detritus | Multidisciplinary Journal for Circular Economy and Sustainable Management of Residues

CISA

ELECTRONIC WASTE TREATMENT FLOWS IN NORWAY: INVESTIGATING RECYCLING RATES AND EMBODIED EMISSIONS

Kim Rainer Mattson, Lærke Lindgreen Lauritsen and Johan Berg Pettersen

Industrial Ecology Program, Department of Energy and Process Engineering, Norwegian University of Science and Technology, Norway

Article Info:
Received:
25 August 2023
Revised:
18 December 2023
Accepted:
18 December 2023
Available online:
31 December 2023

Keywords:
WEEE,
Waste Management
Material Flows
Carbon Footprint
Embedded materials
Circular Economy

ABSTRACT

Norway is one of the countries in Europe generating the most waste from electrical and electronic equipment (WEEE) per capita. Extended producer responsibility schemes are incorporated as part of the national waste policy, with clear goals towards recovery of materials from the waste fraction. Investigating the WEEE flows in Norway, we observe clear improvements needed in the transparency of the sector, and based on the information gathered, we estimate lower recycling of materials than provided through official statistics based on reporting. 68% of WEEE sent to recycling treatments are recycled into reusable material. Accounting for WEEE occurring outside of the treatment system, only 58% is recovered for recycling. We also estimate the CO_2-equivalent emissions of different End-of-Life treatments of WEEE, and the embodied CO_2-eq emissions of each WEEE category, illustrating 1) what category carry the largest environmental burden with respect to its embedded materials, and 2) the environmental impact of specific treatment options within the system. We show how the recycling rate of precious metals have significant influence over the environmental impact recovery potential of the system. It is not just the amount of material that is recycled that is important, including a proxy for expended emissions effectively illustrates the need for more precise policy implementation to ensure a functional circular economy.

1. INTRODUCTION

Collection and safe treatment of Waste from Electrical and Electronic Equipment (WEEE) has been established through regulations, both in Norway and in the EU, with a strong focus on avoiding emissions of hazardous compounds embedded in the waste products (Oguchi et al., 2013). In recent years the focus has expanded to include material recycling of WEEE, with a recent strategic policy shift to recovering critical raw materials (CRMs). The European Commission has introduced the Critical Raw Materials Act, with a clear push towards improving sustainability and circularity within a commodity market with strategic importance for future renewable energy and technology sector growth in Europe (European Commission. Joint Research Centre., 2021).

Waste data from Eurostat and the Norwegian statistical agency (SSB) shows that Norway and the other Scandinavian countries are generating and collecting significant amounts of WEEE per capita compared to the rest of the EU. In 2018 the consumer WEEE per capita collection was 14.5 kg for Norway and 7.8 kg for the EU (Constantinescu et al., 2022).With an average recycling rate of 80.5% between 2015 and 2021 reported by official statistics (SSB, table 10513), one could naturally assume that the system for WEEE management is a significant contributor to the circular economy.

Multiple studies have investigated both the economic and the environmental effects of collecting WEEE for recycling (Messmann et al., 2019; Peng & Shehabi, 2022; Rodriguez-Garcia, n.d.). Ismail & Hanafiah (2019) provide a review of 61 Life Cycle Assessment (LCA) studies performed between 2005 and 2018 on WEEE collection and treatment. Although the quality of the studies was high, one apparent weakness is a general lack of holistic or full-system perspective when analyzing WEEE treatment. Most studies focus their environmental analysis on a limited number of EEE products and/or potential treatments, these studies do generally show significant environmental benefits of recycling WEEE (Ismail & Hanafiah, 2019). A study by De Meester et al., (2019) provide a mixed Material Flow Analysis (MFA) and LCA assessment of WEEE treatment in Belgium, also a country with highly developed collection and treatment systems for waste. The study shows that there is significant material loss in the treatment system due to sorting activity, and significant functional loss due to downcycling of materials, effectively embedding precious

* Corresponding author:
Kim Rainer Mattson
email: kim.r.mattson@ntnu.no

https://doi.org/10.31025/2611-4135/2023.18331

metals and other critical materials in lower value recycled materials (De Meester et al., 2019). Multiple studies have highlighted the same issue with sorting and separation of critical/precious metals from WEEE in Europe (Bigum et al., 2012; Parajuly et al., 2017; Ueberschaar et al., 2017).

Recently the complexity of modern electric and electronic equipment (EEE) has gained more attention in the literature, highlighting the complex composition of the resulting WEEE (Babbitt et al., 2020, 2021; Graedel et al., 2022), and the various treatments available for extended recovery of CRMs (Chen et al., 2021). As this brief review of the literature shows, collection and treatment of WEEE is a significant challenge in terms of separating and valorizing CRMs embedded in the waste stream. However, due to large metal fractions embedded in the waste, recycling generally provides significant benefits compared to landfilling or incineration (Ismail & Hanafiah, 2019).

In Norway few studies have been conducted on the WEEE management system specifically. One study by Baxter et al., (2016) performed an LCA of the downstream treatment of three different kinds of WEEE (refrigerator, mobile phone, and LCD TV). The study shows benefits of recycling metals and highlight the importance of proper handling of smaller trace elements like refrigerator gases and recovery of precious metals as important drivers for GHG emissions (Baxter et al., 2016). Others have analyzed trends in WEEE generation, collection, and alternative routes of disposal outside the Extended Producer Responsibility (EPR) scheme adopted in Norwegian legislation for the collection system (NORSUS, 2021). No full system assessment of the WEEE management has been performed for Norway using LCA or MFA methodology.

Within the Norwegian waste regulation framework, waste policy towards recycling and reuse is clearly formulated and provided as requirements for EPR companies. The regulations set specific requirements for the recovery of eight specific WEEE product categories. These requirements are set as a percentage of the total mass of each product category to be either recovered in the form of reuse, material recovery, or energy recovery (incineration). The recovery target can vary from 75% to 85% when including energy recovery, or between 55% and 80% when excluding energy recovery of products put on market (Avfallsforskriften, 2004). These regulations are clear indications of waste policy, providing a push for increased material recovery. However, as pointed out by Bigum et al., (2012) in the case of Denmark, and Baxter et al., (2016) in the case of Norway, policy focused on optimizing aggregated and simple quantitative measures, like bulk recovery rates and recycling rates, might fail to address significant challenges within the system, or fail to incentivize recovery of small, but important materials.

In this paper we first focus on identifying the flows of consumer WEEE in Norway, addressing both the collection within the EPR schemes, and the potentially missed waste occurring outside regulated collection. We then focus on the flows of materials, and the embedded emissions of these materials. This approach is taken to assess the flow of already expended emissions within the system, and to investigate how much of our GHG emissions are being kept within the economy with the potential to substitute raw materials. Does the current system adequately recover and maintain the environmental load associated with producing consumer electronics? We address these questions by first investigating the reported flows and interviewing EPR companies, and use this data to produce a static material flow analysis model for WEEE flows in Norway. Material flow indicators are supplemented with a greenhouse gas footprint approach using LCA calculation software.

2. METHOD AND APPROACH

In this research work we gathered information from official databases, literature, and direct communication with WEEE management industry on the collection and treatment of WEEE. We follow the material flow of consumer WEEE from collection to end treatment, assessing the waste flow composition. We characterize the content of WEEE using reference products, covering iron/steel containing metals, non-ferrous metals, plastic, glass, and some precious metals. To assess the embodied emissions of different materials embedded in the waste, we use a life cycle assessment screening method to calculate the CO_2-eqvivalent (CO_2-eq) emissions associated with production, and the life cycle emissions generated by recycling, incineration, or landfilling of the respective materials. The term CO_2-eq emissions encompasses "..the amount of carbon dioxide (CO_2) emissions that would cause the same integrated radiative forcing or temperature change, over a given time horizon, as an emitted amount of a greenhouse gas (GHG) or a mixture of GHGs" (IPCC, 2018).

2.1 Waste generation and composition

The generation of WEEE is taken from the Environmental Agency's online database for producer responsibility reporting on electric and electronic waste. The data is generated by approved EPR companies, and report yearly quantity of different categories of waste collected, and the respective aggregate downstream treatments for each category (Table 1).

The amount of WEEE within each category is gathered from the database itself, however, the database has no information on the material composition of each category. To assess this, we gathered composition data from one product within each category, using this as a proxy for material composition. Table 2 shows the composition assumptions used and the reference product assumed for each category.

2.2 WEEE Flows

The total mass flows of generated e-waste are based on data from the Norwegian Environmental Agency reported by the return companies (Miljødirektoratet, 2023). In total, 142,000 tons of e-waste is collected annually (based on a four-year average of 2019-2022) in Norway, which contains all eight product categories including the large industrial equipment and large industrial cables which are not considered in this paper. The large industrial equipment is approximately 13% of the collected e-waste and the large industrial cables are approximately 12%. Excluding the in-

TABLE 1: Average share of WEEE category going to different waste treatments 2019 – 2022 according to the Norwegian Environmental Agency.

WEEE Category	Product mix	Total share (%)	Reuse	Landfilling	Energy recovery	Material recovery
1	Heating and cooling equipment	20%	0%	0%	3%	16%
2	Screens, monitors and equipment containing screens with a surface of more than 100 cm²	6%	1%	0%	0%	4%
3	Light sources	1%	0%	0%	0%	1%
4	Other large products where one of the outer dimensions is over 50 cm other large	36%	0%	2%	4%	30%
5	Other small products where the longest outer dimension is less than 50 cm	30%	0%	1%	4%	24%
6	Smaller IT and telecommunications equipment where the longest outer dimension is less than 50 cm	7%	1%	0%	1%	5%
Total		100%	2%	4%	12%	82%

TABLE 2: Material composition for reference products within each WEEE category based on Baxter et al., 2016 (a), Filimonau et al., 2021 (b), Fulvio et al., 2015 (c), Gallego-Schmid et al., 2018 (d).

Product	Steel	Aluminum	Plastic	Glass	Copper	Precious metals	Source
Refrigerator	73.9%	0.7%	24.7%	0.0%	0.60%	0.0%	(a)
Television (LCD screen)	34.7%	8.1%	44.5%	5.8%	1.7%	5.2%	(a)
CFL Lamp (Compact fluorescent lamp production)	12.2%	7.8%	23.1%	45.7%	6.8%	4.4%	(b)
Dishwasher	78.8%	0.9%	16.9%	0.0%	2.9%	0.4%	(c)
Microwave	66.4%	5.8%	7.4%	11.4%	6.1%	2.9%	(d)
Smartphone	3.0%	2.0%	59.0%	16.0%	15.0%	5.0%	(a)

dustrial waste, the amount of collected e-waste that will be considered in this study is approximately 107,232 tons per year.

The collection and first-hand treatment of WEEE we consider rather transparent, however, the recycling downstream treatment chain is significantly lacking in transparency with low reporting standards, thus leading to certain model assumptions being necessary. Assessing the treatments within the system is limited to available general treatment options in the latest Ecoinvent database (3.9.1).

We added estimates on WEEE being generated but unavailable for the collection and treatment system due to scavenging and illegal export, leading to uncontrolled disposal and highly speculative end-use of the waste. This was estimated by NORSUS (2021) at approximately 10% of EEE put on market. Based on this we estimate that 10,921 tons of WEEE is generated outside of the collection system and illegally exported with no consequent control of the material flow. We assume this category of WEEE is a mix of all categories usually collected within the EPR system.

Using waste composition analysis from residual waste generation in the households (CIVAC, 2021), we can estimate the likely collection of 1-2% of WEEE going directly to municipal solid waste incineration. We assume that this category is heavily represented by smaller WEEE like phones, lamps, electric toys, smaller electrical equipment etc. (NORSUS, 2021). We estimate at that at least 7,500 tons of WEEE are directly sent to incineration.

The total amount of consumer WEEE collected for treatment and other illegitimate End-of-Life (EOL) options are estimated at 125,653 tons per year. Table 1 in the supplementary information shows the amount of each material within each category.

2.3 Treatment assumptions

Collection and manual sorting: Manual sorting is assumed to occur at collection and sorting facilities operated by EPR-companies or affiliated companies with permission from the Environmental Agency to store different kinds of electronic and hazardous waste. The reuse fraction is assumed to be sorted manually according to Table 1. Parts of category 2 and 6 is assumed to be sorted manually due to having high composition of precious metals (phones, laptops, etc.). Based on the WEEE products accepted at UMICORE plants for recycling, we assume that most EPR companies and first order collectors of WEEE would avoid shredding these categories. However, we assume that 20% of these fractions could be shredded together with other electronic waste.

Collection and shredding: Most categories are assumed to undergo shredding to efficiently sort out metallic fractions from the WEEE (Bigum et al., 2012; Baxter et al., 2016). Most of the ABS plastic fraction is assumed to be sorted out for incineration due to hazardous compounds embedded in the plastic (Baxter et al., 2014). A mix of shredder residue from all fractions is assumed to provide the remaining fraction going to incineration. The amount going to landfill from sorting operations is assumed to mainly be glass or other inert masses contained in WEEE.

Recycling yields: Table 3 summarizes the recycling yields assumed for each material fraction making it to material recovery. We assume that all loss is landfilled from recycling plants, except for plastic recycling, where the loss fraction is sent for incineration.

The underlying assumption of waste generated and the specific treatment options within the EPR scheme is based on the assertion that the treatment statistics show fractions sorted for specific treatments and does not include any estimate of recycling loss. Effectively not reporting recycling rates, but the level of sorting for recycling, excluding actual yields. Recycling loss is introduced in the model from the review of the scientific literature referenced in table 3. Communication with industry failed to get a definitive clarification on this assumption (personal communication, 2023), the Environmental Agency is also unclear regarding the nature of the reported numbers (personal communication, 2022).

Incineration and landfilling: These treatments are based on Ecoinvent life cycle inventory models for average treatments and are referenced in section 2.5.

2.4 Material Flow Analysis model

MFA has been widely used within the research field of waste management in order to represent complex systems within an analytic framework, providing a systematic and quantitative overview of any given system over space and time (Allesch & Brunner, 2015; Hendriks et al., 2000). Following the data gathered in section 2.1 to 2.3, we construct a simple static system, representing an average flow of WEEE being generated in Norway, and sent to specific treatments. The system is mass balanced and provides a full account of mass and substance flows within the system boundary.

Figure 1 illustrates the MFA system definition, showing inputs to the system (I) in terms of WEEE collected/delivered to waste facilities of different kinds, and electronics found in the residual bin of households. The system is mass balanced by subtracting the flow of outputs (O) from the inflows. The outputs are further specified into A) Recycling yield, B) Incineration, C) Landfilling, and D) Illegal export.

The recycling rate of the total WEEE system is then calculated by dividing the recycling yield (A), over the sum of all outputs of the system:

$$Recycling\ rate = \frac{A}{O}$$

The transfer coefficients of each process (collection and sorting, material recovery, recycling, incineration etc.) is based on section 2.3, while the input to the system is based on section 2.2. Material flowing through the system are based on section 2.1.

2.5 Embodied category and EOL CO_2-eq emissions

Categories of embodied emissions are calculated based on the material composition of the WEEE reference products and the respective Ecoinvent Life-Cycle-Inventory (LCI) data. Further information on Ecoinvent inventories used in the CO_2-eq footprint calculations is available in the supplementary information. The impacts of production and treatment was scaled up by the respective demand for each material within each category and treatment option and multiplied with environmental impact indicators taken from the IPCC2021 GWP100 method made available by ecoinvent. This quantifies the embodied impacts of production in terms of CO_2-eq emissions over the time-horizon of 100 years (Forster et al., 2021), and include the CO_2-eq emissions due to EOL recycling, incineration or landfilling of the composite materials. The calculations were performed using the Brightway open-source framework for LCA calculations in python (C. Mutel, 2017). This approach does not constitute a full life cycle assessment, but an LCA screening approach similar to other footprint calculation approaches within the environmental assessment literature (Hung et al., 2020; Kaufman et al., 2010; Mora-Sojo et al., 2022; Song et al., 2017).

TABLE 3: Recycling yield based on literature assessing the mass balance of recycling activities, yield indicate the share of material output from a recycling process that can substitute a primary material.

Material	Recycling Yield	Source
Steel	87%	(NORSUS, 2022)
Aluminium	75%	(Bigum et al., 2012)
Plastic mix	70%	(Rigamonti et al., 2020)
Glass	90%	(Haupt et al., 2018)
Copper	60%	(Pfaff et al., 2018)
Palladium	90%	(Bigum et al., 2012)
Gold	90%	(Bigum et al., 2012)
Silver	90%	(Bigum et al., 2012)
Nickel	90%	(Bigum et al., 2012)

3. RESULTS

The following section show the results of the identified WEEE flows and the embedded materials, and the embedded CO_2-eq emissions of materials and the added emissions of electronic waste EOL treatment. The recycling rate of materials is estimated, and consequently the reintroduction of embedded emissions is also estimated and illustrated with the use of Sankey diagrams.

3.1 Material flows

Combining the data on WEEE collection and treatment with the literature data on the material composition of WEEE products, in Figure 2 we can observe the distribution of materials between the main categories included in our system.

Categories 1, 4 and 5 are dominated by steel, with some plastic mix, glass and copper making up the rest of the main mass. Categories 2, 3, and 6 are mainly dominated by plastic mixes. We also observe significant amounts of steel and plastics not entering the EPR-scheme via residuals and illegal exports.

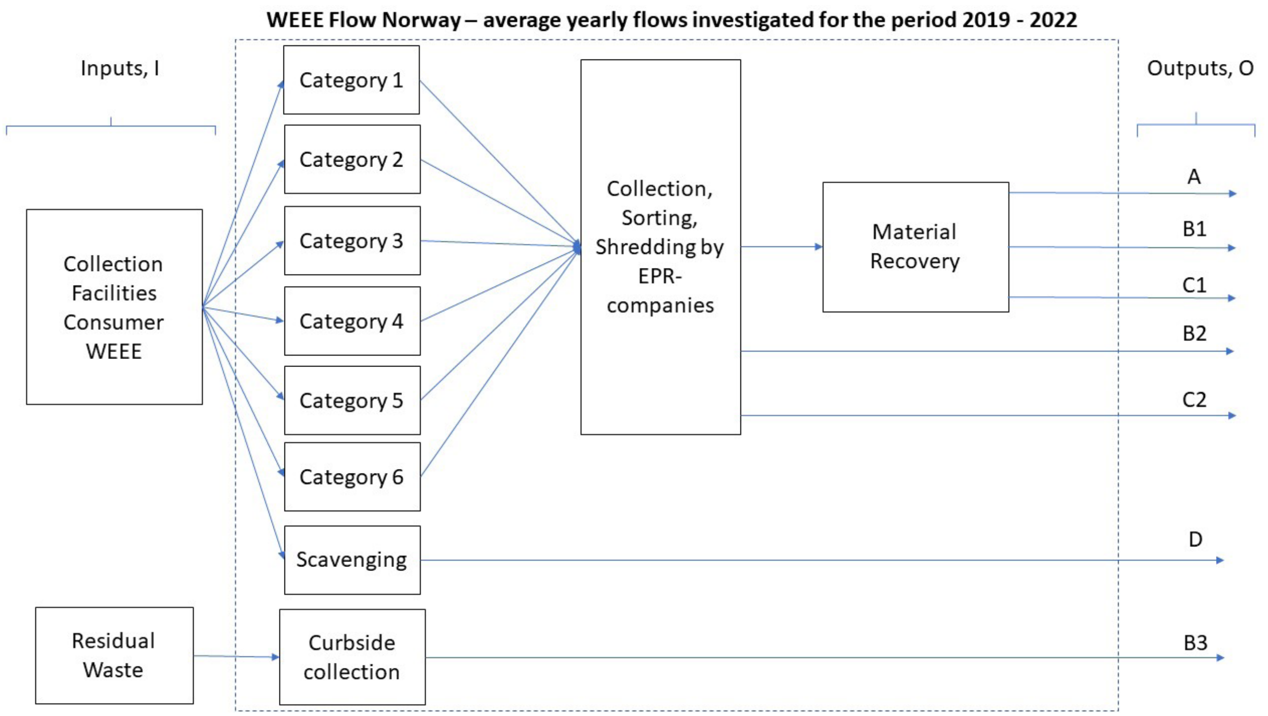

FIGURE 1: Material Flow Analysis system description for calculating the material flows and the recycling rate of the WEEE system. The flows labeled A) for recycling, B) for incineration, C) for landfilling and D) for illegal exports.

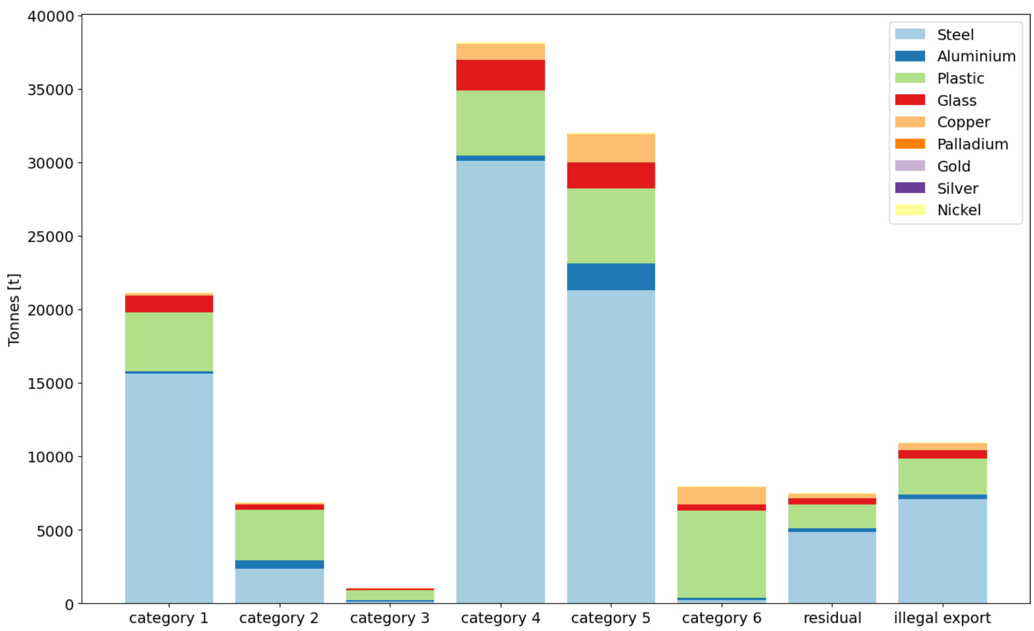

FIGURE 2: Bar graph showing the total yearly mass per WEEE category and its subsequent share of embodied materials.

Figure 3 shows the mass flow of the WEEE management system in Norway, including consumer electronics discarded as part of household residual waste, and electronics scavenged from collection facilities. Based on the assumption that the official statistics are not including losses occurring at the recycling plants, we can observe that 68% of category 1-6 is recycled. 2% is reused. If we include the amounts generated outside the EPR scheme, 58% of generated WEEE is recycled into materials.

Here we denote the recycling and reuse of materials into valuable goods "Circular Material Use", meaning that the system provides a flow of materials that can potentially be used, and contribute to a circular economy. We denote the end treatments that destroy, store, or in other ways lose materials from the downstream treatment chains, making them inaccessible for further valorization, as "Linear Material Loss". This highlights that the loss of material in terms of export, the incineration of materials, and the landfilling of ma-

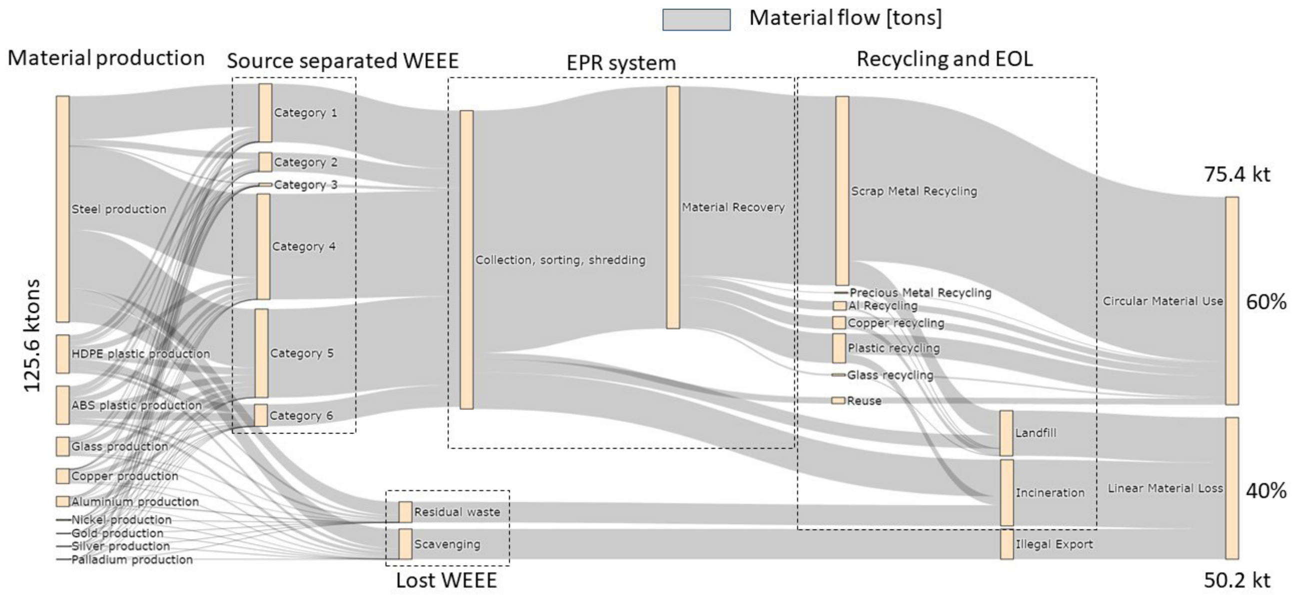

FIGURE 3: Mass flow diagram of material production, collection within and outside the EPR scheme, and materials going to Circular Material Use, or Linear Material Loss based on collection raters and recycling yields.

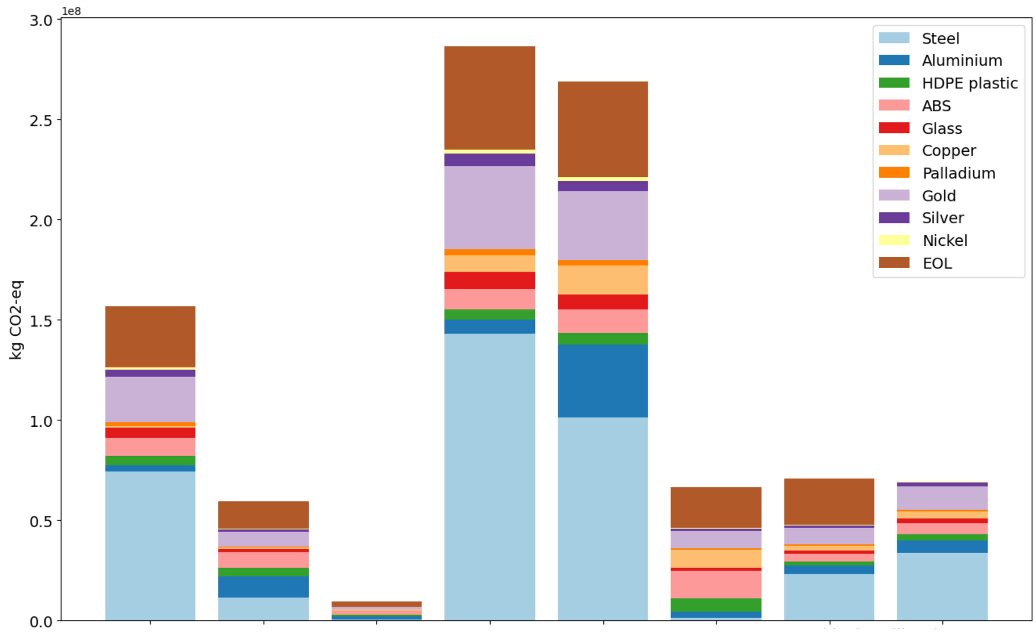

FIGURE 4: Kg CO_2-eq emissions embedded in material production within each waste category, and End-of-Life (EOL) treatment emissions due to incineration, landfilling or recycling processes.

terials, is directly linked to a linear system model, where we extract materials and dispose them with no significant value added in the process (Romero-Hernández & Romero, 2018).

3.2 Embedded carbon footprint and EOL GHG emissions

Figure 4 include the embodied CO_2-eq emissions of the materials within each category and the added aggregate EOL emission. Steel is dominant for the larger categories, however, we see the significant contribution of gold, silver, and nickel, as well as aluminium for smaller fractions. Materials that occupy a tiny fraction in terms of the total mass of a product, still contribute a significant share of the emissions associated with the production of the product. We can also observe that the EOL emissions are comparatively small to the emissions of the embodied materials within the discarded electronic products.

Figure 5 further disaggregates the emissions from EOL into specific treatments. In terms of CO_2-eq emissions from downstream waste treatment, we see that incineration of plastic and shredder debris contribute significantly to the emissions within each WEEE category.

We see that sanitary landfilling of waste have low emissions of CO_2-eq, emissions are mainly from operating vehicles and maintaining landfill operations and infrastructure. We can observe that the treatment of precious metals dominates much of the emissions from EOL recycling treatments. Following a contribution analysis of the results for gold recycling, we can observe that the main emission drivers are high energy use. This being the case for all precious metal recycling processes. Energy use, sorting and pressing, and quicklime production are the main contributors to CO_2-eq emissions within the steel recycling process. In terms of incineration, the process-specific emissions are the most important component, especially since the main waste type sent for incineration is plastic, with high carbon content. (Fiore et al., 2019; Stubbings et al., 2021)

We add the embodied CO_2-eq emissions of materials in WEEE to the material flows shown in Figure 3 and add the EOL CO_2-eq emissions (orange flow) from each EOL treatment, showing the flow of emissions through the WEEE treatment system in Figure 4.

This helps us to illustrate the impacts associated with both production and EOL treatment and the large amounts of embodied emissions being directed back to the economy, or to Linear Material Loss. Figure 6 is based on the official information provided by the Norwegian Environmental Agency, assuming valorization of all material content. In Figure 7, instead of assuming precious metals (PM) going to Circular Material Use, we assume that 50% of these metals are downcycled into steel, other metallic fractions, incinerated or landfilled, which in either case means that the PM is not actually reintroduced in a functional way to the circular economy, they are "functionally lost" (Ortego et al. 2018).

We see from Figure 7 a 50/50 split of total emissions between Linear Material Loss and Circular Material Use. Redirecting a tiny fraction of specific materials contribute to a 9% shift towards loss of materials. Indicating that a significant amounts of expended CO_2-eq emissions are directly lost and permanently unavailable for reintroduction to the circular economy, requiring continued extraction of primary raw material, and consequently a substantial potential for increasing GHG emissions if we are to sustain or grow our consumption of consumer electronics.

4. DISCUSSION

This material flow and life cycle emissions screening approach shows that the embodied emissions of EEE products is mainly coming from metals, with significant shares from precious metals. The CO_2-eq emissions from production of materials embedded in the WEEE categories is estimated at approximately 320 kg CO_2-eq / household in Norway. These results where comparable to previous studies on GHG emissions from EEE in Norwegian households (Hertwich & Roux, 2011). Emissions from EOL is reported by Hertwich & Roux (2011) at 43 kg CO_2-eq per household, where our results indicate approximately 75 kg CO_2-eq per household. Our assessment likely overestimates the amount of precious metals going to recycling, which is the main contributing processes to EOL emissions within our system. It is quite likely that larger shares of precious metal containing category waste is downcycled.

The material composition of each category of WEEE is a significant source of uncertainty in this analysis, however, the data considered does give some useful indication of the potential composition and embodied emissions due to primary production. The lack of transparency into downstream treatment chains within the WEEE management sector is also a significant challenge for assessing both the resource recovery efficiency of different recycling paths, and the environmental impacts generated for these treatments. These issues have been highlighted by other

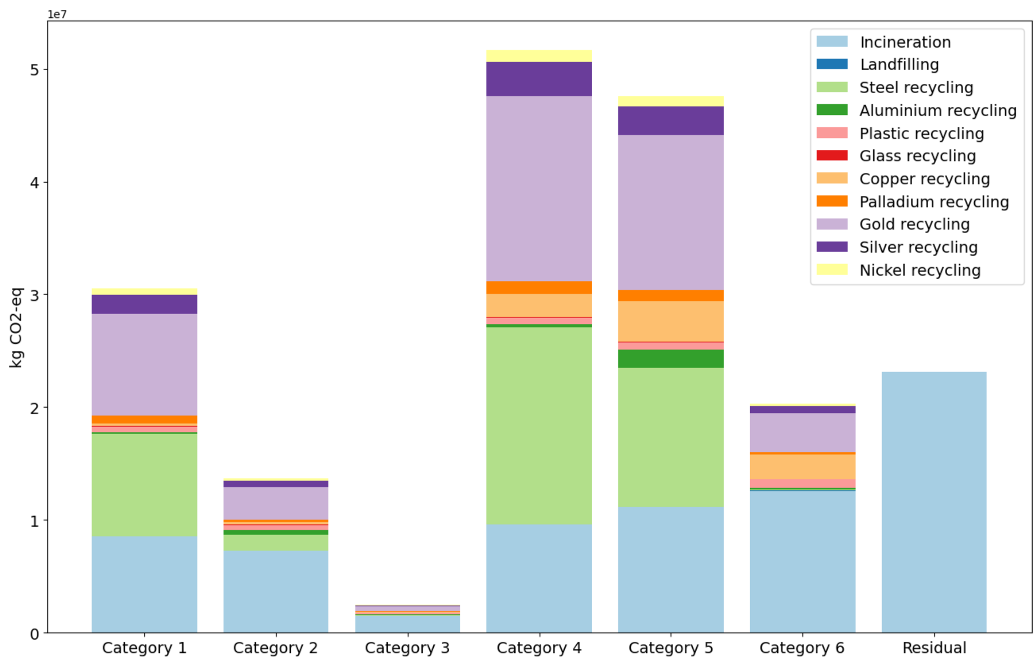

FIGURE 5: Breakdown of CO_2-eq emissions for each End-of-Life treatment, per waste category.

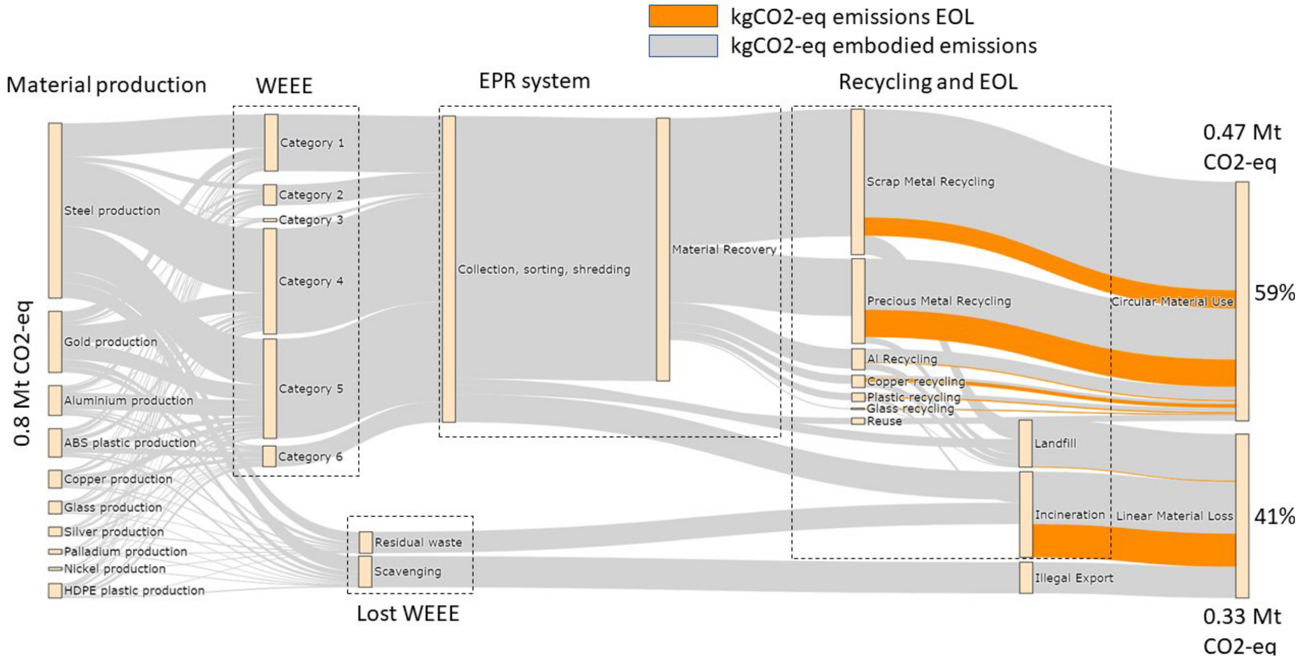

FIGURE 6: Emission flow layer showing embodied CO_2-eq emissions from material production, and emissions occuring at EoL treatments, baseline assumption is recycling of precious metals.

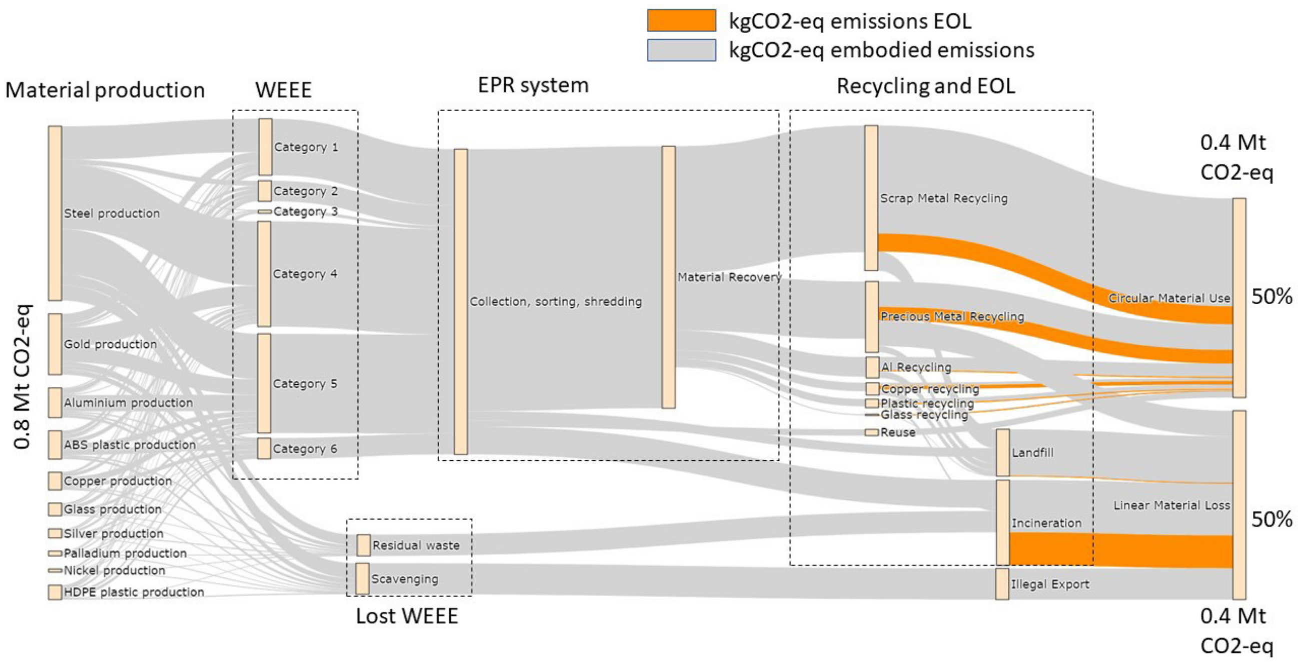

FIGURE 7: Emission flow layer showing embodied CO_2-eq emissions from material production, and emissions occuring at EoL treatments, Scenario with 50% less recycling of precious metals.

studies assessing the environmental impact of WEEE in specific countries or Europe in general (Di Maio et al., 2015; De Meester et al., 2019; Horta Arduin et al., 2019). A recent comprehensive comparative LCA study on gold recycling from WEEE, highlights the variability of human health impacts and energy requirements for the most conventional gold recycling techniques (He et al., 2023). Better resolution on the specific treatments used for recycling is of crucial importance when indicating environmental performance. Amending existing gold refining techniques (Fernando et al., 2019) and optimizing gold refinery operations (Mahyapour & Mohammadnejad, 2022) has a significant part to play when it comes to lowering the environmental burden of continued gold use.

Using waste treatment life cycle inventories available in Ecoinvent, we have added an emissions layer to our material flows, showing that there exists a significant advantage to recycling materials compared to primary production of materials. However, the model approach is simplistic, and as pointed out by Fortier et al., (2022) in the 2021 annual review of critical minerals by the United States Geological Survey; existing LCI data is highly variable and based on information on few mining sites. The LCI would not be able to account for capacity issues due to increased demand, or spatially explicit emissions (Fortier et al., 2022). Only focusing on the climate change indicator in our case, does reduce some of the uncertainty due to GHG emissions representing a global impact. In this study we do not include the environmental impacts of dismantling and shredding operations, this should be included when assessing the direct impacts on human health from WEEE treatment (Lan et al., 2023). Impacts does naturally also occur during collection, dismantling, shredding, and transport operations, for a full LCA study of WEEE, these aspects should be included in the system.

This initial study does not look carefully into the issue of downcycling and substitution rates of the recycled materials, which has been highlighted by multiple previous studies (Rigamonti et al., 2020; Vadenbo et al., 2017; Viau et al., 2020). It is unclear to what extent the output of the recycling processes could be used directly in the production of complex electronic equipment. To assess these issues, plant level studies would be needed in order to model these complex material processes.

In the results we present a Sankey flow chart of embedded CO_2-eq emissions and include the emissions being produced in the treatment of WEEE. This helps us see where the treatment emissions are occurring, what categories of WEEE have the largest footprints, and it also indicates how well the system can avoid spending emissions (in terms of incineration and landfilling) and wasting forgone emissions (embodied emissions from production) in a linear “one-time-use” system. The idea of Linear Material Loss indicate that the emissions expended while producing a certain product, permanently goes to waste. This is contrasted with the Circular Material Use, indicating a renewed life for past emissions, albeit at the cost of some recycling emissions and recycling loss. It is clear from the result that if the current system fails to incentives the delivery of critical or precious materials back into the production system for consumer EEE, we are in dire need of better policy, or radically different use-patterns for electronics.

Adding the layer of emissions to the material flow analysis helps us illustrate that materials that represent a very small fraction of the total weight of embedded materials, has a disproportional level of environmental impact. Seeing the problem in this way, exemplifies the necessity for policies that target these small but critical materials.

This concept could be extended further, looking at more aggregated impacts in terms of human health, ecosystem damage, and resource depletion. For instance using the ReCiPe2016 method for life cycle impact assessment (Hauschild & Huijbregts, 2015; Huijbregts et al., 2017; Woods et al., 2018). Making it possible to assess the impact of multiple environmental emissions, not just the once relevant for climate change. However, in order to capture these impacts to a higher degree of certainty one would need to address spatio-temporal characteristics of production and treatments of WEEE categories (Beloin-Saint-Pierre et al., 2020; C. L. Mutel & Hellweg, 2009; Pfister et al., 2020). This could be done by a combining dynamic-Material Flow Analysis with a regionalized Life Cycle Assessment, making it possible to look at multiple policy options over time, and the potential environmental effects with greater spatial resolution (Barkhausen et al., 2023). Further development of these environmental systems analysis methods is important for developing more concrete policies for resource management, especially for the advanced material demand from technology and energy sectors.

Another aspect of the waste management system that is not assessed in this paper, is the importance of consumer sorting behavior and engagement with the recycling system (Flygansvær et al., 2021). One aspect is the lifetime of the EEE products (Bridgens et al., 2019), another is the willingness to engage with the recycling system by citizens and businesses (Jabbour et al., 2023). The referenced papers highlight the importance of information on the value and benefit of recycling, which is hard to adopt in a regulatory framework. However, national policies should reflect that convenience of recycling (Wang et al., 2019) and information on recycling benefits (Jabbour et al., 2023) could promote better practice. Having transparent information on the WEEE treatment system becomes an important part of any information campaign.

Multiple review studies have highlighted the importance of improved and enforceable waste legislation for WEEE (Constantinescu et al., 2022; Patil & Ramakrishna, 2020; Shahabuddin et al., 2023). In the case of Norway, the type of strict legislation suggested by Patil and Ramakrishna (2020) does exist, but it is clearly not enough to sufficiently incentivize the recycling of smaller and more complex materials (i.e., gold, palladium, nickel etc.). Claims of recycling rates and EPR schemes successful integration of circular economy principles should be scrutinized and assessed critically. Researching the downstream treatments of the different WEEE categories proved difficult, especially due to limited information and proof regarding countries actual recovery rates. It would be advisable for policymakers to engage more directly with the existing EPR structure, and tailor regulations towards value and emission-based recycling goals, in contrast to bulk recycling requirements which is the current practice.

5. CONCLUSIONS

We show that the effectiveness of the Norwegian WEEE management system in terms of resource management is highly dependent on how well the precious metals are recovered at the EOL. There is a significant lack of transparency within the WEEE sector, and there are few indications of an effective valorization of precious and/or critical materials embedded in electronic consumer products. We argue that resource management should include embedded emissions of production, to effectively include the high environmental costs of producing as well as managing our waste. If we fail to reintroduce materials into the economy within the same product niche they initially where part of, we are at a significant risk of not curbing future demand for materials that will require substantial amounts of energy and resources to acquire. Its apparent that current policy enacted in Norway is not sufficient to safeguard the recovery of critical raw materials from WEEE. The first step to improve and incentivize for increased Circular Material Use would be to set recycling goals for specific material fractions (precious metals, aluminium, copper etc.), and introducing stricter reporting requirements, providing clear information on both the real recycling yield, and the limitations of current systems and technologies.

ACKNOWLEDGEMENTS

We would like to thank all companies and organizations that have answered our requests for information regarding the logistics and EOL treatment of WEEE.

This work is part of the CircWtE project co-funded by industry and public partners and the Research Council of Norway under the SIRKULÆRØKONOMI program (CircWtE - 319795).

REFERENCES

Allesch, A., & Brunner, P. H. (2015). Material Flow Analysis as a Decision Support Tool for Waste Management: A Literature Review: MFA for Waste Management: A Literature Review. Journal of Industrial Ecology, 19(5), 753–764. https://doi.org/10.1111/jiec.12354

Babbitt, C. W., Althaf, S., Cruz Rios, F., Bilec, M. M., & Graedel, T. E. (2021). The role of design in circular economy solutions for critical materials. One Earth, 4(3), 353–362. https://doi.org/10.1016/j.oneear.2021.02.014

Babbitt, C. W., Madaka, H., Althaf, S., Kasulaitis, B., & Ryen, E. G. (2020). Disassembly-based bill of materials data for consumer electronic products. Scientific Data, 7(1), 251. https://doi.org/10.1038/s41597-020-0573-9

Barkhausen, R., Rostek, L., Miao, Z. C., & Zeller, V. (2023). Combinations of material flow analysis and life cycle assessment and their applicability to assess circular economy requirements in EU product regulations. A systematic literature review. Journal of Cleaner Production, 407, 137017. https://doi.org/10.1016/j.jclepro.2023.137017

Baxter, J., Lyng, K.-A., Askham, C., & Hanssen, O. J. (2016). High-quality collection and disposal of WEEE: Environmental impacts and resultant issues. Waste Management, 57, 17–26. https://doi.org/10.1016/j.wasman.2016.02.005

Baxter, J., Wahlstrom, M., Castell-Rüdenhausen, M. Z., Fråne, A., Stare, M., Løkke, S., & Pizzol, M. (2014). Plastic value chains: Case: WEEE (Waste Electric and electronic equipment) in the Nordic region.

Beloin-Saint-Pierre, D., Albers, A., Hélias, A., Tiruta-Barna, L., Fantke, P., Levasseur, A., Benetto, E., Benoist, A., & Collet, P. (2020). Addressing temporal considerations in life cycle assessment. Science of The Total Environment, 743, 140700. https://doi.org/10.1016/j.scitotenv.2020.140700

Bigum, M., Brogaard, L., & Christensen, T. H. (2012). Metal recovery from high-grade WEEE: A life cycle assessment. Journal of Hazardous Materials, 207–208, 8–14. https://doi.org/10.1016/j.jhazmat.2011.10.001

Bridgens, B., Hobson, K., Lilley, D., Lee, J., Scott, J. L., & Wilson, G. T. (2019). Closing the Loop on E-waste: A Multidisciplinary Perspective. Journal of Industrial Ecology, 23(1), 169–181. https://doi.org/10.1111/jiec.12645

CIVAC, (2021), "Hvilke ressurser finnes i restavfallet", URL: https://civac.no/dokumentasjon/hvilke-ressurser-finnes-i-restavfallet/ (last visited, 14.12.2023).

Constantinescu, A., Platon, V., Surugiu, M., Frone, S., Antonescu, D., & Mazilescu, R. (2022). The Influence of Eco-Investment on E-Waste Recycling-Evidence From EU Countries. Frontiers in Environmental Science, 10. https://www.frontiersin.org/articles/10.3389/fenvs.2022.928955

De Meester, S., Nachtergaele, P., Debaveye, S., Vos, P., & Dewulf, J. (2019). Using material flow analysis and life cycle assessment in decision support: A case study on WEEE valorization in Belgium. Resources, Conservation and Recycling, 142, 1–9. https://doi.org/10.1016/j.resconrec.2018.10.015

European Commission. Joint Research Centre. (2021). Material composition trends in vehicles: Critical raw materials and other relevant metals : preparing a dataset on secondary raw materials for the raw materials information system. Publications Office. https://data.europa.eu/doi/10.2760/351825

Fernando, W. A. M., Ilankoon, I. M. S. K., Krishnan, S. G., & Chokshi, V. (2019). The effects of packing shape and structure on liquid distribution of heap leaching systems: Addition of PCBs as non-ore particles. Hydrometallurgy, 187, 149–157. https://doi.org/10.1016/j.hydromet.2019.05.010

Fiore, S., Ibanescu, D., Teodosiu, C., & Ronco, A. (2019). Improving waste electric and electronic equipment management at full-scale by using material flow analysis and life cycle assessment. Science of The Total Environment, 659, 928–939. https://doi.org/10.1016/j.scitotenv.2018.12.417

Flygansvær, B., Samuelsen, A. G., & Støyle, R. V. (2021). The power of nudging: How adaptations in reverse logistics systems can improve end-consumer recycling behavior. International Journal of Physical Distribution & Logistics Management, 51(9), 958–977. https://doi.org/10.1108/IJPDLM-12-2020-0389

Forster, P., T. Storelvmo, K. Armour, W. Collins, J.-L. Dufresne, D. Frame, D.J. Lunt, T. Mauritsen, M.D. Palmer, M. Watanabe, M. Wild, and H. Zhang, (2021): The Earth's Energy Budget, Climate Feedbacks, and Climate Sensitivity. In Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change [Masson-Delmotte, V., P. Zhai, A. Pirani, S.L. Connors, C. Péan, S. Berger, N. Caud, Y. Chen, L. Goldfarb, M.I. Gomis, M. Huang, K. Leitzell, E. Lonnoy, J.B.R. Matthews, T.K. Maycock, T. Waterfield, O. Yelekçi, R. Yu, and B. Zhou (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 923–1054, doi:10.1017/9781009157896.009.

Fortier, S. M., Nassar, N. T., Graham, G. E., Hammarstrom, J. M., Day, W. C., & Mauk, J. L. (n.d.). USGS critical minerals review.

Graedel, T. E., Reck, B. K., & Miatto, A. (2022). Alloy information helps prioritize material criticality lists. Nature Communications, 13(1), 150. https://doi.org/10.1038/s41467-021-27829-w

Hauschild, M. Z., & Huijbregts, M. A. J. (2015). Introducing Life Cycle Impact Assessment. In M. Z. Hauschild & M. A. J. Huijbregts (Eds.), Life Cycle Impact Assessment (pp. 1–16). Springer Netherlands. https://doi.org/10.1007/978-94-017-9744-3_1

He, Y., Hosseinzadeh-Bandbafha, H., Kiehbadroudinezhad, M., Peng, W., Tabatabaei, M., & Aghbashlo, M. (2023). Environmental footprint analysis of gold recycling from electronic waste: A comparative life cycle analysis. Journal of Cleaner Production, 432, 139675. https://doi.org/10.1016/j.jclepro.2023.139675

Hendriks, C., Obernosterer, R., Müller, D., Kytzia, S., Baccini, P., & Brunner, P. H. (2000). Material Flow Analysis: A tool to support environmental policy decision making. Case-studies on the city of Vienna and the Swiss lowlands. Local Environment, 5(3), 311–328. https://doi.org/10.1080/13549830050134257

Hertwich, E. G., & Roux, C. (2011). Greenhouse Gas Emissions from the Consumption of Electric and Electronic Equipment by Norwegian Households. Environmental Science & Technology, 45(19), 8190–8196. https://doi.org/10.1021/es201459c

Horta Arduin, R., Grimaud, G., Martínez Leal, J., Pompidou, S., Charbuillet, C., Laratte, B., Alix, T., & Perry, N. (2019). Influence of scope definition in recycling rate calculation for European e-waste extended producer responsibility. Waste Management, 84, 256–268. https://doi.org/10.1016/j.wasman.2018.12.002

Huijbregts, M. A. J., Steinmann, Z. J. N., Elshout, P. M. F., Stam, G., Verones, F., Vieira, M., Zijp, M., Hollander, A., & van Zelm, R. (2017). ReCiPe2016: A harmonised life cycle impact assessment method at midpoint and endpoint level. The International Journal of Life Cycle Assessment, 22(2), 138–147. https://doi.org/10.1007/s11367-016-1246-y

Hung, C. R., Ellingsen, L. A.-W., & Majeau-Bettez, G. (2020). LiSET: A Framework for Early-Stage Life Cycle Screening of Emerging Technologies. Journal of Industrial Ecology, 24(1), 26–37. https://doi.org/10.1111/jiec.12807

IPCC, 2018: Annex I: Glossary [Matthews, J.B.R. (ed.)]. In: Global Warming of 1.5°C. An IPCC Special Report on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change, sustainable development, and efforts to eradicate poverty [Masson-Delmotte, V., P. Zhai, H.-O. Pörtner, D. Roberts, J. Skea, P.R. Shukla, A. Pirani, W. Moufouma-Okia, C. Péan, R. Pidcock, S. Connors, J.B.R. Matthews, Y. Chen, X. Zhou, M.I. Gomis, E. Lonnoy, T. Maycock, M. Tignor, and T. Waterfield (eds.)]. Cambridge University Press, Cambridge, UK and New York, NY, USA, pp. 541-562. https://doi.org/10.1017/9781009157940.008

Ismail, H., & Hanafiah, M. M. (2019). An overview of LCA application in WEEE management: Current practices, progress and challenges. Journal of Cleaner Production, 232, 79–93. https://doi.org/10.1016/j.jclepro.2019.05.329

Jabbour, C. J. C., Colasante, A., D'Adamo, I., Rosa, P., & Sassanelli, C. (2023). Comprehending e-waste limited collection and recycling issues in Europe: A comparison of causes. Journal of Cleaner Production, 427, 139257. https://doi.org/10.1016/j.jclepro.2023.139257

Kaufman, S. M., Krishnan, N., & Themelis, N. J. (2010). A Screening Life Cycle Metric to Benchmark the Environmental Sustainability of Waste Management Systems. Environmental Science & Technology, 44(15), 5949–5955. https://doi.org/10.1021/es100505u

Lan, Y., Liu, Y., Cai, Y., Du, Q., Zhu, H., Tu, H., Xue, J., & Cheng, Z. (2023). Eight novel brominated flame retardants in indoor and outdoor dust samples from the E-waste recycling industrial park: Implications for human exposure. Environmental Research, 238, 117172. https://doi.org/10.1016/j.envres.2023.117172

Mahyapour, H., & Mohammadnejad, S. (2022). Optimization of the operating parameters in gold electro-refining. Minerals Engineering, 186, 107738. https://doi.org/10.1016/j.mineng.2022.107738

Messmann, L., Helbig, C., Thorenz, A., & Tuma, A. (2019). Economic and environmental benefits of recovery networks for WEEE in Europe. Journal of Cleaner Production, 222, 655–668. https://doi.org/10.1016/j.jclepro.2019.02.244

Mora-Sojo, M. C., Krych, K., & Pettersen, J. B. (2022). Evaluating the Current Norwegian Clothing System and a Circular Alternative (SSRN Scholarly Paper 4308369). https://doi.org/10.2139/ssrn.4308369

Mutel, C. (2017). Brightway: An open source framework for Life Cycle Assessment. The Journal of Open Source Software, 2(12), 236. https://doi.org/10.21105/joss.00236

Mutel, C. L., & Hellweg, S. (2009). Regionalized Life Cycle Assessment: Computational Methodology and Application to Inventory Databases. Environmental Science & Technology, 43(15), 5797–5803. https://doi.org/10.1021/es803002j

Norwegian Institute for Sustainability Research (NORSUS), 2021, Collection of Electronic Waste. Authors: Baxter, J., Prestrud, K., Egebæk, K. & Mckinnin, D. Commissioned by Norsk Industri

Oguchi, M., Sakanakura, H., & Terazono, A. (2013). Toxic metals in WEEE: Characterization and substance flow analysis in waste treatment processes. Science of The Total Environment, 463–464, 1124–1132. https://doi.org/10.1016/j.scitotenv.2012.07.078

Parajuly, K., Habib, K., & Liu, G. (2017). Waste electrical and electronic equipment (WEEE) in Denmark: Flows, quantities and management. Resources, Conservation and Recycling, 123, 85–92. https://doi.org/10.1016/j.resconrec.2016.08.004

Patil, R. A., & Ramakrishna, S. (2020). A comprehensive analysis of e-waste legislation worldwide. Environmental Science and Pollution Research, 27(13), 14412–14431. https://doi.org/10.1007/s11356-020-07992-1

Peng, P., & Shehabi, A. (2022). Regional economic potential for recycling consumer waste electronics in the United States. Nature Sustainability, 6(1), 93–102. https://doi.org/10.1038/s41893-022-00983-9

Pfister, S., Oberschelp, C., & Sonderegger, T. (2020). Regionalized LCA in practice: The need for a universal shapefile to match LCI and LCIA. The International Journal of Life Cycle Assessment, 25(10), 1867–1871. https://doi.org/10.1007/s11367-020-01816-7

Rigamonti, L., Taelman, S. E., Huysveld, S., Sfez, S., Ragaert, K., & Dewulf, J. (2020). A step forward in quantifying the substitutability of secondary materials in waste management life cycle assessment studies. Waste Management, 114, 331–340. https://doi.org/10.1016/j.wasman.2020.07.015

Rodriguez-Garcia, G. (n.d.). Chapter 7—Life Cycle Assessment in WEEE Recycling.

Romero-Hernández, O., & Romero, S. (2018). Maximizing the value of waste: From waste management to the circular economy. Thunderbird International Business Review, 60(5), 757–764. https://doi.org/10.1002/tie.21968

Shahabuddin, M., Uddin, M. N., Chowdhury, J. I., Ahmed, S. F., Uddin, M. N., Mofijur, M., & Uddin, M. A. (2023). A review of the recent development, challenges, and opportunities of electronic waste (e-waste). International Journal of Environmental Science and Technology, 20(4), 4513–4520. https://doi.org/10.1007/s13762-022-04274-w

Song, R., Keller, A. A., & Suh, S. (2017). Rapid Life-Cycle Impact Screening Using Artificial Neural Networks. Environmental Science & Technology, 51(18), 10777–10785. https://doi.org/10.1021/acs.est.7b02862

Statistics Norway (SSB), Waste accounts - table 10513: https://www.ssb.no/en/statbank/table/10513 (last accessed 14.12.2023).

Stubbings, W. A., Abdallah, M. A.-E., Misiuta, K., Onwuamaegbu, U., Holland, J., Smith, L., Parkinson, C., McKinlay, R., & Harrad, S. (2021). Assessment of brominated flame retardants in a small mixed waste electronic and electrical equipment (WEEE) plastic recycling stream in the UK. Science of The Total Environment, 780, 146543. https://doi.org/10.1016/j.scitotenv.2021.146543

Ueberschaar, M., Geiping, J., Zamzow, M., Flamme, S., & Rotter, V. S. (2017). Assessment of element-specific recycling efficiency in WEEE pre-processing. Resources, Conservation and Recycling, 124, 25–41. https://doi.org/10.1016/j.resconrec.2017.04.006

Vadenbo, C., Hellweg, S., & Astrup, T. F. (2017). Let's Be Clear(er) about Substitution: A Reporting Framework to Account for Product Displacement in Life Cycle Assessment: A Framework to Account for Substitution in LCA. Journal of Industrial Ecology, 21(5), 1078–1089. https://doi.org/10.1111/jiec.12519

Viau, S., Majeau-Bettez, G., Spreutels, L., Legros, R., Margni, M., & Samson, R. (2020). Substitution modelling in life cycle assessment of municipal solid waste management. Waste Management, 102, 795–803. https://doi.org/10.1016/j.wasman.2019.11.042

Wang, B., Ren, C., Dong, X., Zhang, B., & Wang, Z. (2019). Determinants shaping willingness towards on-line recycling behaviour: An empirical study of household e-waste recycling in China. Resources, Conservation and Recycling, 143, 218–225. https://doi.org/10.1016/j.resconrec.2019.01.005

Woods, J. S., Damiani, M., Fantke, P., Henderson, A. D., Johnston, J. M., Bare, J., Sala, S., Maia de Souza, D., Pfister, S., Posthuma, L., Rosenbaum, R. K., & Verones, F. (2018). Ecosystem quality in LCIA: Status quo, harmonization, and suggestions for the way forward. The International Journal of Life Cycle Assessment, 23(10), 1995–2006. https://doi.org/10.1007/s11367-017-1422-8