by PC and PC/ABS, which have demand in the recycling market because they can be recovered using existing infrastructure (Deubzer 2021; Riise 2020; MGG Polymers 2019) (Figure 4.3). However, a significant portion of total ABS (28%) and HIPS (72%) streams in e-waste are also observed to contain BFRs, which can hamper recovery and recycling of polymers (Ragaert, Delva, and Van Geem 2017; Wagner and Schlummer 2020) (Figure 4.3). Most ABS and HIPS polymer streams with BFRs are observed to originate from CRT and flat panel displays. While this study did not identify specific type of FRs, past studies showed that CRT displays contain "legacy" BFRs (Hennebert and Filella 2018; Wagner and Schlummer 2020), and as a result, cannot be recycled back into new products. Further, the difficulty in removal of these legacy BFRs makes the fraction of polymers containing BFRs less economically viable for recycling (Wagner and Schlummer 2020).

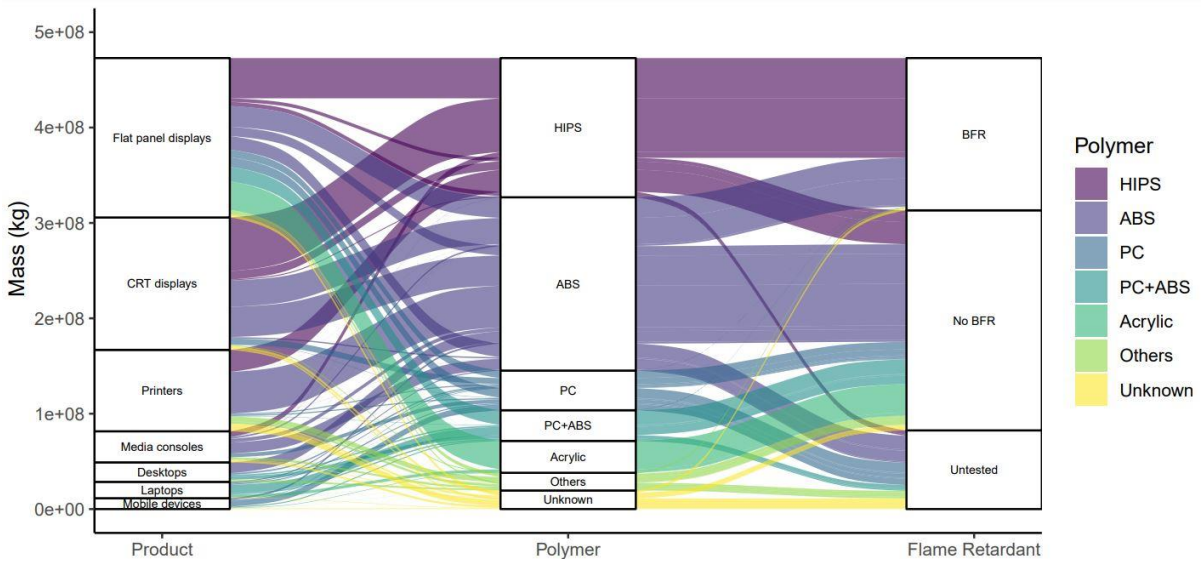

Figure 4.3 Baseline national level e-waste flows in kilograms for polymer types observed from each product category and then disaggregated by the presence or no presence of brominated flame retardants.

Note: Flat panel displays include TVs and monitors, Media consoles include DVD players, gaming consoles and Blu ray disc players. Mobile devices include phones, tablets, e-readers, MP3 players, cameras, and digital camcorders

However, given that CRT displays are no longer used by consumers and expected to phase out of the U.S. e-waste stream by 2030 (Althaf, Babbitt, and Chen 2019), the amount of legacy polymer streams are expected to decline in the future. Additionally, EU ban on all BFRs in electronic displays might eventually phase out of BFR streams from flat panel displays as well. As BFR content is declining, increase in other FRs content in the waste stream, especially organophosphorus FRs that were used a replacement for legacy BFRs is observed in our analysis. While these organophosphorus FRs are not currently restricted, these materials may also impact the environment and human health (Blum et al. 2019). Further, organophosphorus FRs may limit the recyclability of e-plastics because they undergo chemical degradation during processing (Haarman, Magalini, and Courtois 2020). These challenges suggest that a tradeoff analysis should be carried out before replacing one chemical with another to avoid regrettable substitution (Blum et al. 2019).

It is challenging to benchmark the results of this study to other analyses, as most of the literature focuses on case studies in Europe, where different products are covered under electronic waste regulations, and includes data collected in a different time period. Comparison of the plastics

flows with past study in 2000s (Fisher et al. 2000) specific to the U.S. shows that the contributions of ABS, PC and PC/ABS have increased, however, HIPS has decreased compared to 2000s. The increase in PC/ABS could be attributed to the electronic industry's shift toward halogen free plastics to replace restricted BFRs in ABS and HIPS (Riise 2020). The polymer flows observed in this study are partially consistent with the recent U.S. study (Anderson, Yu, and Chen 2022) that evaluated a fraction of e-waste (electronic shredder residue), which also found that ABS and PS are observed in higher proportions. Additionally, our study found that e-waste also contains other recyclable polymer streams, such as PC and PC/ABS, as scope of our analysis included a diverse range of products along with shredder residue.

3.4 Uncertainty analysis

To further interpret how the e-waste polymer composition would change with variability in product designs, an example of display product with casing components made with different plastic types was investigated (see Table B10 and Table B11). In our baseline results, display products represented a significant fraction of plastics in the e-waste. Therefore, if a different plastic type compared to our baseline was assumed as a casing material in TV, findings showed that the net value of dominant polymer flows (i.e., ABS, HIPS and PC/ABS) changed (see Figure 4). For instance, in the scenario of TV casing made of ABS, ABS stream increased by 23% and HIPS content decreased by 34% in the waste stream. Whereas for HIPS as a casing material, HIPS content increased by 31%, and ABS and PC/ABS content decreased by 12% and 46%. Finally, when HIPS and PC/ABS are used as casing materials, PC/ABS and HIPS content in the waste increased by 81% and 23%, while ABS decreased by 21%. As the use of PC/ABS is becoming more prominent in the electronics sector, it is expected that organophosphorus FRs content would increase as well. Further, the BFR content in the e-waste stream will decrease with the decline of ABS and HIPS polymers, which contain these FRs the most. This analysis suggests that estimated e-plastics waste composition is sensitive to the plastic types contained in the products entering end of life. The model developed in this study can be adapted to include newer products as e-waste is continually evolving due to rapid innovation cycles and changing product consumption patterns (Althaf, Babbitt, and Chen 2019)

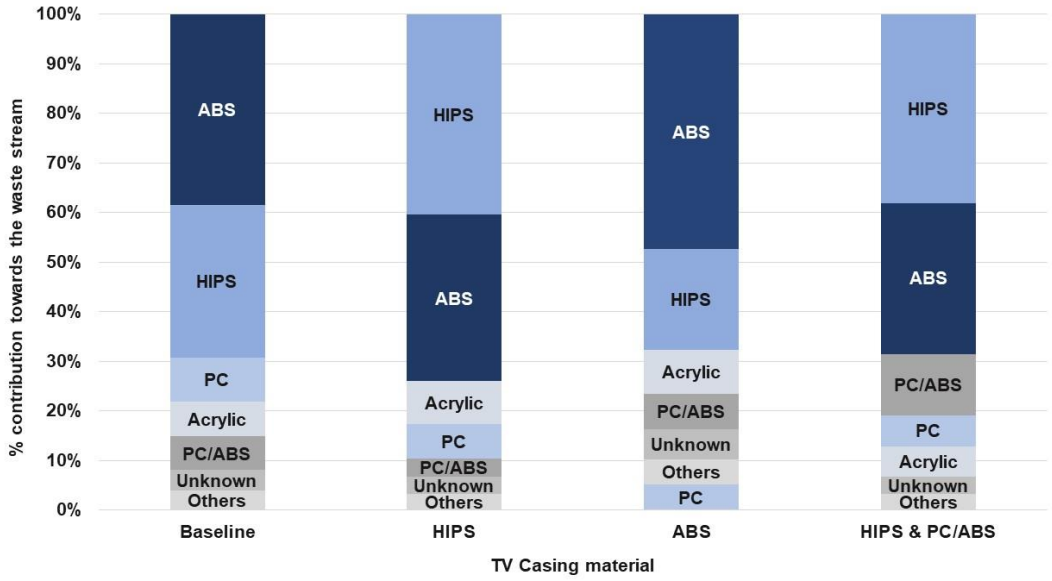

Figure 4.4 Polymers flows found in the e-waste stream for baseline scenario and scenarios with different casing material used in TVs.

4. Implications

National level material flow analysis shows that total amount of plastics from consumer electronics currently entering the e-waste stream is approximately 472.8 thousand metric tons, which is about 3% of the total plastic waste estimated compared to other major sectors (e.g., packaging) in the U.S. (approximately 14,000 thousand metric tons) (Di et al. 2021). However, the presence of chemical additives and contaminants identified in this study may cause e-plastics to have a disproportionately high impact on water pollution and ecological health. Though this research did not measure the effect of e-plastics on water resources, potential impacts can be inferred using existing literature. For example, incineration of polymers containing BFRs has been shown to result in FR emissions and formation of toxic dioxins and furans (Tue et al. 2016; Zhang, Buekens, and Li 2016). Organophosphate flame retardants (FR (40)s) observed in this study can also pollute the environment when incinerated (Li et al. 2019). These emissions may partition into sediments (Li et al. 2019; Zhou et al. 2017) or soil (Cristale et al. 2019; Matsukami et al. 2015) around e-waste facilities before migrating to nearby freshwater resources. Further, BFRs, antimony, lead, and mercury were observed to leach out of landfills (Choi, Lee, and

Osako 2009; Morin et al. 2017; Singh, Duan, and Tang 2020; Zhan et al. 2020), from where they can potentially enter soil and migrate to ground water resources.

Minimizing these risks will require systemic changes to electronic waste and plastics management systems. However, such changes are hampered by product design, recycling infrastructure, and a lack of effective policies on e-waste in the U.S. To overcome these barriers, a combined effort from all stakeholders in the electronics industry is required to drive changes in product design, policy, and recycling infrastructure. Each of these aspects are further explained below in detail.

4.1 Design

In our analysis, consumer electronics are observed to contain a heterogeneous mix of polymers. This diverse mix of polymers increases the complexity and cost of recovering plastic at end of life. One way to address this is by using fewer plastic types that are common and recyclable (e.g., ABS, HIPS, PC/ABS) (Berwald et al. 2021; Raudaskoski et al. 2019), especially in housing components of flat panel displays that are observed to have the greatest contribution to mass of e-waste.

Further, newer products should focus on design that can facilitate use of recycled plastics, which could help closing the loop and reducing environmental impacts. By increasing demand for recycled content, it could then encourage creation of a market for recycling (Raudaskoski et al. 2019). However, as seen in our analysis, polymer blends (e.g., ABS/PET) were used to integrate recycled content in consumer electronics using ocean bound plastics. While these blends are helping in promoting use of recycled plastics, they are also diversifying the e-waste stream increasing the complexity of plastic recovery using existing recycling technologies (e.g., density separation) (Maris et al. 2018; Dorigato 2021; Vazquez and Barbosa 2016). Therefore, when introducing a new blend or polymer to incorporate recycled content, a collaboration of manufacturers and recyclers is essential for careful analysis on potential impact to recycling infrastructure. This kind of collaboration has shown to strengthen the plastics circulation, which is evident by Dell products (e.g., Dell OptiPlex 3030 All-in-One) that have closed loop plastics (MBA Polymers 2014).

In our analysis, plastics in the e-waste also observed to contain additives, such as flame retardants, pigments, and traces of heavy metals. Presence of these additives reduces the purity of polymer streams from electronics sector (Ragaert, Delva, and Van Geem 2017; Wagner and Schlummer 2020). Brominated flame retardants, especially from CRT displays are of a main concern, however, as these products are expected to phase out of e-waste stream sooner, the legacy BFR content will decline in the future. However, there will still be other BFRs, which were observed in flat panel displays in our analysis, and restrictions have been placed on them only recently. As a result, it will take time to phase out these BFRs from the e-waste stream. Additionally, other FR types, especially organophosphorus FRs that are used to replace legacy BFRs are observed to increase in the waste stream that may limit recycling (Haarman, Magalini, and Courtois 2020), and as a result, need to be substituted or phased out of products. Future research should focus on implementing product designs that would not require use of FRs, especially in casing components that could be easily targeted for recycling (Raudaskoski et al. 2019).

Another potential opportunity to facilitate higher material recovery is material identification (Berwald et al. 2021; Raudaskoski et al. 2019). In our analysis, we observed that less than 50% of total samples have markings suggesting that more half of the polymer could be lost if identified inaccurately. While relatively newer product models in our analysis were observed to have markings, often these are hard to read and inconsistent hampering polymer identification. For example, many samples in our analysis had markings that contained multiple polymers with a small arrow that is not easily visible pointing towards the actual polymer type used, creating difficulty in identifying exact plastic type. Added to this, markings are observed to be printed in the same color as the polymer part causing an obstacle in locating the marking on the plastic part at the first place. These poor labeling practices call for manufacturers to print an easy to read, visible and accurate marking for the newer products following the industry standard (e.g., ISO 11469) (ISO, 2016). This ISO standard requires for the marking to include both the polymer type and presence of any additives (including FRs) which could drastically improve segregation and recovery of polymers from the waste (Wei and Liu 2012; Mohammed et al. 2022).

Finally, integrating design for disassembly principles in the product design implementation phase could be beneficial as well (Berwald et al. 2021). From the product disassembly activity in

Chapter 2. it is observed that the current design of electronics makes it challenging to recover polymers. This is mainly because as consumer electronics have become compact over the years, as a result, there has been increasing use of adhesives to glue components, and use of fasteners that are small and hidden to accommodate this design (Babbitt et al. 2021; Olson and Riess 2012). Therefore, in newer products, especially for housing components in larger products (e.g., flat panel displays, desktops), using standardized joints or quick snap fasteners would facilitate easy disassembly (Giudice and Kassem 2009) for quick recovery of materials.

4.2 Recycling infrastructure

In the United States, many activities are considered as recycling. Some e-waste facilities first might separate equipment for refurbishment or reuse purposes, and then may harvest valuable components, such as PCB (Ryen et al. 2018; GEC 2009; Schumacher 2016). After these steps, some recyclers might separate plastics from housing components of larger products (e.g., flat panel displays, CRTs, desktops), crush and bale these plastics before shipping them to a plastic recycler (GEC 2009). In many cases, facilities send low-grade electronics to an automated shredder processing rather than disassembling (Ryen et al. 2018; GEC 2009). As e-waste recycling is mainly driven by the recovery of precious metals (Cucchiella et al. 2015), most of the plastics might end up in this shredder residue that is sent to secondary metal recycling facilities. While plastics in this fraction might be used for energy recovery, embedding a new plastics recovery step into metal processing systems could help recover lost plastic fraction. Additionally, e-waste processing facilities can partner with facilities that have ability to separate plastics and metals fraction from shredder residue before sending residue to metal processing. One such example is Owl Recycling facility in the U.S., which recovers plastics fraction from shredder residue from electronics, then sort recovered plastics, and process them to produce recycled plastics, including ABS, PS, PC (Staub 2021).

In addition to recovering plastics fraction, there is a need to integrating broader range of spectroscopic techniques along with manual sorting to improve efficiency of plastics identification. Some of the example technologies include FTIR, Raman, LIBS or automated sorting technologies using NIR (Beigbeder et al. 2013; Wu et al. 2020) along with XRF that identifies halogens (especially BFRs) (Aldrian, Ledersteger, and Pomberger 2015) Using these technologies also ensure the separation of BFR containing plastics from non BFR fraction that

can impact quality of recycled plastics (Wagner et al. 2019; 2020) as well as restrict the carry over contamination to the next product life. Once the BFR fraction is separated, mechanical recycling technologies such as density separation or electrostatic separation can be adapted, which can efficiently separate basic plastics from e-waste (Sahajwalla and Gaikwad 2018).

In parallel to separating BFR fraction effectively, there is also need for technologies that can remove these contaminants for safe management. Research has shown that solvent-based technologies (e.g., the CreaSolv process) can chemically remove BFRs increasing the yield of ABS and HIPS as well as ensure environment safe management of BFR plastics (Buekens and Yang 2014; Ma et al. 2016; Schlummer et al. 2016; 2007; Wagner and Schlummer 2020). Additionally, thermo-chemical recycling methods, such as pyrolysis (Achilias and Antonakou 2015; Charitopoulou et al. 2021; Evangelopoulos et al. 2019) is also observed as a promising alternative to treat BFR plastics fraction.

4.3 Policy

Policy can be an overarching tool driving both green product design changes and recycling infrastructure in the United States. While our analysis shows that e-waste contains recyclable plastics, there is a need to increase the quantities of these plastics from e-waste, which can be achieved by increasing e-waste collection rates. Currently, in the United States, 25 states have legislation to promote e-waste recycling, however, each of these state programs vary widely in terms of product scope and requirements (Schumacher and Agbemabiese 2021). As a result, research has shown that this variability in requirements is creating confusion for the recyclers on how to handle e-waste (Schumacher and Agbemabiese 2019; Hickle 2014).Therefore, it is recommended that the United States move towards a unified e-waste legislation that include requirements, which were shown to increase e-waste collection rates effectively (Schumacher and Agbemabiese 2019). These requirements include convenience standards, entities financially responsible for waste management, expanding end user groups and setting annual collection targets (Schumacher and Agbemabiese 2019). Further educating consumers can also improve collection and material recovery systems (Gaustad et al. 2018). Policies should facilitate programs to create awareness in consumers about e-waste recycling and increased convenience of e-waste collection and management options (Schumacher 2016).

Further, the biggest opportunity to speed up circular transition and drive market is requiring use of recycled content in newer products (Carr et al. 2019). One way to achieve this by developing environmentally preferable procurement policies, which incentivizes purchase of products with environmental benefits, such as recycled content (Plastics Industry Association 2017). These policies generally leverage green product standards or ecolabels that have requirements to have recycled content (Plastics Industry Association 2017). For example, US federal procurement policy requires purchase of products on the Electronic Product Environmental Assessment Tool (EPEAT) registry (US EPA 2014; Plastics Industry Association 2017). EPEAT is an ecolabel that requires products to use post-consumer recycled plastics in products like desktop computers, printers, and televisions. Additionally, EPEAT also have requirements targeting multiple aspects of green design including use of single type of plastic that is recyclable and have markings for identification as per industry standard, design for disassembly (GEC 2020).

In addition to increasing market demand and collection rates, there is also a need for funding opportunities to expand and upgrade current e-waste facilities enabling plastics recovery. There are many ways to provide funding, such as tax credits, grants or subsidies at federal, state, or local level (US EPA 2021). For instance, Biden administration passed an infrastructure bill that included grants to improve municipal recycling programs and also to upgrade local waste management systems, which are targeted towards reducing plastic waste (Quinn 2021).

5. Conclusion

In summary, this is the first study to create an updated profile on the polymers contained in consumer electronic waste in the United States. Results indicate that e-waste contains a wide variety of polymers, flame retardants and other additives that may pollute water resources if not properly managed. Further, national level e-waste flows are evaluated to understand implications to establishing e-plastics circular systems that can help reduce these impacts. E-waste flows predominantly contain ABS, HIPS, PC, and PC/ABS polymers that are recyclable using existing recycling infrastructure. However, the presence of brominated flame retardants can impede the recovery of these plastics. For the United States to transition to circular systems and manage plastic pollution, a joint effort from stakeholders in the electronics industry is required to drive changes in product design, e-waste policies, and recycling infrastructure.

Chapter 5 Conclusion, Limitation, and Future Research

Materials provide the functionalities and features of modern electronic products on which our society depends. However, consumption of these materials comes at a cost to the environment. Most literature on consumer electronics has focused on supply chain energy, carbon footprint, or end of life management impacts, however, water impacts have not been fully studied yet. To be able to conserve water resources and improve overall sustainability of consumer electronics, understanding how materials used in consumer electronics contribute to water impacts is essential. The ability to evaluate water impacts hinges on availability of information on material content per product, life cycle inventory data on water consumption and emissions, and impact assessment models that connect LCI data to water impacts. Data on these aspects are available at varied levels for different materials used in the electronics. Therefore, this research aimed to close this knowledge gap by contributing novel data on material profile, evaluating freshwater scarcity and pollution impacts for metals, and establishing linkages to water impacts for plastics found in consumer electronics.

This research first created bill of materials for 95 consumer electronic products establishing data on quantity of various bulk metals and plastics found in the electronics. By connecting BOM data to existing LCI and LCIA models, supply chain water impacts of metals found in consumer electronics are evaluated to identify hotspots to evaluate opportunities for future improvements. Here, impacts are evaluated on individual metal level and then for a representative profile of metals for smartphone and laptop. Further, opportunities to reduce these supply chain water impacts of metals are also identified through product design changes and sourcing from alternated supply chains. Finally, national level e-waste flows of polymers used in common consumer electronics are quantified to establish linkage between these materials and their potential impact on water resources. Furthermore, implications to establishing circular systems in the U.S. are studied to keep plastics entering the environment. Major findings of this research are summarized below:

Chapter 5

Key takeaways

- Transparent and comprehensive bill of material data are created for 95 unique consumer electronics. These BOM data identified metals and plastics as the major contributors towards the total mass of products evaluated, suggesting that these materials could have major impact on water resources.
- For metals supply chains, freshwater scarcity impact was greatly influenced by water consumption for mining and production activities and indirect water consumption for fuel extraction and energy production. Additionally, geographic location of metals production is also observed as a contributing factor as water stress varies spatially.
- For water quality impacts, disposal of sulfidic mine tailings contributed the greatest towards the water quality.
- Precious metals have the highest supply chain water impacts. At product level, water impacts are influenced by material intensity as well as individual material level impacts. Aluminum and gold are identified as hotspots for both smartphone and laptop.
- Sourcing metals from alternate supply chains and increasing use of recycled content in a product showed a greater potential for reducing water impacts of these materials.
- Consumer electronics are observed contain a diverse mix of polymers (up to 23 unique types), flame retardants, including brominated and phosphorous FRs, and other contaminants, such as heavy metals and pigments. If e-waste is mismanaged, these materials are observed to potential to leach into the environment, from where they can migrate to freshwater resources.
- Trends in national level e-waste flows show that significant portion of plastics fraction contains recyclable polymers, including ABS, HIPS, PC, PC/ABS polymers. However, major portion of these polymer streams contain BFRs that can hinder circular systems planning. There is a critical need for investment in recycling technology and infrastructure, designing effective e-waste policies, and changes in product design for efficient plastics recovery.

Recommendations

After evaluating the findings of this study, three common solutions are recommended that are identified as important players in minimizing water impacts of materials and building resilience towards the future demand of both water and materials. First recommendation is to integrate green product design strategies in new products, such as design for disassembly, material identification, use of recycled materials, and substitution or reduction of high impact materials. However, solutions including use of recycled materials and material substitution may also introduce sustainability tradeoffs that should be proactively analyzed and mitigated. Further, a collaboration is needed across all levels of the supply chain to better connect design and material choices to product usage, lifespan, and end-of-life management.

The second recommendation is investing in improvements to recycling infrastructure to be resilient to an evolving product ecosystem in the e-waste stream. Currently, the major fraction of materials that are recovered from electronics are precious metals and copper, while plastics are often lost to energy recovery. Closing the loop on the plastics can not only conserve resources but also reduce the impact of these materials on water resources. Transition to closed loop systems will require creating awareness among secondary metal processing facilities about the value of plastics and drive changes in the processes for plastics recovery. Further, investment in sorting and identification technologies, such as spectroscopic techniques (e.g., FTIR, XRF) can help in greater recovery of polymers with high purity. Further, solvent- based technologies can help in treat BFR polymer fractions to increase the volume of recyclable plastics as well as for safe disposal.

Finally, policy interventions are needed to drive both design and recycling improvements. Minimizing supply chain water impacts calls for multi-stakeholder engagement to invest in sustainable water management solutions. These solutions can be facilitated by the policy development on driving engagement with suppliers to raise awareness about the water risks, support water use efficiency, proper management of mine tailings, and increase the use of low-carbon and low-energy sources and conversion technologies (Hoekstra 2014; Morgan 2018; Stuckenberg and Contento 2018). Additionally, leveraging spatial variability in material availability, policies can be developed that can reduce water impacts of supply chain holistically,

for example, through supplier selection based on water intensity, water stress of the local region, and potential tradeoffs to supply chain disruptions. Further, developing a unified e-waste legislation in the United States can help drive collection and recycling infrastructure for material recovery. Procurement policies mandating using of products meeting green product standards or ecolabels that mandate use of recycled content would accelerate the transition to circular economy as well as reduce the impact of materials on water resources. For consumers, policies should facilitate greater education about e-waste recycling and increased convenience of e-waste collection and management options. Further, policies requiring products to have water footprinting labeling could help inform consumers about the environmental impact of the product to drive behavior towards use of products with lower water impacts (Steiner, Peschel, and Grebitus 2017).

Limitations and extensions

This research utilized quality data that were available to evaluate water impacts of materials found in electronics. However, there is always an opportunity for developing LCI data on water withdrawals and discharges at spatial level, especially for supply chain of plastics production. For instance, when evaluating supply chain water impacts of metals, an initial analysis was conducted using the generic average data available on plastics to understand how they compare to metals. This initial analysis showed that metals have higher supply chain impacts compared to plastics on individual material level. However, given that water impacts vary with geographic region, better data are required for accurate comparison. Further, studies can extend the model developed in this research to product manufacturing and use, as these phases also require water and energy inputs and create emissions that degrade water quality. Taking advantage of data created on e-plastics profile in this study, another potential extension is to studying fate, exposure, and effects of plastics and contaminants found in electronics in the environment. This could help in building life cycle inventory and impact assessment models for plastics, which can then be used to evaluate impact of plastics in the waste stream on water more completely. Finally, improving the overall sustainability of consumer electronics water footprint should be considered in addition to carbon footprint, energy demand to avoid any burden shifting.

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Appendix A

1. Life cycle inventory

1.1. Case study products

This study evaluated two commonly used electronic products (smartphone and laptop) to understand the material impacts at product level. This section outlines the data used to calculated metal content in an average smartphone and laptop. Multiple literature sources have been utilized to accomplish this task and are indicated below in respective tables.

1.1.1. Product Bill of Material (BOM) profiles

Table A1. Estimate average BOM profile of smartphone and laptop (mass percent)

Product name	Al	Cu	Fe	Plastic	Li-ion battery	PCB	Flat panel glass	Other metals	Others	Total mass (g)
Smartphone	9.90%	1.20%	6.30%	23.8%	22.7%	14.0%	17.4%	2.60%	4.10%	135
Laptop	11.5%	15.4%	1.80%	5.80%	28.3%	12.4%	8.20%	14.0%	2.40%	2706.4

Data source: (Babbitt et al. 2020)

1.1.2. Estimated metal content in lithium-ion batteries

Table A2. Average metal content in batteries estimated by the disassembly of consumer electronics battery cells

Prismatic smartphone cell	
Materials	**Mass %**
Aluminum	19.0%
Cobalt	5.30%
Copper	7.00%
Lithium	1.90%
Manganese	5.00%
Nickel	5.30%
Oxygen	8.70%
Steel	1.70%
Graphite	15.0%
Carbon Black	4.80%
LiPF6	5.50%
EC/other	1.40%
Binders	1.90%
Plastics	14.0%
Cardboard	0.100%
PCB	2.90%
Total mass: 23.6 g per one prismatic (pouch) cell	

Cylindrical (18650) laptop cell	
Materials	**Mass %**
Aluminum	5.40%
Cobalt	18.0%
Copper	12.0%
Lithium	2.10%
Nickel	0.40%
Steel	22.0%
Graphite	16.0%
Carbon black	3.40%
LiPF6	4.50%
EC/other	7.80%
Binders	3.40%
Plastics	5.70%
Total mass: 38 g per one 18650 cell	

Data source: Wang et al. 2012 as summarized by Althaf, Babbitt, and Chen (2021)

1.1.3. Average gold content in PCBs

Table A3. Average content of gold in printed circuit boards (PCBs) found in smartphone

Source	Product	Gold content reported (%)
JOGMEC (2008) as reported by Oguchi et al. (2011)	Mobile Phone	0.18
JOGMEC (2008) as reported by Oguchi et al. (2011)	Mobile Phone	0.14
JOGMEC (2008) as reported by Oguchi et al. (2011)	Mobile Phone	0.12
Vats and Singh (2015)	Mobile phone	0.017
Vats and Singh (2015)	Mobile phone	0.012
Vats and Singh (2015)	Mobile phone	0.016
Vats and Singh (2015)	Mobile phone	0.0090
Vats and Singh (2015)	Mobile phone	0.010
Vats and Singh (2015)	Mobile phone	0.010
Vats and Singh (2015)	Mobile phone	0.0030
Vats and Singh (2015)	Mobile phone	0.017
Vats and Singh (2015)	Mobile phone	0.016
Vats and Singh (2015)	Mobile phone	0.010
MoE and METI (2010) as reported by Oguchi et al. (2011)	Mobile Phone with camera	0.16
MoE and METI (2010) as reported by Oguchi et al. (2011)	Mobile Phone with camera	0.16
MoE and METI (2010) as reported by Oguchi et al. (2011)	Mobile Phone with camera	0.17
MoE and METI (2010) as reported by Oguchi et al. (2011)	Mobile Phone – no camera	0.11
MoE and METI (2010) as reported by Oguchi et al. (2011)	Mobile Phone – no camera	0.14
MoE and METI (2010) as reported by Oguchi et al. (2011)	Mobile Phone – no camera	0.13
MoE and METI (2010) as reported by Oguchi et al. (2011)	Mobile Phone – no camera	0.17
MoE and METI (2010) as reported by Oguchi et al. (2011)	Mobile Phone – no camera	0.15
Kim et al. (2011) as reported by Tan et al. (2017)	Mobile Phone	0.0045
Kasper et al. (2011) as reported by Tan et al. (2017)	Mobile Phone	0.080
Li et al. (2012b) as reported by Tan et al. (2017)	Mobile Phone	0.0043
Petter et al. (2014) as reported by Tan et al. (2017)	Mobile Phone	0.088
Xiu et al. (2015) as reported by Tan et al. (2017)	Mobile Phone	0.0065
Tan et al. (2017)	Mobile Phone	0.12
Sahan et al. (2019)	Mobile Phone	0.14
Average content in a mobile phone		**0.078**

Data source: Average content of gold in PCBs of smartphone is estimated by using the data (Oguchi et al. 2011; Vats and Singh 2015) compiled by Althaf, Babbitt, and Chen (2021), Tan et al. (2017) and Sahan et al. (2019)

Table A4. Average content of gold in printed circuit boards (PCBs) found in a laptop

Source	Product	Gold content reported (%)
Hageluken (2006)	Computer	0.0250
Yamane et al. (2011)	Computer	0.130
Yazici et al. (2010)	Computer	0.000600
Bizzo et al. (2014)	Computer	0.0142
Hino et al (2009) as reported by Bizzo et al (2014)	Computer	0.0205
Zhou et al (2007) as reported by Szałatkiewicz (2014)	Computer	0.00760
Guo et al. (2011)	Computers	0.00600
JOGMEC (2008) as reported by Oguchi et al. (2011)	laptop	0.0900
Average content in a laptop		**0.0367**

Data source: Average value of gold in laptop is calculated using the values reported by (Bizzo, Figueiredo, and de Andrade 2014; Guo et al. 2011; Hagelüken 2006; Szalatkiewicz 2014; Yamane et al. 2011; Yazıcı et al. 2010) as compiled in Althaf, Babbitt, and Chen (2021)

1.1.4. Average content of lead in PCBs

Table A5. Average content of lead in PCBs found in a smartphone

Source	Product	Lead content reported (%)
JOGMEC (2008) as reported by Oguchi et al. (2011)	Mobile Phone	2.70
JOGMEC (2008) as reported by Oguchi et al. (2011)	Mobile Phone	1.30
JOGMEC (2008) as reported by Oguchi et al. (2011)	Mobile Phone	0.0900
Yamane et al. (2011)	Mobile Phone	1.87
MoE and METI (2010) as reported by Oguchi et al. (2011)	Mobile Phone with camera	0.540
MoE and METI (2010) as reported by Oguchi et al. (2011)	Mobile Phone with camera	1.60
MoE and METI (2010) as reported by Oguchi et al. (2011)	Mobile Phone with camera	0.600
MoE and METI (2010) as reported by Oguchi et al. (2011)	Mobile Phone – no camera	2.20
MoE and METI (2010) as reported by Oguchi et al. (2011)	Mobile Phone – no camera	2.50
MoE and METI (2010) as reported by Oguchi et al. (2011)	Mobile Phone – no camera	1.50
MoE and METI (2010) as reported by Oguchi et al. (2011)	Mobile Phone – no camera	1.50
MoE and METI (2010) as reported by Oguchi et al. (2011)	Mobile Phone – no camera	1.40
Average content in a mobile phone		1.48

Data source: Average value of lead in mobile phone is calculated from multiple sources reported by (Oguchi et al. (2011) and Yamane et al. (2011)) as compiled by Althaf, Babbitt, and Chen (2021)

Table A6. Average content of lead in PCBs found in a laptop

Source	Product	Lead content reported (%)
Hageluken (2006)	Computer	1.50
Yamane et al. (2011)	Computer	5.53
Yazici et al. (2010)	Computer	3.90
Bizzo et al. (2014)	Computer	2.50
Hino et al. (2009) as reported by Bizzo et al. (2014)	Computer	2.96
Veit et al. (2002) as reported by Bizzo et al. (2014)	Computer	0.350
Veit et al. (2002) as reported by Bizzo et al. (2014)	Computer	0.950
Veit et al. (2002) as reported by Bizzo et al. (2014)	Computer	1.35
ZHOU et al. (2007) as reported by Szałatkiewicz (2014)	Computer	0.63
Guo et al. (2011)	Computer	2.91
JOGMEC, 2008 as reported by Oguchi et al. (2011)	laptop	1.10
Average content in a laptop		**2.15**

Data source: Average value of lead in laptop is calculated from multiple sources (Bizzo, Figueiredo, and de Andrade 2014; Guo et al. 2011; Hagelüken 2006; Oguchi et al. 2011; Szalatkiewicz 2014; Yamane et al. 2011; Yazıcı et al. 2010) as compiled by Althaf, Babbitt, and Chen (2021)

1.1.5. Literature data on In, Ta, Sn, Ga, Pd

Table A7. Average content of indium, tin, tantalum, gallium, palladium, and silver

Materials	Content
Indium in LCD screens (kg/kg)	0.0002
Tantalum in PCB (mean wt %)	0.0172
Tin in PCB (mean wt %)	2.922
Gallium in PCB (mean wt %)	0.0035
Palladium in PCB (mean wt %)	0.117
Silver in PCB (mean wt %)	0.13

Data source: Average values of these metals are from data compiled by (Althaf, Babbitt, and Chen 2021). Average content of indium in LCD glass is calculated as average of values from multiple sources (Hasegawa et al. 2013; Rocchetti et al. 2015; Ruan, Guo, and Qiao 2012; Wang 2009) by Boundy et al.(2017). Average content of materials present in PCBs reported by Wang and Gaustad (2012) based on data from multiple sources.

1.1.6. Literature data on REEs and other elements

Table A8. Average content of rare earth elements (REEs) and other elements in a phone

Element	Phones (mg/kg)
Neodymium	1372
Praseodymium	169
Chromium	2415
Antimony	84.09
Tellurium	14.31
Titanium	1332
Barium	5158
Cadmium	0.55
Zinc	5701

Data source: Buechler et al. (2020)

Table A9. Average content of rare earth elements (REEs) and other elements in PCBs

Element	PCBs (mg/kg)
Lanthanum	13.14
Neodymium	155
Praseodymium	7.69
Chromium	606
Antimony	3652
Tellurium	15.89
Titanium	1365
Manganese	6928
Barium	5798
Cadmium	1.39
Zinc	10233

Data source: Buechler et al. (2020)

The data summarized in above tables were combined to estimate elemental composition of metals identified as being present in the case study smartphone and laptop, as shown in Table S10.

Table A10. Average material composition of a smartphone and laptop

Material	Smartphone (g)	Laptop (g)
Al	19.16	437.2
Cu	3.755	94.18
Fe	9.0262	394.6
Ni	1.627	1.516
Zn	0.7696	3.434
Ti	0.1798	0.4580
Au	0.01480	0.1233
Ag	0.02457	0.4363
Pt	NA	NA
Pd	0.02211	0.3927
Sb	0.01135	1.2256
Ba	0.6963	1.946
Co	1.6271	68.20
Ga	0.0006600	0.01175
In	0.00005000	0.0004400
Li	0.5833	7.957
Mn	1.535	2.3250
Ta	0.003250	0.0577
Te	0.001930	0.005330
Sn	0.5523	9.806
Nd	0.1852	0.05202
Pr	0.02282	0.002580
La	NA	0.004410
Pb	0.2804	7.225
Cr	0.3260	0.2034
Cd	0.00007	0.0004700

Note: “NA” represents data were not available

1.2. Global material usage for consumer electronics sector

The global demand for the metals for the consumer electronics sector is collected from various industry reports and literature sources.

Table A11. Global material consumption of metals in the electronics sector in metric tons

Materials	Annual mine production (metric tons)	Electronics sector consumption (% of total production)	Electronics sector consumption (metric tons)
Al	64,000,000	9%	5,760,000
Cu	20,000,000	4%	800,000
Mg	1,100,000	6%	66,000
Fe	1,900,000,000	5%	95,000,000
Ni	2,700,000	5%	135,000
Zn	13,000,000	17%	2,210,000
Ti	210,000	12%	25,200
Au	3,300	8%	264
Ag	27,000	23%	6,210
Pt	180	3%	5
Pd	210	17%	36
Rh	180	1%	2
Sb	160,000	26%	41,600
Ba	9,500,000	16%	1,520,000
Co	140,000	80%	112,000
Ga	320	98%	314
In	760	84%	638
Li	77,000	65%	50,050
Mn	19,000,000	2%	380,000
Ta	1,800	48%	864
Te	470	8%	38
Sn	310,000	54%	167,400
La	54,092	16%	8,655
Pr	9,909	75%	7,432
Nd	33,665	94%	31,645
Pb	4,500,000	3%	135,000
Hg	4,000	15%	600
Cr	44,000,000	5%	2,200,000
Cd	25,000	17%	4,250

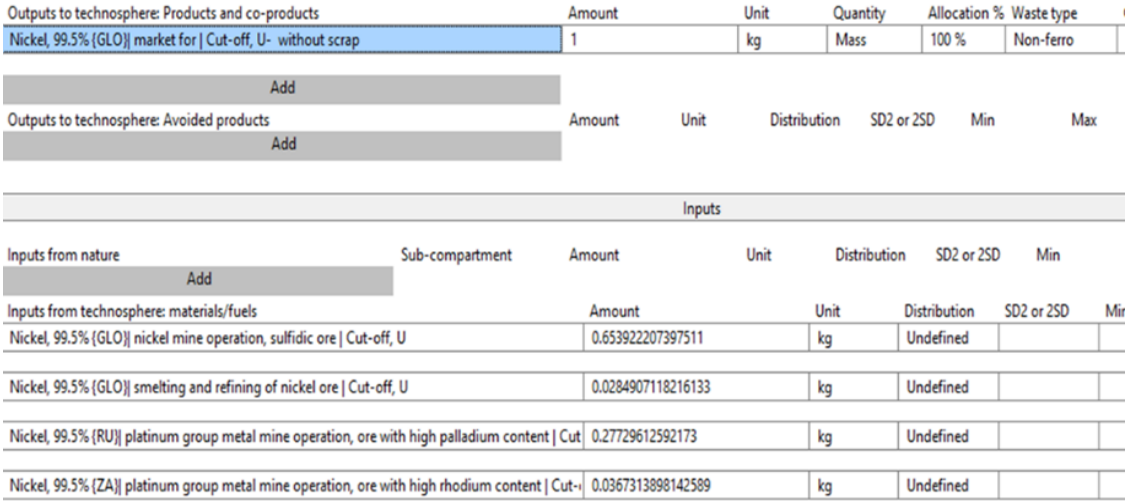
Data source: The main data source used for annual production for most of the materials is USGS mineral commodity summaries 2020 (U.S. Geological Survey 2020) as compiled by Althaf and Babbitt (2021). Annual production data for individual elements were taken from British Geological Survey (2011) as reported by (Althaf and Babbitt 2021). Electronics sector consumption data were taken from USGS and Graedel et al. (2015) as reported by Althaf and Babbitt (2021).

1.3. Ecoinvent processes

Table A12. Ecoinvent processes used to model per kg of material production

Metals	Ecoinvent process block
Steel	Steel, unalloyed {GLO}| market for | Cut-off, U
Zinc	Zinc {GLO}| market for | Cut-off, U
Magnesium	Magnesium {GLO}| market for | Cut-off, U
Copper	Copper {GLO}| market for | Cut-off, U
Aluminum	Aluminium, primary, ingot {|A| Area, EU27 & EFTA}| market for |Cut-off, U
Nickel	Nickel, 99.5% {GLO}| market for | Cut-off, U
Titanium	Titanium, primary {GLO}| market for | Cut-off, U
Gold	Gold {GLO}| market for | Cut-off, U
Palladium	Palladium {GLO}| market for | Cut-off, U
Platinum	Platinum {GLO}| market for | Cut-off, U
Rhodium	Rhodium {GLO}| market for | Cut-off, U
Silver	Silver {GLO}| market for | Cut-off, U
Graphite	Graphite, battery grade {GLO}| market for | Cut-off, U
Manganese	Manganese {GLO}| market for | Cut-off, U
Tellurium	Tellurium, semiconductor-grade {GLO}| market for | Cut-off, U
Barium	Barite {GLO}| market for | Cut-off, U
Cobalt	Cobalt {GLO}| market for | Cut-off, U
Antimony	Antimony {GLO}| market for | Cut-off, U
Tin	Tin {GLO}| market for | Cut-off, U
Lithium	Lithium {GLO}| market for | Cut-off, U
Indium	Indium {GLO}| market for | Cut-off, U
Gallium	Gallium, semiconductor-grade {GLO}| market for | Cut-off, U
Tantalum	Tantalum, powder, capacitor-grade {GLO}| market for | Cut-off, U
Cadmium	Cadmium, semiconductor-grade {GLO}| market for | Cut-off, U
Chromium	Chromium {GLO}| market for | Cut-off, U
Lead	Lead {GLO}| market for | Cut-off, U
Mercury	Mercury {GLO}| market for | Cut-off, U
Lanthanum	Lanthanum oxide {GLO}| market for | Cut-off, U
Praseodymium	Praseodymium oxide {GLO}| market for | Cut-off, U
Neodymium	Neodymium oxide {GLO}| market for | Cut-off, U

The processes blocks are modified to represent the primary production of a metal alone. An example of modified process block is shown below, wherein the equivalent amount of primary production was substituted for any existing inputs of secondary production.



Outputs to technosphere: Products and co-products	Amount	Unit	Quantity	Allocation %	Waste type
Nickel, 99.5% {GLO}| market for | Cut-off, U- without scrap	1	kg	Mass	100 %	Non-ferro

Add

Outputs to technosphere: Avoided products	Amount	Unit	Distribution	SD2 or 2SD	Min	Max

Add

Inputs

Inputs from nature	Sub-compartment	Amount	Unit	Distribution	SD2 or 2SD	Min

Add

Inputs from technosphere: materials/fuels	Amount	Unit	Distribution	SD2 or 2SD	Mir
Nickel, 99.5% {GLO}| nickel mine operation, sulfidic ore | Cut-off, U	0.653922207397511	kg	Undefined		
Nickel, 99.5% {GLO}| smelting and refining of nickel ore | Cut-off, U	0.0284907118216133	kg	Undefined		
Nickel, 99.5% {RU}| platinum group metal mine operation, ore with high palladium content | Cut	0.27729612592173	kg	Undefined		
Nickel, 99.5% {ZA}| platinum group metal mine operation, ore with high rhodium content | Cut-(	0.0367313898142589	kg	Undefined		

Figure A1. Example ecoinvent process block

1.4. Geographic Variability

Table A13 illustrates the variability in the drivers of water scarcity footprint for an example of gold. As illustrated, water scarcity footprint is driven by various factors, including ore mix (coproduct mix), water consumption specific to each coproduct mix and region, AWARE water stress characterization factors of regions where consumption occurs. Leveraging this variability in water consumption and water stress characterization factors, water scarcity footprint of 1 m3 of water consumption is estimated, which is summarized in Table A14. To capture potential geographic variability in these results, the water quality index (WQI) was used as a proxy, because the aggregated nature of the underlying LCI data used here prevented attribution of impacts to specific countries. As summarized in Table S13, WQI represents the amount of wastewater that undergoes at least primary treatment and ranges from 0% to 100% treated (Wendling et al. 2018). While this index does not quantify what the magnitude or nature of such impacts may be, it suggests areas where added water pollution may compound existing water degradation and ecosystem impacts.

Table A13. Geographic variability in drivers of water scarcity footprint including co-product mix, water consumption, AWARE factors for gold

Geography	Coproduct mix	Water consumption (m^3)	AWARE factors (m^3eq/m^3)	Water scarcity footprint (m^3eq)
Canada - Quebec	Gold-silver-zinc-lead-copper	124	7.4	1655
Australia	gold	88	72.11	1865
Canada - Quebec	gold-silver	906	7.4	67781
Canada	gold	568	7.4	33066
Chile	silver-gold	109	80.1	2869
Peru	gold-silver	90	24.93	4200
Papa New Guinea	gold-silver	276	1.43	102485
Sweden	gold-silver-zinc-lead-copper	141	4.41	3699
Tanzania	gold	52	19.45	1936
United States	gold	99	33.84	4678
South Africa	gold	86	36.35	6140

Table A14 represents the water scarcity footprint for 1 m^3 water consumption, and water quality index for the countries (called as "producer countries") in which metals used in consumer electronics are produced. Producer countries for each metal were identified using USGS.

Table A14. Water scarcity footprint for 1 m^3 water consumed and % wastewater treated for the countries in which consumer electronics metals are being mined and produced.

Country	**Abbreviation**	**WSF of 1 m^3 water consumed**	**% Wastewater treated in a country (WQI)**
Argentina	AR	47.1	NA
Australia	AU	72.11	99.44
Baharain	BH	9.93	96.54
Bolivia	BO	6.62	63.66
Brazil	BR	2.17	81.08
Bulgaria	BG	25.63	93.93
Burma	MM	5.02	0
Canada	CA	7.4	96.35
chile	CL	80.1	98.55
China	CN	42.43	80.2
Colombia	CO	0.68	77.21
congo	CG	0.86	0
cuba	CU	5.3	72.52
Ethiopia	ET	28.65	0
Germany	DE	1.36	99.65
Ghana	GH	20.79	53.73
Guatemala	GT	1.2	72.59
Hungary	HU	1.26	93.42
Iceland	IS	0.6	94.47
India	IN	0.97	58.83
Indonesia	ID	23.6	13.6
Iran	IR	66.59	64.32
Israel	IL	82.03	99.49
Japan	JP	0.9	94.06
Kazakhstan	KG	52.58	79.83
kenya	KE	19.5	41.57
Korea, republic of (South Korea)	KR	1.66	0
Kyrgyzstan	KG	68.88	65.61
Laos	LA	5.71	0
madagascar	MG	2.74	0
Malaysia	MY	1.64	82.3
Mali	ML	15.73	NA
Mexico	MX	33.45	91.48
Mozambique	MZ	4.42	45.65
Netherlands	NL	1.17	99.9

Country	Abbreviation	WSF of 1 m^3 water consumed	% Wastewater treated in a country (WQI)
New caledonia	NC	6.9	NA
Nigeria	NG	8.91	30.76
North korea	KP	2.5	NA
Norway	NO	0.63	96.11
Pakistan	PK	61.44	26.03
Papua New Guinea	PG	1.43	0
Peru	PE	24.93	88.49
Philippines	PH	7.82	60.34
Poland	PL	1.96	92.35
Russia	RW	12.51	97
Rwanda	SN	80.66	0
Senegal	SN	81.75	42.69
South Africa	ZA	36.35	87.8
Sudan	SD	38.17	NA
Sweden	SE	4.41	98.49
Taiwan	TW	4.99	71.26
Tajikistan	TJ	72	58.85
Tanzania	TZ	19.45	NA
Turkey	TR	55.57	92.21
UAE	UAE	18.56	96.87
Ukraine	UA	26.85	78.76
United Kingdom	UK	3.5	99.82
United States	USA	33.84	92.57
Uzbekistan	UZ	72.31	0
Vietnam	VN	13.35	33.06
Zambia	ZM	5.58	65.61
Zimbabwe	ZW	4.97	75.44

Data source: Percentage wastewater treated values are taken from Wendling et al. (2018). Water scarcity footprint for 1m3 of water consumed are calculated as a product of AWARE factor (Boulay et al. 2018b) with 1m3 of water consumed.

Water Quality index

Evaluating WQI values, the DRC, from which a significant fraction of cobalt is extracted, is observed to be one of the countries with the lowest WQI score. Considering WSF and WQI together, results suggest that geographic variability may lead to tradeoffs in water impacts. Here, an example of gold is again used to illustrate these challenges. Globally, the top producer of gold is China (19% of global production in 2018), which has a relatively high WSF (Table A14). The same amount of gold could be sourced from the countries that have lower WSF, including Australia, Russia, Canada, Brazil, Ghana, Indonesia, Papua New Guinea (see Table A13). Sourcing from Papa New Guinea would reduce the WSF of gold by 98%, but this country has significant water pollution risks, given that the percentage of wastewater treated is close to zero (Wendling et al. 2018). On the other hand, materials may be sourced from countries with better

water quality (i.e., those that treat a higher percentage of wastewater), but that are experiencing water stress, as is the case for Australia for gold sourcing (see Fig. A2).

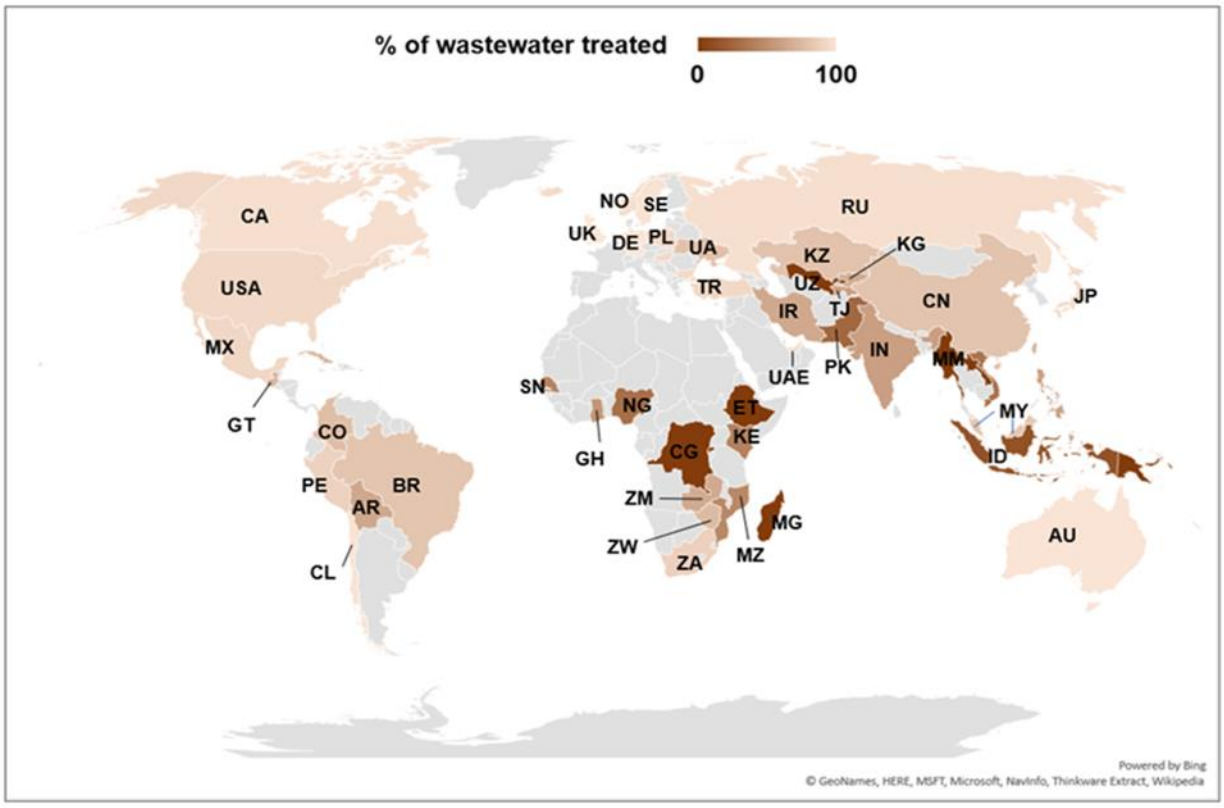


Figure A2. Water Quality Index, which is a measure of the percent of wastewater undergoing at least primary treatment in each producer countries. Color intensity corresponds to relative magnitude of risks, where darker shades suggest greater water quality risks.

1.5. Material recycling:

This study adapted the e-waste processing from Bakas et al. (2016) and Bigum et al. (2012). The scope of the recycling processes in shown in the Fig. A3. In general, the e-waste is collected and transferred to a facility to pre-treat. During this phase, ferrous and aluminum are separated and sent to the refining facility to recover the final product, as shown in the Fig. A3. The remaining mix of the scrap, which is rich in the critical and precious metals undergoes smelting and refining process to recover a wide range of metals. In this study, the metals that are considered to be recovered are copper, gold, silver, palladium, lead and indium, as suggested in Umicore documents (Hagelüken 2006). The decision to include certain metals is also influenced by the data available on the composition of these metals in e-waste, and recovery rates. Due to unavailability of quality data, the only input flow considered is the electricity. Any emissions due to the recycling processes are excluded due to data limitations. As a result, the recycling benefits in the study should be considered as an initial assessment. It is reported that 0.32 KWh of

electricity is required to process 1 kg of e-waste (Bakas, Herczeg, and Blikra Vea 2016). Mass composition of the metals recovered in the e-waste flows in 2018 were calculated using MFA model developed by (Althaf, Babbitt, and Chen 2019). Material recovery rates are collected from various sources (Bigum, Brogaard, and Christensen 2012; Caffarey 2012; Hagelüken 2006). To be consistent with the Simapro, we followed economic allocation principles while modelling e-waste recycling process. To do so, we collected the price information from (Althaf 2019), and an assumption is made that price of secondary metal is same as a primary metal (Söderholm and Ekvall 2019).

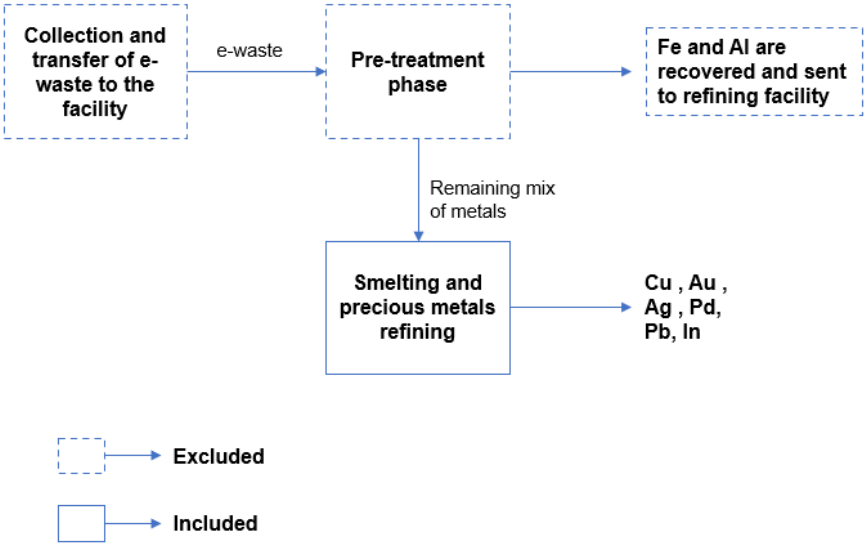


Figure A3. Scope of e-waste recycling processes considered in the study

Table A15. Summary of the data required to estimate the economic allocation of the values required to estimate the water scarcity footprint of material recycling process.

Metal	Mass of metals in 1 kg of e-waste	Recovery rates	Price of the metal ($/lb)	Price of the metal ($/kg)	Metal recovered (kg)	Economic value	Economic allocation
Cu	0.03	95%	2.90	6	0.0285	0.1800	0.600%
Au	0.00041	97%	18,000.00	40000	0.0003977	15.91	53.8%
Ag	0.019	98%	250.00	556	0.01862	10.34	35.0%
Pd	0.0001	98%	12,000.00	26667	0.000098	2.610	8.80%
Pb	0.213	95%	1.10	2	0.20235	0.4900	1.70%
In	0.0001	95%	160.00	356	0.000095	0.03000	0.100%
Total	0.26261	-	-	-	0.2500607	29.58	100.0%

Table A16. Impacts of a recycled gold

Impact	Unit	Value
Water scarcity footprint	m^3eq	93.15
Freshwater Ecotoxicity	kg 1,4 - DCB	10.0262
Freshwater Eutrophication	kg P eq	0.319563

1.5.1. Estimating impacts of recycled aluminum

We modelled the recycling of aluminum in the Simapro by adapting the processes block (Aluminium, cast alloy {RER}| treatment of aluminium scrap, post-consumer, prepared for recycling, at refiner | Cut-off, U). Electricity and water consumed in this process was used to model the recycling of aluminum from e-waste. from the ecoinvent database. Recovery rate of aluminum (98%) is collected from Boin and Bertram (2005). Water scarcity footprint of a recycled aluminum is estimated as 0.77 m^3 per kg.

Table A17. Impacts of a recycled aluminum

Impact	Unit	Value
Water scarcity footprint	m^3eq	0.77
Freshwater Ecotoxicity	kg 1,4 - DCB	0.00072
Freshwater Eutrophication	kg P eq	2.29E-05

2. Baseline results

2.1. Material level impacts

Table A18. Impacts per kilogram of metal produced

Materials	Pfister et al. (2009) (m^3eq)	AWARE (m^3eq)	Freshwater eutrophication (kg P eq)	Freshwater ecotoxicity (kg 1,4 DCB)	Aquatic acidification (kgSO2 eq)
Steel	7.27E-03	5.34E-01	1.05E-03	5.61E-02	1.05E-02
Aluminium	5.08E-02	3.67E+00	4.49E-03	3.86E-01	6.55E-02
Copper	4.60E-02	3.28E+00	9.80E-02	1.76E+01	4.75E-01
Nickel	6.09E-02	4.53E+00	3.14E-02	5.62E+00	2.53E+00
Magnesium	2.79E-02	1.90E+00	1.08E-02	8.00E-01	1.09E-01
Titanium	1.32E-01	8.37E+00	1.56E-02	6.76E-01	1.48E-01
Zinc	1.92E-02	1.40E+00	5.20E-03	7.81E-01	4.72E-02
Gold	7.16E+01	5.04E+03	5.07E+02	9.23E+04	1.79E+02
Silver	9.92E-01	7.15E+01	1.89E+00	3.31E+02	3.83E+00
Palladium	2.21E+01	1.80E+03	1.40E+01	2.57E+03	1.79E+03
Platinum	8.05E+01	6.47E+03	8.26E+01	1.52E+04	2.39E+03
Rhodium	9.31E+01	7.52E+03	8.39E+01	1.54E+04	4.27E+03
Manganese	1.16E-02	8.36E-01	2.23E-03	8.61E-01	2.38E-02
Tellurium	3.32E-02	2.39E+00	4.69E-02	8.04E+00	2.08E-01
Barite	1.74E-02	1.27E+00	2.76E-05	3.45E-03	1.04E-03
Cobalt	8.28E-02	5.98E+00	4.08E-03	3.66E-01	1.11E-01
Antimony	8.27E-02	6.09E+00	2.46E-01	4.42E+01	1.84E-01
Tin	1.24E-01	9.53E+00	9.17E-03	4.82E-01	4.58E-01
Lithium	3.61E-01	2.67E+01	3.17E-02	1.62E+00	2.82E-01
Indium	8.57E-01	6.23E+01	2.48E-01	3.72E+01	2.11E+00
Gallium	9.89E-01	7.47E+01	4.12E-02	3.06E+00	5.41E-01
Tantalum	1.32E+00	9.57E+01	1.49E-01	1.42E+01	2.12E+00
Cadmium	9.01E-03	6.59E-01	2.96E-03	1.46E-01	1.71E-02
Chromium	6.44E-02	4.75E+00	1.22E-02	1.09E+00	1.45E-01
Lead	7.39E-03	5.43E-01	2.80E-03	4.57E-01	3.22E-02
Mercury	1.28E-02	8.43E-01	4.48E-03	3.20E-01	1.10E-01
Lanthanum	1.26E-01	9.47E+00	7.35E-03	8.53E-01	9.56E-02
Praseodymium	1.97E-01	1.49E+01	1.15E-02	1.34E+00	1.50E-01
Neodymium	2.10E-01	1.58E+01	1.23E-02	1.43E+00	1.59E-01

2.2. Product level impact

2.2.1. Per smartphone

Table A19. Impacts of metals analyzed per smartphone

Materials	Water scarcity footprint (m^3eq)	Freshwater eutrophication (kg P eq)	Freshwater ecotoxicity (kg 1,4 DCB)	Aquatic acidification (kg SO2 eq.)
Steel	4.82E-03	9.52E-06	5.07E-04	9.50E-05
Aluminium	7.04E-02	8.60E-05	7.40E-03	1.26E-03
Copper	1.23E-02	3.68E-04	6.59E-02	1.78E-03
Nickel	7.37E-03	5.11E-05	9.15E-03	4.12E-03
Titanium	1.50E-03	2.81E-06	1.21E-04	2.67E-05
Zinc	1.08E-03	4.00E-06	6.01E-04	3.63E-05
Gold	7.46E-02	7.50E-03	1.37E+00	2.65E-03
Silver	1.76E-03	4.65E-05	8.13E-03	9.40E-05
Palladium	3.99E-02	3.09E-04	5.68E-02	3.96E-02
Manganese	1.28E-03	3.42E-06	1.32E-03	3.65E-05
Tellurium	4.61E-06	9.05E-08	1.55E-05	4.01E-07
Barite	8.84E-04	1.92E-08	2.40E-06	7.23E-07
Cobalt	9.73E-03	6.64E-06	5.96E-04	1.80E-04
Antimony	6.91E-05	2.80E-06	5.02E-04	2.08E-06
Tin	5.26E-03	5.07E-06	2.66E-04	2.53E-04
Lithium	1.56E-02	1.85E-05	9.45E-04	1.64E-04
Indium	3.12E-06	1.24E-08	1.86E-06	1.05E-07
Gallium	4.93E-05	2.72E-08	2.02E-06	3.57E-07
Tantalum	3.11E-04	4.83E-07	4.62E-05	6.90E-06
Cadmium	4.61E-08	2.07E-10	1.02E-08	1.20E-09
Chromium	1.55E-03	3.99E-06	3.54E-04	4.73E-05
Lead	1.52E-04	7.85E-07	1.28E-04	9.04E-06
Praseodymium	3.39E-04	2.63E-07	3.05E-05	3.42E-06
Neodymium	2.93E-03	2.27E-06	2.64E-04	2.95E-05

2.2.2. Per laptop

Table A20. Impacts of metals analyzed per laptop

Materials	Water scarcity footprint (m^3eq)	Freshwater eutrophication (kg P eq)	Freshwater ecotoxicity (kg 1,4 DCB)	Aquatic acidification (kg SO2 eq)
Steel	2.11E-01	4.16E-04	2.22E-02	4.15E-03
Aluminum	1.61E+00	1.96E-03	1.69E-01	2.87E-02
Copper	3.09E-01	9.23E-03	1.65E+00	4.48E-02
Nickel	6.86E-03	4.76E-05	8.52E-03	3.84E-03
Titanium	3.83E-03	7.15E-06	3.09E-04	6.80E-05
Zinc	4.80E-03	1.79E-05	2.68E-03	1.62E-04
Gold	6.22E-01	6.25E-02	1.14E+01	2.21E-02
Silver	3.12E-02	8.25E-04	1.44E-01	1.67E-03
Palladium	7.08E-01	5.49E-03	1.01E+00	7.03E-01
Manganese	1.94E-03	5.18E-06	2.00E-03	5.53E-05
Tellurium	1.27E-05	2.50E-07	4.29E-05	1.11E-06
Barite	2.47E-03	5.37E-08	6.71E-06	2.02E-06
Cobalt	4.08E-01	2.78E-04	2.50E-02	7.56E-03
Antimony	7.46E-03	3.02E-04	5.42E-02	2.25E-04
Tin	9.35E-02	8.99E-05	4.73E-03	4.49E-03
Lithium	2.12E-01	2.52E-04	1.29E-02	2.24E-03
Indium	2.74E-05	1.09E-07	1.64E-05	9.27E-07
Gallium	8.78E-04	4.84E-07	3.60E-05	6.36E-06
Tantalum	5.52E-03	4.83E-07	4.62E-05	6.90E-06
Cadmium	3.10E-07	1.39E-09	6.84E-08	8.03E-09
Chromium	9.66E-04	2.49E-06	2.21E-04	2.95E-05
Lead	3.92E-03	2.02E-05	3.30E-03	2.33E-04
Lanthanum	4.18E-05	3.24E-08	3.76E-06	4.21E-07
Praseodymium	3.83E-05	2.97E-08	3.45E-06	3.86E-07
Neodymium	8.23E-04	6.39E-07	7.42E-05	8.30E-06

2.3. Sector level impacts

Table A21. Impacts analyzed for the materials consumed in the electronics sector globally

Materials	Water scarcity footprint (m^3eq)	Freshwater eutrophication (kg P eq.)	Freshwater ecotoxicity (kg 1,4 DCB)	Aquatic acidification (kg SO2 eq.)
Steel	5.07E+10	1.00E+08	5.33E+09	1.00E+09
Aluminum	2.12E+10	2.59E+07	2.22E+09	3.77E+08
Copper	2.63E+09	7.84E+07	1.40E+10	3.80E+08
Nickel	6.11E+08	4.24E+06	7.59E+08	3.42E+08
Magnesium	1.26E+08	7.11E+05	5.28E+07	7.19E+06
Titanium	2.11E+08	3.93E+05	1.70E+07	3.74E+06
Zinc	3.09E+09	1.15E+07	1.73E+09	1.04E+08
Gold	1.33E+09	1.34E+08	2.44E+10	4.73E+07
Silver	4.44E+08	1.17E+07	2.05E+09	2.38E+07
Palladium	6.49E+07	5.04E+05	9.24E+07	6.44E+07
Platinum	3.23E+07	4.13E+05	7.60E+07	1.19E+07
Rhodium	1.50E+07	1.68E+05	3.09E+07	8.54E+06
Manganese	3.17E+08	8.47E+05	3.27E+08	9.05E+06
Tellurium	9.09E+04	1.78E+03	3.06E+05	7.89E+03
Barite	1.93E+09	4.19E+04	5.24E+06	1.58E+06
Cobalt	6.70E+08	4.57E+05	4.10E+07	1.24E+07
Antimony	2.53E+08	1.03E+07	1.84E+09	7.64E+06
Tin	3.97E+08	3.82E+05	2.00E+07	1.90E+07
Lithium	1.33E+09	1.59E+06	8.11E+07	1.41E+07
Indium	3.98E+07	1.58E+05	2.38E+07	1.34E+06
Gallium	2.35E+07	1.29E+04	9.62E+05	1.70E+05
Tantalum	8.27E+07	1.28E+05	1.23E+07	1.83E+06
Cadmium	2.80E+06	1.26E+04	6.19E+05	7.26E+04
Chromium	1.04E+10	2.69E+07	2.39E+09	3.19E+08
Lead	7.32E+07	3.78E+05	6.17E+07	4.35E+06
Mercury	5.06E+05	2.69E+03	1.92E+05	6.61E+04
Lanthanum	8.20E+07	6.36E+04	7.38E+06	8.27E+05
Praseodymium	1.10E+08	8.56E+04	9.94E+06	1.11E+06
Neodymium	5.01E+08	3.89E+05	4.51E+07	5.05E+06

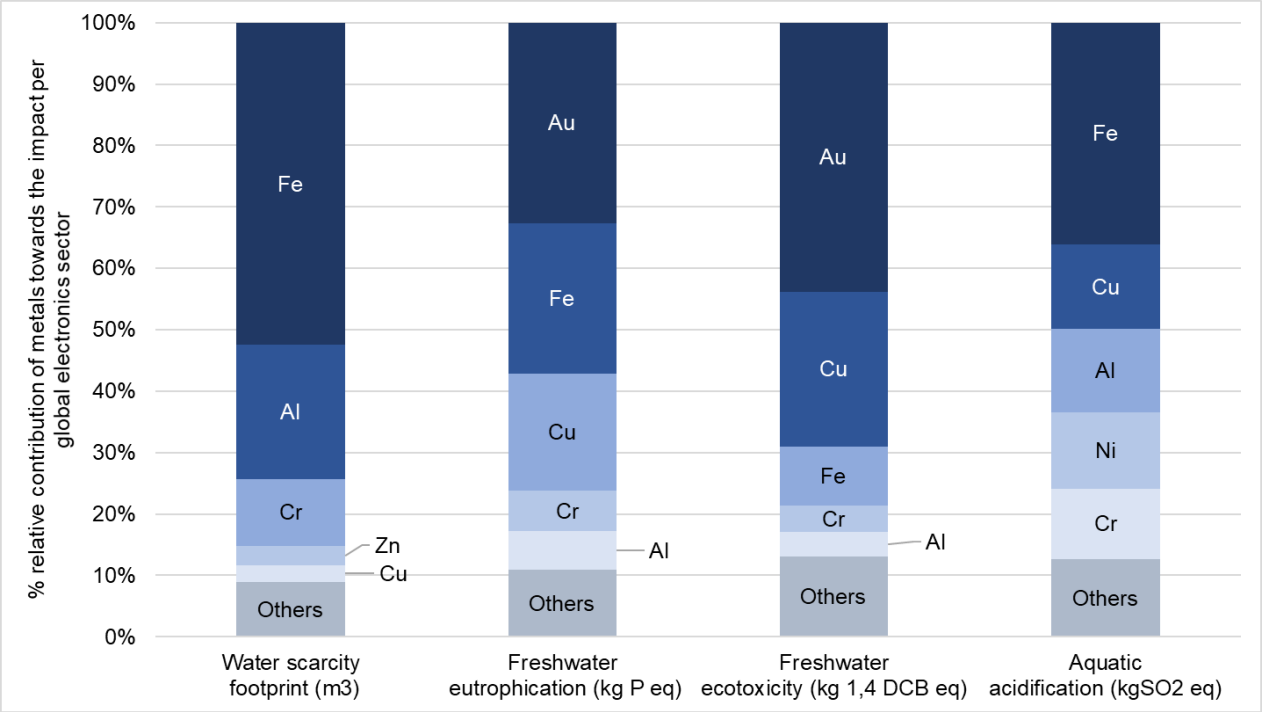


Figure A4. Material hotspot trends observed for the metals per global electronics sector for all the impact categories quantified in this study. Only top 5 metals are shown. “Others” include the aggregate value of rest of the metals considered in the scope of the study. Darker the color of the metal, higher the impact.

2.4. Process contribution and network analysis

In this section, the processes contributing the greatest towards water quality impacts are shown for gold as an example

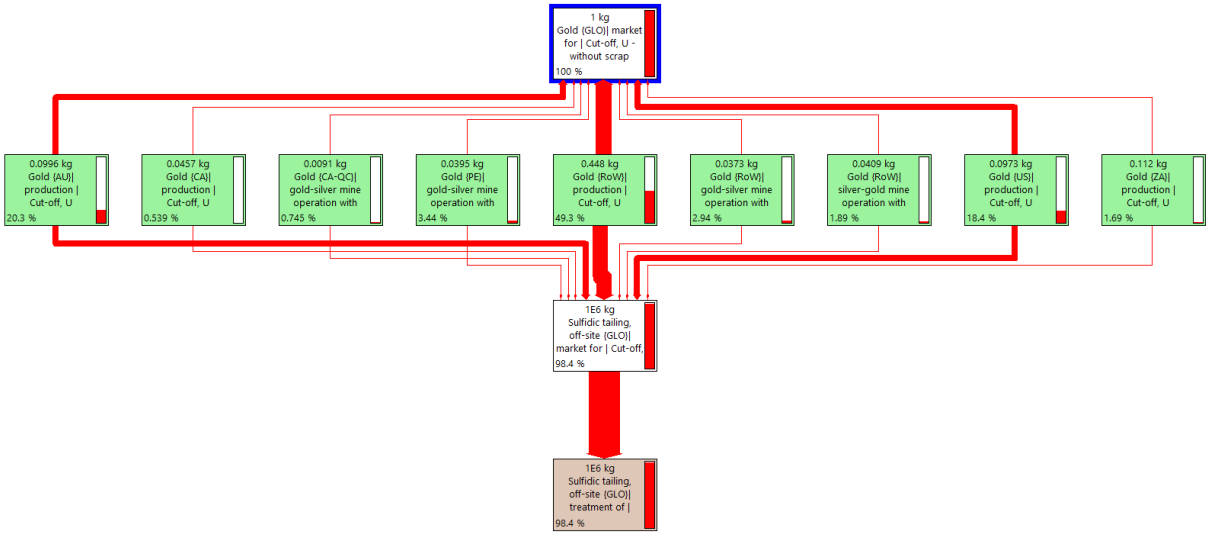


Figure A5. Process contribution and network analysis of global block of gold mining and production for freshwater ecotoxicity (kg 1,4 DCB eq)

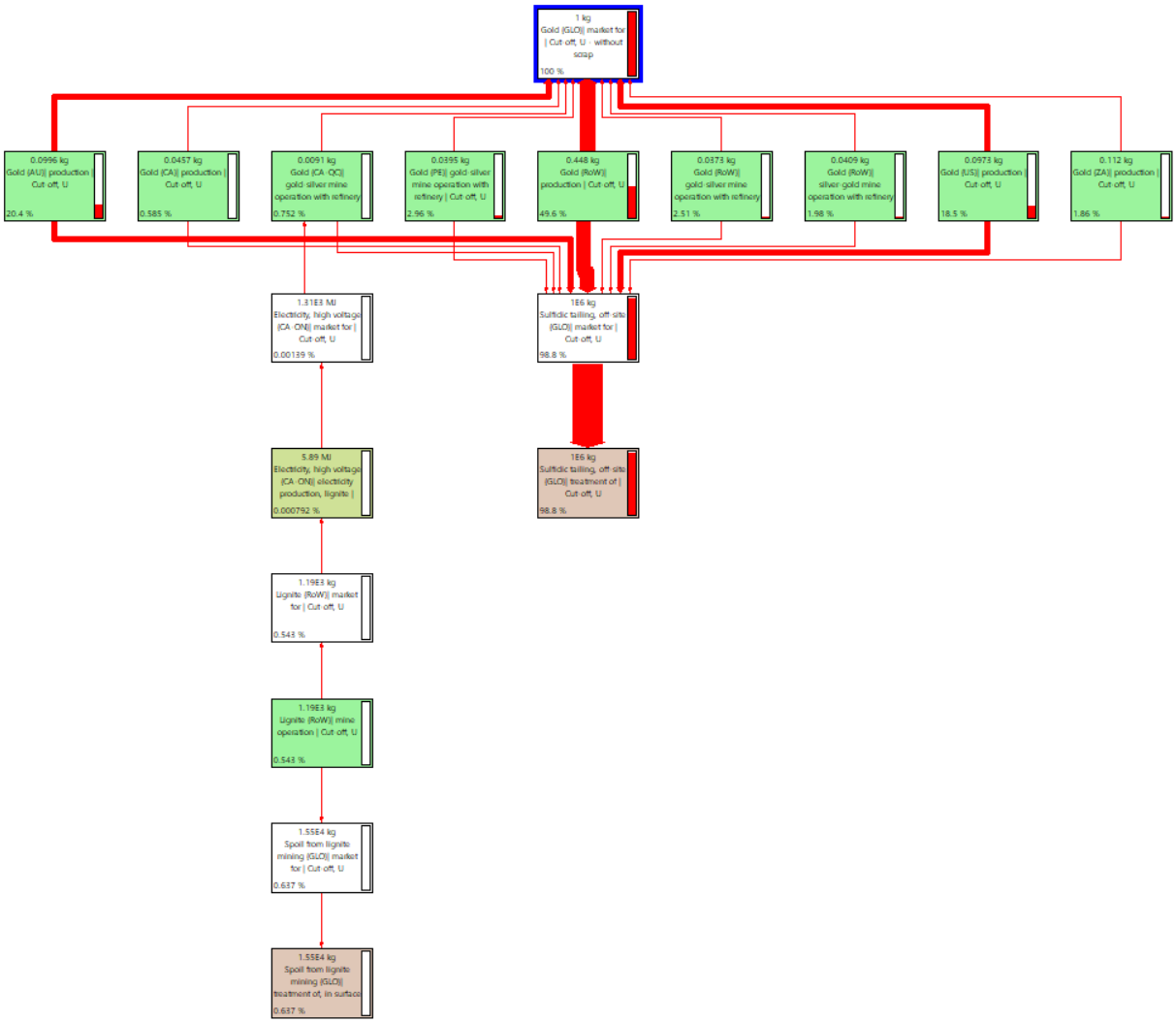


Figure A6. Process contribution and network analysis of global block of gold mining and production for freshwater eutrophication (kg P eq.)

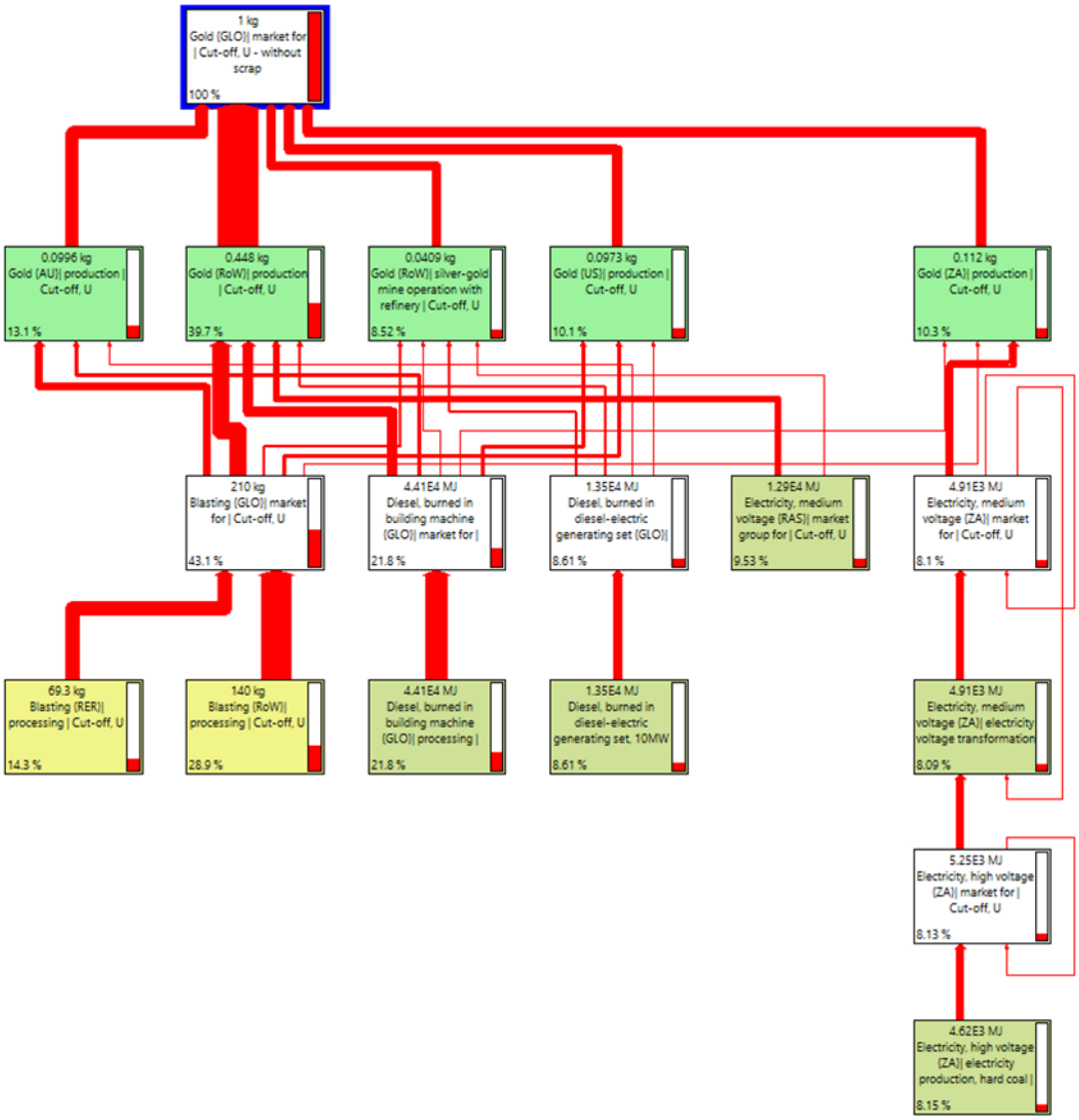


Figure A7. Process contribution and network analysis of global block of gold mining and production for aquatic acidification (kg SO2 eq.)

2.5. Contributing factors towards product level impacts

Table A22. Average metal content per smartphone vs relative contribution of per smartphone impacts

Material	Mass in (g)	WSF (m^3 eq /kg of metal)	Per smartphone (relative contribution)
Al	19.16	3.671	28%
Cu	3.755	3.283	5%
Fe	9.0262	0.5341	2%
Ni	1.627	4.527	3%
Zn	0.7696	1.397	0.43%
Ti	0.1798	8.366	0.60%
Au	0.0148	5042.7	30%
Ag	0.02457	71.51	1%
Pd	0.0221	1803.7	16%
Sb	0.01135	6.0904	0.03%
Ba	0.6963	1.269	0.35%
Co	1.627	5.981	4%
Ga	0.0006620	74.7	0.02%
In	0.00005	62.34	0.001%
Li	0.5833	26.66	6%
Mn	1.535	0.8355	1%
Ta	0.003251	95.73	0.12%
Te	0.001932	2.3908	0.002%
Sn	0.5522	9.531	2%
Nd	0.1852	15.82	1%
Pr	0.02282	14.85	0.13%
Pb	0.28035	0.5425	0.06%
Cr	0.04564	4.748	0.06%
Cd	0.000014	0.659	0.000018%

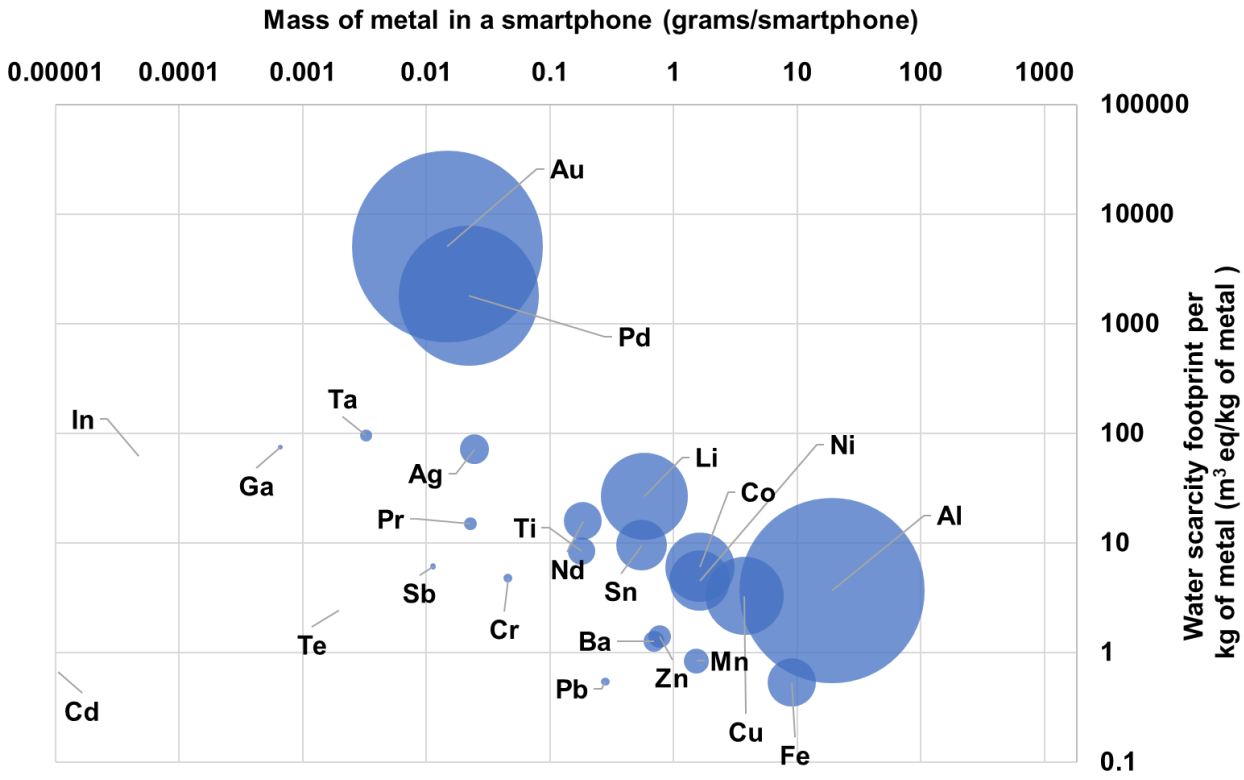


Figure A8. Comparison of material concentration in a smartphone and water scarcity footprint on per kilogram basis. Both axes are presented on a log scale. Circle size corresponds to the percent of the material contribution towards water scarcity of a smartphone (from 1 to 100%).

Table A23. Average metal content per laptop vs relative contribution of per laptop impacts

Material	Mass in (g)	WSF (m^3 eq/kg of metal)	Per laptop (relative contribution)
Al	437.2	3.672	38%
Cu	94.18	3.283	7%
Fe	394.5	0.5341	5%
Ni	1.515	4.527	0.2%
Zn	3.434	1.397	0.1%
Ti	0.4580	8.366	0.090%
Au	0.1232	5042.7	15%
Ag	0.4362	71.51	1%
Pd	0.3926	1803.7	17%
Sb	1.2256	6.0904	0.2%
Ba	1.945	1.269	0.1%
Co	68.20	5.981	10%
Ga	0.01174	74.72	0.02%
In	0.000444	62.34	0.001%
Li	7.956	26.66	5%
Mn	2.325	0.8355	0.05%
Te	0.005333	2.391	0.0003%
Sn	9.806	9.531	2%
Nd	0.05201	15.83	0.02%
Pr	0.002581	14.85	0.001%
La	0.00441	9.470	0.001%
Pb	7.224	0.543	0.1%
Cr	0.2033	4.748	0.02%
Cd	0.000466	0.6590	0.00001%
Ta	0.0577	95.73	0.13%

3. Scenario analysis

3.1. Supply chain diversification

To evaluate the water scarcity footprint of gold in the countries it is mined and produced, the first step was to identify the water consumption during mining and production processes. Due to lack of data available for each individual mining site, we used global average water consumed for a gold mining and production processes by using ReCIPE (H) midpoint method. This average value is multiplied with the water scarcity index of each country to estimate the water scarcity footprint of gold mining and production.

Water consumption of 1 kg of gold = 143 m^3

Table A24. Water scarcity footprint values per kg of gold

Sourcing of gold	**Water scarcity footprint (m^3eq/kg of metal)**
Current global production	5043
Russia	1793
Canada	1061
Brazil	311

Data source: Current global production value is estimated using Simapro (See Table S14). Values for Russia, Canada, and Brazil are estimated using data from Table S12 and water consumption estimated per kg of gold

Table A25. Material hotspots trends observed in Supply chain diversification scenario analysis of gold in a smartphone towards water scarcity footprint (m^3eq)

Material	**Gold- Current global production**	**Material**	**Gold - Russia**	**Material**	**Gold - Canada**	**Material**	**Gold - Brazil**
Rest	0.03000	Rest	0.03	Rest	0.03	Rest	0.03
Cobalt	0.009732	Cobalt	0.009732	Cobalt	0.009732	Gold	0.00046
Copper	0.01233	Copper	0.01233	Copper	0.01233	Cobalt	0.009732
Lithium	0.01555	Lithium	0.01555	Lithium	0.01555	Copper	0.01233
Palladium	0.03988	Gold	0.02654	Gold	0.01570	Lithium	0.01555
Aluminum	0.0703699	Palladium	0.03988	Palladium	0.03988	Palladium	0.03988
Gold	0.07465	Aluminum	0.07036	Aluminum	0.07036	Aluminum	0.07036

3.2.Material substitution

This study considered magnesium and ABS as the alternatives for the aluminum casing material in a 14-inch laptop. The BOM of a 14-inch laptop with three different casing materials is summarized below in the Table -16. Water scarcity footprint of ABS is estimated using Simapro as 2.43 m3.

Table A26. Material composition of 14.1 in Laptop BOM for different casing materials

Material	Aluminum	Plastics	Magnesium
Casing	444.9	336.2	388.3
Al	299.1	299.1	299.1
Cu	94.12	94.12	94.12
Fe	281.9	281.9	281.9
Ni	1.0608	1.0608	1.0608
Au	0.1306	0.1306	0.1306
Ag	0.4620	0.4620	0.4620
Pd	0.4158	0.4158	0.4158
Co	47.74	47.74	47.74
Ga	0.01243	0.01243	0.01243
In	0.0004162	0.0004162	0.0004162
Li	5.569	5.569	5.569
Ta	0.06112	0.06112	0.06112
Sn	10.38	10.38	10.38
Pb	7.650	7.650	7.650
La	0.004670	0.004670	0.004670
Nd	0.05509	0.05509	0.05509
Pr	0.002733	0.002733	0.002733
Cr	0.2154	0.2154	0.2154
Sb	1.298	1.298	1.298
Te	0.005647	0.005647	0.005647
Ti	0.4851	0.4851	0.4851
Mn	2.462	2.462	2.462
Ba	2.0606	2.0606	2.0606
Cd	0.0004940	0.0004940	0.0004940
Zn	3.637	3.637	3.637

Data source: Babbitt et al. (2020)

Water scarcity footprint

Table A27. Material hotspot trends observed for substituting aluminum casing with alternate materials in a laptop casing for water scarcity footprint (m^3eq)

Material	Aluminum	Material	Plastic	Material	Magnesium
Rest	0.758676	Rest	0.758676	Rest	0.758676
Cu	0.309009	Cu	0.309009	Cu	0.309009
Au	0.658404	Au	0.658404	Au	0.658404
Pd	0.750013	Pd	0.750013	Casing material	0.739482
Al	1.098354	Casing material	0.816966	Pd	0.750013
Casing material	1.633919	Al	1.098354	Al	1.098354

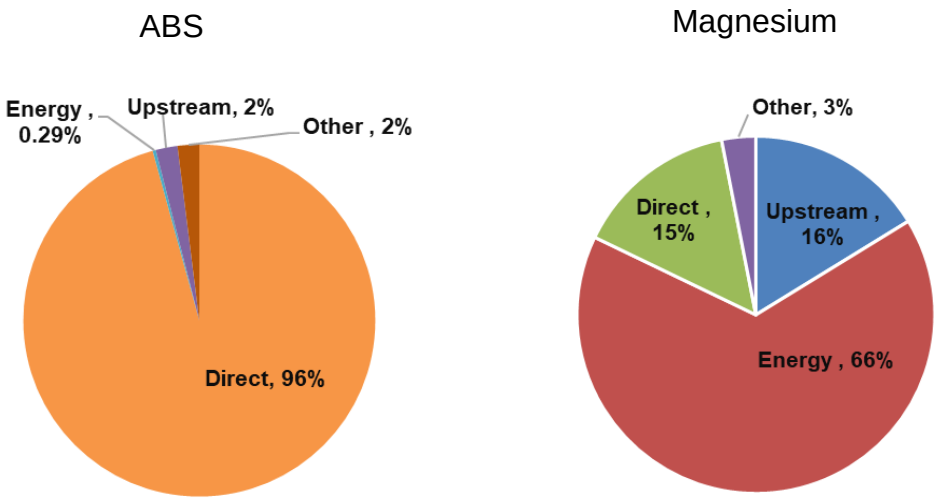
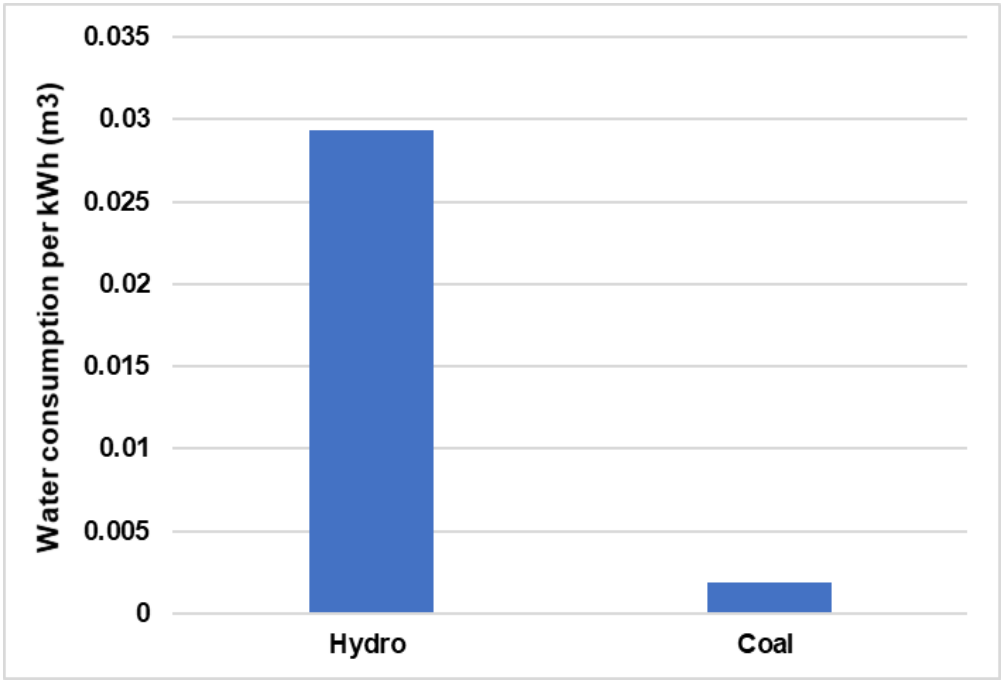


Figure A9. Process contribution analysis of magnesium and plastic for water scarcity footprint (m^3eq)

Hydroelectricity vs coal generated power

We compared the water consumption and water scarcity footprint of 1 kWh of electricity generated through hydro and coal using Simapro. ReCIPE method was used to estimate water consumption and AWARE was used to estimate scarcity.



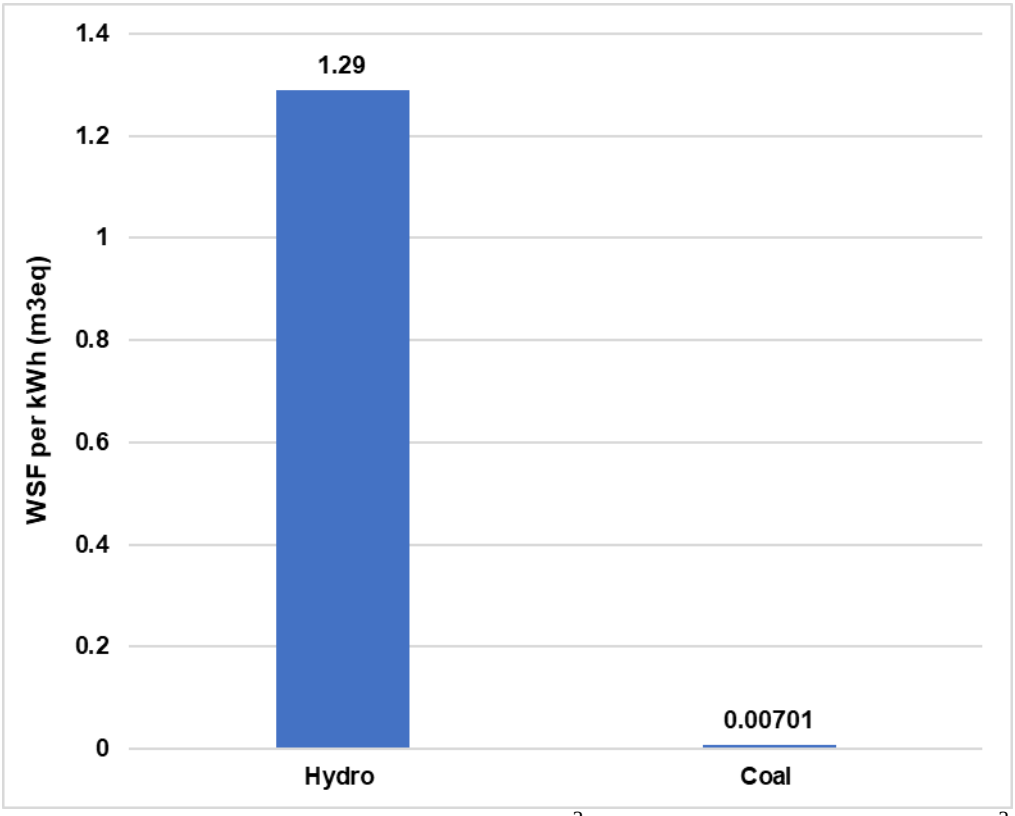


Figure A10. Comparison of water consumption (m^3) and water scarcity footprint (m^3eq.) per kWh of electricity generation through hydroelectric power and coal generated power

Water quality impacts

Freshwater ecotoxicity

Table A28. Material hotspot trends observed for substituting aluminum casing with alternate materials in a laptop casing for freshwater ecotoxicity (kg 1,4 DCB eq)

Material	Aluminum	Material	ABS	Material	Magnesium
Cd	7.19E-08	Cd	7.19E-08	Cd	7.19E-08
Pr	3.66E-06	Pr	3.66E-06	Pr	3.66E-06
La	3.98E-06	La	3.98E-06	La	3.98E-06
Ba	7.10E-06	Ba	7.10E-06	Ba	7.10E-06
In	1.55E-05	In	1.55E-05	In	1.55E-05
Ga	3.81E-05	Ga	3.81E-05	Ga	3.81E-05
Te	4.54E-05	Te	4.54E-05	Te	4.54E-05
Nd	7.85E-05	Nd	7.85E-05	Nd	7.85E-05
Cr	2.34E-04	Cr	2.34E-04	Cr	2.34E-04
Ti	3.28E-04	Ti	3.28E-04	Ti	3.28E-04
Ta	8.68E-04	Ta	8.68E-04	Ta	8.68E-04
In	1.55E-03	In	1.55E-03	In	1.55E-03
Mn	2.12E-03	Mn	2.12E-03	Mn	2.12E-03
Zn	2.84E-03	Zn	2.84E-03	Zn	2.84E-03
Pb	3.49E-03	Pb	3.49E-03	Pb	3.49E-03
Sn	5.00E-03	Sn	5.00E-03	Sn	5.00E-03
Ni	5.96E-03	Ni	5.96E-03	Ni	5.96E-03
Li	9.03E-03	Li	9.03E-03	Li	9.03E-03
Fe	1.58E-02	Casing material	9.03E-03	Fe	1.58E-02
Co	1.75E-02	Fe	1.58E-02	Co	1.75E-02
Sb	5.74E-02	Co	1.75E-02	Sb	5.74E-02
Al	1.15E-01	Sb	5.74E-02	Al	1.15E-01
Ag	1.53E-01	Al	1.15E-01	Ag	1.53E-01
Casing material	1.72E-01	Ag	1.53E-01	Casing material	3.11E-01
Pd	1.07E+00	Pd	1.07E+00	Pd	1.07E+00
Cu	1.65E+00	Cu	1.65E+00	Cu	1.65E+00
Au	1.21E+01	Au	1.21E+01	Au	1.21E+01

Freshwater eutrophication

Table A29. Materia hotspot trends observed for substituting aluminum with alternate materials in a laptop casing for freshwater eutrophication (kg P eq.)

Material	Aluminum	Material	Plastics	Material	Magnesium
Cd	1.46E-09	Cd	1.46E-09	Cd	1.46E-09
Pr	3.15E-08	Pr	3.15E-08	Pr	3.15E-08
La	3.43E-08	La	3.43E-08	La	3.43E-08
Ba	5.68E-08	Ba	5.68E-08	Ba	5.68E-08
In	1.03E-07	In	1.03E-07	In	1.03E-07
Te	2.65E-07	Te	2.65E-07	Te	2.65E-07
Ga	5.12E-07	Ga	5.12E-07	Ga	5.12E-07
Nd	6.76E-07	Nd	6.76E-07	Nd	6.76E-07
Cr	2.63E-06	Cr	2.63E-06	Cr	2.63E-06
Mn	5.49E-06	Mn	5.49E-06	Mn	5.49E-06
Ti	7.57E-06	Ti	7.57E-06	Ti	7.57E-06
Ta	9.08E-06	Ta	9.08E-06	Ta	9.08E-06
Zn	1.89E-05	Zn	1.89E-05	Zn	1.89E-05
Pb	2.14E-05	Pb	2.14E-05	Pb	2.14E-05
Ni	3.33E-05	Ni	3.33E-05	Ni	3.33E-05
Sn	9.53E-05	Sn	9.53E-05	Sn	9.53E-05
Li	1.77E-04	Casing material	1.13E-04	Li	1.77E-04
Co	1.95E-04	Li	1.77E-04	Co	1.95E-04
Fe	2.97E-04	Co	1.95E-04	Fe	2.97E-04
Sb	3.20E-04	Fe	2.97E-04	Sb	3.20E-04
Ag	8.74E-04	Sb	3.20E-04	Ag	8.74E-04
Al	1.34E-03	Ag	8.74E-04	Al	1.34E-03
Casing material	2.00E-03	Al	1.34E-03	Casing material	4.18E-03
Pd	5.82E-03	Pd	5.82E-03	Pd	5.82E-03
Cu	9.22E-03	Cu	9.22E-03	Cu	9.22E-03
Au	6.62E-02	Au	6.62E-02	Au	6.62E-02

3.3. Material recycling

3.3.1. Gold in a smartphone

The benefits that one can achieve from material recycling are shown in terms of increasing recycled content. Table 15 summarizes the net impact by introducing 0 to 100% recycled gold to the primary material.

Water scarcity footprint

Table A30. Water scarcity footprint values of gold that is a mix of both primary and recycled material

Gold	Water scarcity footprint (m^3eq/kg)
35% RC	3310
50% RC	2568
70% RC	1578

Note: RC indicates recycled content. The values of a primary and secondary material mix are estimated by interpolating values from Table S14

Table A31. Material hotspots trends observed in a using a recycled content scenario for gold in a smartphone towards water scarcity footprint (m^3)

Material	**Gold-Current global production**	**Material**	**Gold (35% RC)**	**Material**	**Gold (50% RC)**	**Material**	**Gold (70% RC)**
Rest	0.028032	Rest	0.028032	Rest	0.028032	Rest	0.028032
Cobalt	0.009733	Cobalt	0.009733	Cobalt	0.009733	Cobalt	0.009733
Copper	0.01233	Copper	0.01233	Copper	0.01233	Copper	0.01233
Lithium	0.015551	Lithium	0.015551	Lithium	0.015551	Lithium	0.015551
Palladium	0.039885	Palladium	0.039885	Gold	0.038018	Gold	0.023362
Aluminum	0.07037	Gold	0.049004	Palladium	0.039885	Palladium	0.039885
Gold	0.0747	Aluminum	0.07037	Aluminum	0.07037	Aluminum	0.07037

Water quality impacts

Table A32. Freshwater eutrophication and ecotoxicity values of gold that is a mix of both primary and recycled material

Gold	Freshwater eutrophication (kg P eq.)	Freshwater ecotoxicity (kg SO2 eq.)
35% RC	329.4	60012
50% RC	253.5	46165
70% RC	152.2	27703

Freshwater ecotoxicity

Table A33. Material hotspots trends observed in a using a recycled content scenario for gold in a smartphone for freshwater ecotoxicity (kg 1,4 DCB eq)

Material	Gold - current global production	Material	Gold (35% RC)	Material	Gold (50% RC)	Material	Gold (70% RC)
Rest	0.00540	Rest	0.00540	Rest	0.00540	Rest	0.00540
Aluminum	0.007397	Aluminum	0.007397	Aluminum	0.007396646	Aluminum	0.007397
Silver	0.008123	Silver	0.00813	Silver	0.00813	Silver	0.00813
Nickel	0.009147	Nickel	0.009147	Nickel	0.009147	Nickel	0.009147
Palladium	0.05676	Palladium	0.05676	Palladium	0.05676	Palladium	0.05676
Copper	0.06592	Copper	0.06592	Copper	0.06592	Copper	0.06592
Gold	1.367	Gold	0.8884	Gold	0.6834	Gold	0.4101

Freshwater eutrophication

Table A34. Material hotspots trends observed in a using a recycled content scenario for gold in a smartphone for freshwater eutrophication (kg P eq.)

Material	Gold-Current global production	Material	Gold (35% RC)	Material	Gold (50% RC)	Material	Gold (70% RC)
Rest	0.00006	Rest	0.00006	Rest	0.00006	Rest	0.00006
Silver	0.00004647	Silver	0.00004647	Silver	0.00004647	Silver	0.00004647
Nickel	0.00005108	Nickel	0.00005108	Nickel	0.00005108	Nickel	0.00005108
Aluminum	0.00008606	Aluminum	0.00008606	Aluminum	0.00008606	Aluminum	0.00008606
Palladium	0.000309	Palladium	0.000309	Palladium	0.000309	Palladium	0.000309
Copper	0.000368	Copper	0.000368	Copper	0.000368	Copper	0.000368
Gold	0.007501	Gold	0.004877	Gold	0.003753	Gold	0.002253

3.3.2. Aluminum in a laptop

Water scarcity footprint

Table A35. Water scarcity footprint values of aluminum that is a mix of both primary and recycled material

Aluminum	Water scarcity footprint (m^3eq/kg)
45% RC	2.36
70% RC	1.64
90% RC	1.06

Table A36. Material hotspots trends observed in a using a recycled content scenario for aluminum in a laptop for water scarcity footprint (m^3eq)

Material	Aluminum	Material	Aluminum (45% RC)	Material	Aluminum (70% RC)	Material	Aluminum (90% RC)
Rest	0.758676	Rest	0.758676	Rest	0.758676	Rest	0.758676
Cu	0.309009	Cu	0.309009	Cu	0.309009	Cu	0.309009
Au	0.658404	Au	0.658404	Au	0.658404	Casing material	0.471759
Pd	0.750013	Pd	0.750013	Casing material	0.730017	Au	0.658404
Al	1.098354	Casing material	1.051437	Pd	0.750013	Pd	0.750013
Casing material	1.633919	Al	1.098354	Al	1.098354	Al	1.098354

Water quality impacts

Table A37. Freshwater ecotoxicity and eutrophication values for aluminum that is a mix of both primary and recycled material

Aluminum	Freshwater Ecotoxicity (kg 1,4 DCB)	Freshwater Eutrophication (kg P eq.)
45%	0.212602	0.00248
70%	0.116292	0.001363
90%	0.039244	0.00047

Freshwater ecotoxicity

Table A38. Material hotspots trends observed in a using a recycled content scenario for aluminum in a laptop for freshwater ecotoxicity (kg 1,4DCB eq)

Materials	Aluminum	Materials	Al (45% RC)	Materials	Al (70%RC)	Materials	Al (90%RC)
Cd	7.19E-08	Cd	7.19E-08	Cd	7.19E-08	Cd	7.19E-08
Pr	3.66E-06	Pr	3.66E-06	Pr	3.66E-06	Pr	3.66E-06
La	3.98E-06	La	3.98E-06	La	3.98E-06	La	3.98E-06
Ba	7.10E-06	Ba	7.10E-06	Ba	7.10E-06	Ba	7.10E-06
In	1.55E-05	In	1.55E-05	In	1.55E-05	In	1.55E-05
Ga	3.81E-05	Ga	3.81E-05	Ga	3.81E-05	Ga	3.81E-05
Te	4.54E-05	Te	4.54E-05	Te	4.54E-05	Te	4.54E-05
Nd	7.85E-05	Nd	7.85E-05	Nd	7.85E-05	Nd	7.85E-05
Cr	2.34E-04	Cr	2.34E-04	Cr	2.34E-04	Cr	2.34E-04
Ti	3.28E-04	Ti	3.28E-04	Ti	3.28E-04	Ti	3.28E-04
Ta	8.68E-04	Ta	8.68E-04	Ta	8.68E-04	Ta	8.68E-04
Mn	2.12E-03	Mn	2.12E-03	Mn	2.12E-03	Mn	2.12E-03
Zn	2.84E-03	Zn	2.84E-03	Zn	2.84E-03	Zn	2.84E-03
Pb	3.49E-03	Pb	3.49E-03	Pb	3.49E-03	Pb	3.49E-03
Sn	5.00E-03	Sn	5.00E-03	Sn	5.00E-03	Sn	5.00E-03
Ni	5.96E-03	Ni	5.96E-03	Ni	5.96E-03	Ni	5.96E-03
Li	9.03E-03	Li	9.03E-03	Li	9.03E-03	Li	9.03E-03
Fe	1.58E-02	Fe	1.58E-02	Fe	1.58E-02	Fe	1.58E-02
Co	1.75E-02	Co	1.75E-02	Co	1.75E-02	Casing material	1.75E-02
Sb	5.74E-02	Sb	5.74E-02	Casing material	5.17E-02	Co	1.75E-02
Al	1.15E-01	Casing material	9.46E-02	Sb	5.74E-02	Sb	5.74E-02
Ag	1.53E-01	Al	1.15E-01	Al	1.15E-01	Al	1.15E-01
Casing material	1.72E-01	Ag	1.53E-01	Ag	1.53E-01	Ag	1.53E-01
Pd	1.07E+00	Pd	1.07E+00	Pd	1.07E+00	Pd	1.07E+00
Cu	1.65E+00	Cu	1.65E+00	Cu	1.65E+00	Cu	1.65E+00
Au	1.21E+01	Au	1.21E+01	Au	1.21E+01	Au	1.21E+01

Freshwater eutrophication

Table A39. Material hotspots trends observed in a using a recycled content scenario for aluminum in a laptop for freshwater eutrophication (kg P eq)

Materials	Aluminum	Materials	Al (45% RC)	Materials	Al (70% RC)	Materials	Al (90% RC)
Cd	1.46E-09	Cd	1.46E-09	Cd	1.46E-09	Cd	1.46E-09
Pr	3.15E-08	Pr	3.15E-08	Pr	3.15E-08	Pr	3.15E-08
La	3.43E-08	La	3.43E-08	La	3.43E-08	La	3.43E-08
Ba	5.68E-08	Ba	5.68E-08	Ba	5.68E-08	Ba	5.68E-08
In	1.03E-07	In	1.03E-07	In	1.03E-07	In	1.03E-07
Te	2.65E-07	Te	2.65E-07	Te	2.65E-07	Te	2.65E-07
Ga	5.12E-07	Ga	5.12E-07	Ga	5.12E-07	Ga	5.12E-07
Nd	6.76E-07	Nd	6.76E-07	Nd	6.76E-07	Nd	6.76E-07
Cr	2.63E-06	Cr	2.63E-06	Cr	2.63E-06	Cr	2.63E-06
Mn	5.49E-06	Mn	5.49E-06	Mn	5.49E-06	Mn	5.49E-06
Ti	7.57E-06	Ti	7.57E-06	Ti	7.57E-06	Ti	7.57E-06
Ta	9.08E-06	Ta	9.08E-06	Ta	9.08E-06	Ta	9.08E-06
Zn	1.89E-05	Zn	1.89E-05	Zn	1.89E-05	Zn	1.89E-05
Pb	2.14E-05	Pb	2.14E-05	Pb	2.14E-05	Pb	2.14E-05
Ni	3.33E-05	Ni	3.33E-05	Ni	3.33E-05	Ni	3.33E-05
Sn	9.53E-05	Sn	9.53E-05	Sn	9.53E-05	Sn	9.53E-05
Li	1.77E-04	Li	1.77E-04	Li	1.77E-04	Li	1.77E-04
Co	1.95E-04	Co	1.95E-04	Co	1.95E-04	Co	1.95E-04
Fe	2.97E-04	Fe	2.97E-04	Fe	2.97E-04	Casing material	2.09E-04
Sb	3.20E-04	Sb	3.20E-04	Sb	3.20E-04	Fe	2.97E-04
Ag	8.74E-04	Ag	8.74E-04	Casing material	6.07E-04	Sb	3.20E-04
Al	1.34E-03	Casing material	1.10E-03	Ag	8.74E-04	Ag	8.74E-04
Casing material	2.00E-03	Al	1.34E-03	Al	1.34E-03	Al	1.34E-03
Pd	5.82E-03	Pd	5.82E-03	Pd	5.82E-03	Pd	5.82E-03
Cu	9.22E-03	Cu	9.22E-03	Cu	9.22E-03	Cu	9.22E-03
Au	6.62E-02	Au	6.62E-02	Au	6.62E-02	Au	6.62E-02

3.4.Material substitution of recycled magnesium in a laptop casing

A high-level analysis is done to understand the benefits of substituting a recycled magnesium in a casing of a laptop. To estimate water scarcity footprint of a recycled magnesium, we adapted the electricity value (i.e., 0.32 kWh) used to model recycling of e-waste scrap (Appendix A Section 1.5). Recovery rate of magnesium is assumed to be around 75 – 80% (Sibley 2011), and recycled content of magnesium in current supply chain is around 36 - 41% (Graedel et al. 2011). Water scarcity footprint of recycled magnesium is estimated as 0.047 m^3/kg.

Table A40. Reduction in total water scarcity footprint (m^3eq.) of materials in a laptop by substituting recycled magnesium as a casing

Materials	Magnesium	Materials	Magnesium (45% RC)	Materials	Magnesium (80% RC)	Materials	Magnesium (90% RC)
Cd	3.26E-07	Cd	3.26E-07	Cd	3.26E-07	Cd	3.26E-07
Te	1.35E-05	Te	1.35E-05	Te	1.35E-05	Te	1.35E-05
Pr	4.06E-05	Pr	4.06E-05	Pr	4.06E-05	Pr	4.06E-05
In	2.59E-05	In	2.59E-05	In	2.59E-05	In	2.59E-05
La	4.42E-05	La	4.42E-05	La	4.42E-05	La	4.42E-05
Nd	8.72E-04	Nd	8.72E-04	Nd	8.72E-04	Nd	8.72E-04
Ga	9.29E-04	Ga	9.29E-04	Ga	9.29E-04	Ga	9.29E-04
Cr	1.02E-03	Cr	1.02E-03	Cr	1.02E-03	Cr	1.02E-03
Mn	2.06E-03	Mn	2.06E-03	Mn	2.06E-03	Mn	2.06E-03
Ba	2.62E-03	Ba	2.62E-03	Ba	2.62E-03	Ba	2.62E-03
Ti	4.06E-03	Ti	4.06E-03	Ti	4.06E-03	Ti	4.06E-03
Pb	4.15E-03	Pb	4.15E-03	Pb	4.15E-03	Pb	4.15E-03
Ni	4.80E-03	Ni	4.80E-03	Ni	4.80E-03	Ni	4.80E-03
Zn	5.08E-03	Zn	5.08E-03	Zn	5.08E-03	Zn	5.08E-03
Ta	5.85E-03	Ta	5.85E-03	Ta	5.85E-03	Ta	5.85E-03
Sb	7.90E-03	Sb	7.90E-03	Sb	7.90E-03	Sb	7.90E-03
Ag	3.30E-02	Ag	3.30E-02	Ag	3.30E-02	Ag	3.30E-02
Sn	9.90E-02	Sn	9.90E-02	Sn	9.90E-02	Casing material	9.05E-02
Li	1.48E-01	Li	1.48E-01	Li	1.48E-01	Sn	9.90E-02
Fe	1.51E-01	Fe	1.51E-01	Fe	1.51E-01	Li	1.48E-01
Co	2.86E-01	Co	2.86E-01	Casing material	2.35E-01	Fe	1.51E-01
Cu	3.09E-01	Cu	3.09E-01	Co	2.86E-01	Co	2.86E-01
Au	6.58E-01	Casing material	4.87E-01	Cu	3.09E-01	Cu	3.09E-01
Casing material	7.39E-01	Au	6.58E-01	Au	6.58E-01	Au	6.58E-01
Pd	7.50E-01	Pd	7.50E-01	Pd	7.50E-01	Pd	7.50E-01
Al	1.10E+00	Al	1.10E+00	Al	1.10E+00	Al	1.10E+00

4. Uncertainty analysis

Table A41. Maximum, minimum, and average content of gold present in a smartphone, and respective water scarcity footprint values.

Gold content in a smartphone	Mass in kg	WSF (m^3eq)
Average	0.0000148	0.07463
Max	0.000034	0.17150
Min	0.00000057	0.002858

Table A42. Percentage contribution of gold towards total WSF(m^3eq) of a smartphone in an average, minimum and maximum concentration scenario

Material	**Gold-Current global production**	**Material**	**Gold-Current global production**	**Material**	**Gold-Current global production**
Rest	0.028032	Rest	0.028032	Rest	0.028032
Cobalt	0.009733	Cobalt	0.009733	Gold	0.002858
Copper	0.01233	Copper	0.01233	Cobalt	0.009733
Lithium	0.015551	Lithium	0.015551	Copper	0.01233
Palladium	0.039885	Palladium	0.039885	Lithium	0.015551
Aluminum	0.07037	Aluminum	0.07037	Palladium	0.039885
Gold	0.074635	Gold	0.171504	Aluminum	0.07037
	0.250536		0.347405		0.17876
	30%		49%		2%

Appendix B

1. Resolving uncertainties and discrepancies in polymer identification

Polymers in the electronic products have additives, including flame retardants, pigments, and stabilizers. As a result, challenges were encountered when analyzing the spectrum given out by know-it-all software. A methodology was developed that accounts for marking, know it all software output, spectrum and properties of polymer when finalizing a polymer type. Most of the challenges were observed in identifying the ABS samples and a blend of polymers. Some of the examples included below

Example -1:

In several instances, polymer samples that were marked as acrylonitrile butadiene styrene (ABS) were matched with Copolymer polystyrene grafted fullerene or styrene/acrylonitrile. In such cases, the spectrum was examined for the peaks (e.g., between 2000 and 2300 nm) and compared to ABS spectrums observed in the past literature (Makri, Hahladakis, and Gidarakos 2019). When peaks of the spectrum match with the ones in the literature, final determination was made to be ABS. An example of this scenario is shown in Figure B1.

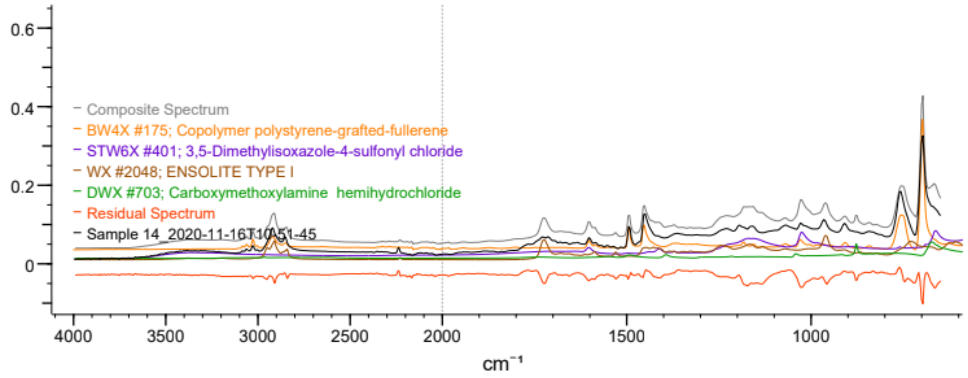
Example - 2:

In case of polymer samples that are blends (e.g., PC+ABS or ABS+PET) as per the marking, only one polymer type (e.g., ABS or PC) was matched in the know-it-All software output for some. In such cases, the spectrum of the sample was examined for the peaks of the other polymer that is not observed in the software output. If the spectrum of polymer blends is available in the past literature, they are used to compare to the sample spectrum to examine for peaks. In cases where spectrums are not readily available, polymer type manufacturer labels were a final one.

Example - 3:

In a few cases, polymer samples with ABS label matched with Poly (methyl methacrylate) (PMMA) in the Know-it-All software. PMMA is usually added to ABS to have higher impact strength, hardness, and excellent sheen in the products (Szabo et al. 2011). In those cases, visual cues and polymer properties were taken into consideration along with the software output and label and final determination was a mix of ABS and PMMA.

A)



B)

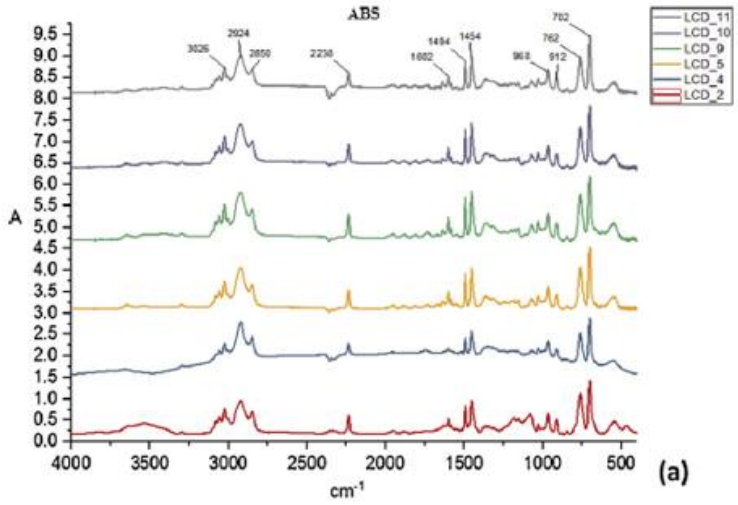


Figure B1. A) Spectrum of the ABS sample from the polymer samples analyzed in the lab and B) Spectrum of ABS sample obtained from literature (Makri, Hahladakis, and Gidarakos 2019)

2. Identifying unlabeled polymers using broader methodology developed

During the sample preparation phase, some of the polymer samples were observed to be either too small or thick making it harder to be readily analyzed. In such instances, many ways were applied to identify the unknown. One way was that polymer samples of same color and component were grouped together based on visual identification. Then, one or two samples from each group are analyzed under FTIR to identify the polymer type, which is then expected to be the polymer type of the entire group. This step was mainly applied when identifying polymer samples from e-waste recyclers. For example, in LCD mixed plastics box, black color polymer samples that looked like they came from the bezel part of the display are grouped together. From this group, a polymer sample is analyzed and found to be polystyrene. Then, it was assumed that all other samples in this group were polystyrene.

Second, if there was a product from the same brand and series, and polymer types are known, then this was extended to the products of series of the same brand with unknown polymer samples. For example, let's say, the polymer type of a keyboard from Dell laptop D600 is unknown, and we know the polymer type of keyboard in laptop D610, then we assumed that polymer type of keyboard from D600 is the same as D610. In cases where we were not able to group samples, then the polymer sample is considered to be unknown.

3. Estimating average composition of product categories

To match the material analysis flow model the average of the product categories listed in the Table B1 were estimated. The data sources for each of these product categories are listed in the Table. Data collected from e-waste recycler is integrated with polymer composition data from disassembled products in the lab for TVs, Monitors and Desktop categories. However, to integrate data from e-waste recycler, these values needed to be recalculated to match the product data. This is because e-waste recycler samples were mainly from casing components and does not represent the entire product. To estimate the polymer composition for the rest of product and acrylic, the following assumptions were made.

Table B1. Data sources used to estimate average composition of polymers for each product category

Product	Data sources
Basic mobile phone	Lab disassembly
Blu-ray player	Lab disassembly
CRT monitor	E-waste recycler casings
CRT TV	E-waste recycler casings
Desktop CPU	Lab disassembly + E-waste recycler casings
Digital camcorder	E-waste recycler shredded residue
Digital camera	E-waste recycler shredded residue
DVD player	Lab disassembly
Gaming console	Lab disassembly
Laptop	Lab disassembly
LED Monitor	Lab disassembly + E-waste recycler casings
LED TV	Lab disassembly + E-waste recycler casings
LCD monitor	Lab disassembly + E-waste recycler casings
LCD TV	Lab disassembly + E-waste recycler casings
MP3 player	E-waste recycler shredded residue
Plasma TV	Literature (Stobbe, 2007)
Printer	Lab disassembly
Smart phone	Lab disassembly
Tablet and e-reader	Lab disassembly

First, the average percentage contribution of casings, rest of the product and screen (when applicable) in the products was estimated using the data from products disassembled in the lab. This percentage along with the mass of the total sampled plastics from the boxes of e-waste recycler was used to estimate the total mass of a product for each category. Then, this new total mass was used to recalculate the polymer composition of casings in e-waste recycler samples. Then, for displays, the average composition of acrylic calculated is estimated to represent the amount of acrylic from e-waste recycler samples. Finally, polymer composition of shredded residue from e-waste recyclers is assumed to represent the polymer composition of the rest of the products for all the categories. This assumption was reasonable as shredded residue as per recycling facility represents the composition of polymers that were hard to separate from the product. The polymer composition of shredded mixed residue is recalculated to represent the percentage contribution for this portion for each category obtained from lab data. Once, the e-waste recycler samples are calculated to represent the whole product, these data points were combined along with our lab data to calculate average composition. For other mobile products, which include digital camcorders, digital cameras, and MP3 players, polymer composition of shredded residue is assumed to be representative as these products would most likely end up in an electronic shredder.

Another assumption was made in calculating averages when data from products disassembled was considered. The products in which mass of total sampled plastics accounted for 70% or greater of the total mass of plastics in a product (obtained from BOM) are only included in the average composition calculation of product categories. As a result, an assumption was made that polymer breakdown observed in the samples analyzed is representative of the product category and extends to any other polymer samples that were not analyzed.

4. Polymer diversity

Table B2. Polymer type and number identified for each product category

Product	Number	Type of polymers
TV	6	ABS, PC/ABS, PC, HIPS, PP, Acrylic
Monitors	6	ABS, PC/ABS, PC, ABS/PMMA, ABS/PET, Acrylic
Desktop	7	ABS, PC/ABS, PC, HIPS, ABS/PMMA, PBT, PPS, PS/PPE
CRT displays	4	ABS, PC, HIPS, ABS/PMMA
Printers	11	ABS, PC/ABS, PC, HIPS, ABS/PMMA, PC/PTFE, ABS/PTFE, PS/PPE, PS/PPO, POM, PPO
Laptops	9	Acrylic, ABS, PC/ABS, PC, ABS/PMMA, PS/PPE, PET, POM, PPTP
Phones	2	PC/ABS, PC
DVD/gaming	8	ABS, PC/ABS, PC, HIPS, PBT, PET, PPE, POM
Tablets	3	PC, PPA, Acrylic
Blu ray disc players	5	ABS/PMMA, PMMA, HIPS, POM, ABS
Shredded residue	7	ABS, PC/ABS, PC, HIPS, PET, PPE, PP

5. Average profile of polymers

Table B3. Average composition of polymers per product category illustrating breakdown by presence of BFRs or no presence and untested. Values are in percent by mass

Polymers	TV	Monitor	Desktop	CRT Display	Printer	Laptop	Phone	Media consoles	Tablet	Shredded residue
Acrylic	16.40%	34.43%				17.95%			19.50%	
ABS + BFR	14.0%	1.31%	0.689%	19.84%	0.125%			5.20%		2.02%
ABS+ No BFR	3.66%	31.45%	54.79%	22.82%	51.15%			12.07%		87.22%
ABS + untested	8.54%	8.69%	0.60%	0.99%	0.15%	6.06%		34.87%		1.06%
PC+ ABS + BFR	0.00%					1.16%				
PC+ ABS+ No BFR	9.54%	4.22%	7.43%		0.45%	50.70%				0.96%
PC+ABS + untested	0.00%		17.69%			8.19%	16.67%			
PC + BFR	0.00%									
PC+ No BFR	5.02%	1.49%	7.50%	4.55%	1.37%	5.38%			16.30%	4.06%
PC+ untested	6.34%	2.83%	0.42%	0.25%	1.43%	6.44%	73.50%	12.22%	55.37%	1.22%
HIPS + BFR	27.31%	1.65%		40.53%				18.88%		
HIPS + No BFR	2.30%	0.98%	0.22%	6.36%	27.03%					0.65%
HIPS + untested	2.89%			0.78%				0.56%		
Unknown + BFR				1.90%						
Unknown + No BFR	1.93%	0.14%			2.30%			2.32%		
Unknown + untested		0.27%		1.23%	7.01%	2.73%	9.83%	6.63%	5.81%	
Others + BFR			0.00%	0.75%						
Others + No BFR	1.16%	12.48%	1.59%		6.14%	0.87%		0.97%		1.49%
Others + untested	0.941%	0.06%	9.07%		2.84%	0.53%		6.27%	3.02%	1.32%

6. National level e-waste flows

Table B4. Baseline polymer flows entering the e-waste stream in the U.S. in 2018. Quantities are in kg

Polymers	Acrylic	ABS	PC+ABS	PC	HIPS	Unknown	Others
TVs	25,186,665	40,196,854	14,661,407	17,450,075	49,927,193	2,972,116	3,227,623
Monitors	4,643,326	5,589,624	569,597	582,129	355,358	55,283	1,691,048
CRT Displays	0	60,588,958	0	6,655,666	66,185,155	4,348,072	1,040,336
Desktop	0	11,457,301	5,148,459	1,617,790	45,014	0	2,178,982
Printer	0	43,889,606	388,002	2,389,816	23,074,358	7,946,318	7,663,497
Laptop	3,071,695	1,037,088	10,276,625	2,022,399	0	468,021	239,645
Phones	0	0	1,091,391	4,813,290	0	643,666	0
Media consoles	0	16,987,340	0	3,982,141	6,333,819	2,916,532	2,359,094
Tablets	533,688	0	0	1,961,905	0	159,056	82,746
Other mobile products	0	1,938,955	23,952	113,510	13,909	0	60,465

Note: TVs include LCD, LED and plasma models. Monitors include LED and LCD models.CRT Displays include both TVs and Monitors. Other mobile products include MP3 players, cameras, and digital camcorders. Media consoles include DVD players, gaming consoles and Blu-ray disc players. E-readers are included in the Tablets

Table B5. Baseline polymer flows entering the waste stream breakdown by presence of BFRs, no presence and untested for FPDs, CRTs, Desktops and printers. Values are in kilograms

Polymers	Flat panel Displays	CRT Displays	Desktops	Printers
Acrylic	29,829,990	0	0	0
ABS + BFR	21,626,961	27,538,819	140,924	106,655
ABS+ No BFR	9,863,236	31,680,972	11,204,073	43,654,873
ABS + untested	14,298,563	1,369,166	123,096	128,078
PC+ ABS + BFR	0	0	0	0
PC+ ABS+ No BFR	15,228,721	0	1,520,055	388,002
PC+ABS + untested	0	0	3,617,611	0
PC + BFR	0	0	0	0
PC+ No BFR	7,915,199	6,313,375	1,532,763	1,167,450
PC+ untested	10,117,004	342,291	85,027	1,222,366
HIPS + BFR	42,174,933	56,269,812	0	0
HIPS + No BFR	3,662,726	8,826,143	45,014	23,074,358
HIPS + untested	4,444,892	1,089,199	0	0
Unknown + BFR	0	2,640,853	0	0
Unknown + No BFR	2,990,817	0	0	1,960,875
Unknown + untested	36,582	1,707,218	0	5,985,443
Others + BFR	0	1,040,336	0	0
Others + No BFR	3,466,234	0	324,812	5,239,792
Others + untested	1,452,437	0	1,854,170	2,423,705
Total	167,108,295	138,818,186	20,447,546	85,351,597

Note: Flat panel displays include LED TVs, LED Monitors, LCD Monitors, LCD TVs, Plasma TVs.

Table B6. Baseline polymer flows entering the waste stream breakdown by presence of BFRs, no presence and untested for Laptops, Phones, Media consoles, Tablets and Other mobile products. Values are in kilograms

Polymers	Laptops	Phones	Media consoles	Tablets	Other mobile products
Acrylic	3,071,695	0	0	533,688	0
ABS + BFR	0	0	1,693,182	0	43,545
ABS+ No BFR	0	0	3,933,551	0	1,875,941
ABS + untested	1,037,088	0	11,360,607	0	22,804
PC+ ABS + BFR	198,343	0	0	0	0
PC+ ABS+ No BFR	8,676,834	0	0	0	20,618
PC+ABS + untested	1,401,448	1,091,391	0	0	0
PC + BFR	0	0	0	0	0
PC+ No BFR	920,132	0	0	446,193	87,237
PC+ untested	1,102,267	4,813,290	3,982,141	1,515,712	26,273
HIPS + BFR	0	0	6,150,145	0	0
HIPS + No BFR	0	0	0	0	13,909
HIPS + untested	0	0	183,674	0	0
Unknown + BFR	0	0	0	0	0
Unknown + No BFR	0	0	756,843	0	0
Unknown + untested	468,021	643,666	2,159,689	159,056	0
Others + BFR	0	0	0	0	0
Others + No BFR	149,421	0	315,849	0	32,076
Others + untested	90,224	0	2,043,245	82,746	28,389
Total	17,115,472	6,548,347	32,578,926	2,737,395	2,150,792

Note: Other mobile products include MP3 players, cameras, and digital camcorders. Media consoles include DVD players, gaming consoles and Blu-ray disc players. E-readers are included in the Tablets

Table B7. Polymer flows aggregated as “Others”

Polymers	Quantity (kg)
ABS/PMMA	5,767,339
ABS/PTFE	314,496
ABS/PET	708,455
PC/PTFE	1,285,256
PS/PPE	1,628,657
PS/PPO	202,819
POM	1,082,417
PPO	1,774,398
PP	1,976,674
PET	292,780
PPE	294,778
PVC	448,202
PA	101,663
PE	53,042
PUR	707,222
PBT	1,816,368
PPTP	6,123
PPA	82,746

7. XRF data

A total of 550 samples were tested by XRF in this study

Table B8. Samples observed for each element tested under XRF

Element	Number of observed	% Of samples observed
Bromine	435	79%
Antimony	136	25%
Chlorine	24	4%
Titanium	475	86%
Lead	141	26%
Chromium	121	22%
Mercury	51	9%

8. FTIR validation

Table B9. Validation data of FTIR instrument and know it all software

Product category	Product	Actual marking on the plastic part	Spectrum match in Lab solutions IR	Spectrum match in know it All	Polymer conclusion comparison
LCD TV	Samsung 40" LN40B500P3F (2009)	HIPS	PS	PS	Same
LED TV	LG Smart TV 43" 43UH610A-UJ (2016)	PC	PC	Propylene	Diff
LED TV	Sanyo	ABS	ABS	Copolymer of polystyrene grafted fullerene	Diff
LED Monitor	Dell U2417H (June 2017)	ABS	PMMA	PMMA	Same
LED Monitor	Dell U2417H (June 2017)	ABS	PMMA	PMMA	Same
LED Monitor	Dell U2417H (June 2017)	ABS	PMMA	PMMA	Same
LED Monitor	Dell U2417H (June 2017)	PC+ABS	PC	PC	Same
LED Monitor	Dell U2417H (June 2017)	ABS	PMMA	PMMA	Same
LED Monitor	Dell U2417H (June 2017)	ABS	ABS	Copolymer of polystyrene grafted fullerene	Same
LED Monitor	Dell U2417H (June 2017)	ABS	PMMA	PMMA	Same
LED monitor	Dell 23" S2316M LED monitor (2016)	ABS+PET	blend of PMMA	Copolymer of polystyrene grafted fullerene	Diff
LED monitor	Dell 23" S2316M LED monitor (2016)	ABS+PET	ABS	Styrene acrylonitrile copolymer	Same
LED monitor	LG 24" 24MP56HA LED monitor (2014)	ABS	Styrene/Acrylonitrile copolymer	Styrene acrylonitrile copolymer	Same
LCD monitor	HP 19" W1907 flat panel monitor (2006)	ABS	ABS	Copolymer of polystyrene grafted fullerene	Same
LCD monitor	HP 19" W1907 flat panel monitor (2006)	ABS	PMMA	Unknown compound	Diff
LCD monitor	HP 19" W1907 flat panel monitor (2006)	ABS	ABS	Styrene acrylonitrile copolymer	Same
LCD monitor	Dell 15" E157FPB (approx. 2008)	ABS	blend of PMMA	Unknown compound	Diff
LCD monitor	HP 19" W1907 flat panel monitor (2006)	ABS	Abs and blend of others	PMMA and styrene acrylonitrile copolymer	Same
Laptop	HP EliteBook 17" 8730w (2008)	Unlabeled	PC	Unknown compound	Diff
Laptop	Dell CPX H5005T PPX (2000)	Unlabeled	ABS	Matched with PS	Diff
Laptop	HP EliteBook 14.1" 6930p (2008)	Unlabeled	PC/PBT	Unknown compound	Diff
Desktop	Dell Desktop (Precision T3500) 2010	ABS	Styrene/acrylonitrile copolymer	Copolymer of polystyrene grafted fullerene	Same

Desktop	Dell Desktop (OPTIPLEX) DCNE1F 2010	ABS	PS	Unknown compound	Diff
Desktop	HP TouchSmart 20" 310-1125f (2011)	ABS	PC	Poly(bisphenol-A-carbonate) and Acrylonitrile/butadiene/styrene	Same
DVD/gaming	RCA DVD/CD player (approx. 2004)	ABS	PMMA	Unknown compound	Diff
DVD/gaming	Sony	PS	PMMA	Unknown compound	Diff
DVD/gaming	Play station	Unlabeled	PMMA	Acrylic	Same
E-waste recycling	CRT Plastic box - 1	Unlabeled	PMMA	PMMA as well	Same
E-waste recycling	CRT Plastic box - 1	Unlabeled	PMMA	Matched with tetraphenyl naphthalene	Diff
Shredded samples	Sample bag 1	Unlabeled	PMMA (80%) AND 79.9% match with ABS	Unknown compound	Diff
Shredded samples	Sample bag 2	ABS	ABS	Copolymer of polystyrene grafted fullerene	Same
Shredded samples	Sample bag 3	ABS	ABS	Copolymer of polystyrene grafted fullerene	Same
Shredded samples	Sample bag 4	Unlabeled	ABS	Styrene acrylonitrile copolymer	Same
Shredded samples	Sample bag 5	Unlabeled	ABS	Unknown compound	Diff

9. Uncertainty analysis

Table B10. Relative composition of observed polymers for three TV models in which external casings are made of HIPS, ABS, HIPS&PC/ABS compared to baseline average

Polymers	Flat panel TVs -baseline	Flat panel TVs - HIPS	Flat panel TVs - ABS	Flat panel TVs – HIPS & PC/ABS
Acrylic	16%	21%	22%	14%
ABS	26%	11%	54%	2%
PC/ABS	10%	0%	10%	26%
PC	11%	6%	0%	3%
HIPS	33%	62%	0%	55%
Unknown	2%	0%	8%	0%
Others	2%	0%	6%	0%
Total	100.0%	100.0%	100.0%	100.0%

Table B11. Polymer flows entering the e-waste stream in three scenarios modeled

Polymers	Baseline	HIPS	ABS	HIPS & PC/ABS
Acrylic	33,435,373	40,578,433	42,338,969	29,116,059
ABS	181,685,725	159,102,996	223,909,626	144,037,555
PC+ABS	32,159,432	17,498,025	33,464,350	58,165,465
PC	41,588,721	32,834,227	24,138,646	29,459,024
HIPS	145,934,806	190,990,114	96,007,613	180,225,692
Unknown	19,509,063	16,536,948	28,835,333	16,536,948
Others	18,543,436	15,315,814	24,162,020	15,315,814
Total	472,856,557	472,856,557	472,856,557	472,856,557

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