# De la gestion des déchets à l'approvisionnement de matières secondaires : développement d'indicateurs pour la gestion des DEEE - focus sur la filière française

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# From waste management to supplier of secondary raw materials: development of indicators to support WEEE chain management - focus on the French system

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T H È S E


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**Summary |**

Waste Electrical and Electronic Equipment (WEEE) is among the key urban mining stream due to its composition and rising volume. In the European Union (EU), WEEE management is regulated by the WEEE Directive (Directive 2012/19/EU). Currently, in the EU, e-waste chain performance is mainly assessed by technical indicators that aim to ensure system compliance with collection and recovery targets set by the WEEE Directive.

The goal of this thesis is to establish a robust set of indicators covering multidimensional aspects related to the collection and treatment of WEEE. These indicators intend to improve the visibility on the progress of the WEEE official schemes in a circular economy. We propose indicators to evaluate beyond the current overall weight-based approach, considering the French WEEE schemes framework. Different environmental, economic and criticality priorities related to the recovery of raw materials from e-waste are assessed.

The existing indicators and targets are measured in an overall weight-based approach, without considering treatment losses, and excluding flows not collected by the official schemes but captured by complementary flows. Moreover, the legislation does not set incentives to recover any specific materials, in particular those contained in low or trace amounts, among which are many Critical Raw Materials (CRMs).

The WEEE Directive and French regulation target higher collection and treatment rates in the coming years. Therefore, to ensure an increase in quantity and quality of e-waste collected, reused and recycled, it is necessary to improve our knowledge and control of the WEEE flows. Thus, it is important to enlarge the scope of the indicators to better monitor treatment losses and flows not collected by the official schemes but captured by complementary flows.

The complexity of e-waste implies a multidisciplinary approach to ensure a holistic comprehension of the challenges associated with its management. The regulatory framework of e-waste in Europe, and more precisely in France, is the guideline for the discussions and propositions in this thesis.

Quantifying e-waste flows is crucial for calculating the performance of the chain. Thus, Material Flow Analysis (MFA) is the main approach used to assess the technical performance of WEEE chains. MFA is applied in this study to track the mass balance of flows, products, components, materials, and substances in e-waste. From the environmental viewpoint, Life Cycle Assessment (LCA) is the methodology used to assess the environmental impacts of e-waste, followed by a discussion on the effectiveness and difficulties of using the results to support decision making in WEEE management.

Seeing the importance of the economic balance of the WEEE schemes, an assessment of the WEEE chain potential from the perspective of the market value of the materials it contains is presented.

From the standpoint of the compliance schemes, based on cost analysis of the end-of-life stages, the costs and revenues associated with e-waste treatment in France are also converted into economic indicators. The criticality of the materials generated, collected and recycled from e-waste is assessed based on the criticality assessment developed by the European Commission (EC).

The indicators are presented and validated with a case study focused on waste screens (category II of WEEE Directive), considering data and particularities of the e-waste chain in France. Nevertheless, we believe that the metrics introduced in this study could be used in the other Member States.

Given the level of data currently available, the indicators' operationalization is proposed in phases. Besides the suggestion of a set of indicators, the methods to calculate them, and the approach for data acquisition are presented.

The methodological approach adopted in this work is also a contribution for future research in the field and goes hand in hand with discussions presented by other authors. The methodological approach is based on five assessment levels: flows (WEEE categories or e-waste streams), products, components, materials and elements. The current approach only monitors the macroscopic level (WEEE categories), and we believe that to improve material and element recovery it is necessary to monitor e-waste flows at a higher level of detail.

The multidimensional approach presented in this study can support future policies and best practices in WEEE management in order to improve e-waste tracking and the recovery of (critical) raw materials. In so doing, more targeted WEEE management activities have the potential to extend the scope from waste and hazardous substances management to enhancing the supply of quality secondary raw materials.

This research is funded by the French Environment and Energy Management Agency (ADEME) and Ecologic (a French compliance scheme). It intends to support the development of monitoring indicators for the upcoming national specifications.

**Résumé |**

Les Déchets d'Équipements Électriques et Électroniques (DEEE) sont le principal flux de mine urbaine en raison de leur composition complexe et leur volume croissant. Dans l'Union Européenne (UE), la gestion des DEEE est réglementée par la Directive DEEE, la dernière en vigueur date de 2012 (Directive 2012/19/UE). Actuellement, dans l'UE, la performance de la filière DEEE est évaluée principalement au moyen d'indicateurs techniques qui visent à garantir la conformité aux objectifs de collecte et de valorisation fixés par la Directive DEEE.

L'objectif de cette thèse est d'établir un groupe d'indicateurs couvrant les aspects multidimensionnels liés à la collecte et au traitement des DEEE. Ces indicateurs visent à mieux évaluer les progrès réalisés par la filière réglementaire et à rendre compte de l'ensemble des aspects de l'économie circulaire. Au-delà de l'approche globale actuelle basée sur le poids, nous proposons des indicateurs à évaluer en tenant compte les particularités de la filière DEEE française. En plus de l'évaluation technique de la filière, nous proposons une évaluation des aspects environnementaux, économiques et de criticité, liées à la récupération des matières premières des déchets électroniques.

Les indicateurs et objectifs existants sont mesurés selon une approche globale basée sur le poids, sans tenir compte des pertes de traitement, et en excluant les flux non collectés par la filière réglementaire et captés par les flux complémentaires. En outre, la législation n'incite pas au recyclage de matériaux spécifiques, en particulier ceux contenus en faible quantité, parmi lesquelles des Matières Premières Critiques (CRM).

La Directive DEEE et la réglementation française fixent des taux de collecte et de traitement plus élevés pour les années à venir. Par conséquent, pour garantir une augmentation de la quantité et de la qualité des DEEE collectés, réutilisés et recyclés, il est nécessaire d'améliorer la connaissance et le contrôle des flux de DEEE. Il est donc important d'élargir le champ d'application des indicateurs pour mieux suivre les pertes et intégrer l'ensemble des flux de traitement (filière réglementaire et les flux complémentaires).

La complexité des DEEE implique une approche multidisciplinaire pour assurer une compréhension holistique des défis associés à leur gestion. Le cadre réglementaire des déchets électroniques en Europe, et plus précisément en France, est le fil conducteur des discussions et des propositions de cette thèse.

La quantification des flux de DEEE est essentielle pour le calcul des performances de la filière. Ainsi, l'Analyse de Flux de Matières (MFA) est la principale approche utilisée pour évaluer les performances techniques de la filière des DEEE. La MFA est utilisée dans cette étude pour suivre le bilan massique des flux, des produits, des composants, des matériaux et des substances dans les DEEE. Du point de vue environnemental, l'Analyse du Cycle de Vie (ACV) est la méthodologie utilisée pour évaluer les impacts environnementaux des déchets électroniques, suivie d'une discussion sur l'efficacité et les difficultés d'utiliser les résultats pour aider à la prise de décision dans la gestion de la filière des DEEE.

Compte tenu de l'importance de l'équilibre économique de la filière réglementaire, une évaluation du potentiel des DEEE en considérant la valeur des matériaux est présentée. Du point de vue des éco-organismes, les coûts et les revenus associés au traitement des déchets électroniques en France sont également convertis en indicateurs économiques. La criticité des matières générées, collectées et recyclées à partir des déchets électroniques est évaluée en considérant la méthode de criticité développée par la Commission Européenne (CE).

Les indicateurs sont présentés et validés avec un cas d'étude sur les écrans (catégorie II de la Directive DEEE) en tenant compte des données et des particularités de la filière française. Néanmoins, nous pensons que l'approche méthodologique et les indicateurs proposés dans cette thèse peuvent être utilisés par d'autres États membres de l'UE, quel que soit le modèle de filière de fin de vie pour les DEEE.

En considérant la qualité et la quantité des données actuellement disponibles, l'opérationnalisation des indicateurs est proposée par étapes. Au-delà de la proposition d'un groupe d'indicateurs, les méthodes de calcul ainsi que l'approche pour obtenir et fiabiliser les données sont présentées.

L'approche méthodologique adoptée dans ce travail contribue à faire progresser l'état de l'art dans ce domaine, et corrobore des discussions présentées par d'autres auteurs. Au-delà de la filière DEEE, la multiplication des filières REP oblige à construire un cadre de référence pour l'évaluation des filières de traitement des déchets, ces travaux pouvant facilement être dupliqués à d'autres filières.

L'approche méthodologique est basée sur cinq niveaux d'évaluation : flux (catégories de DEEE ou flux de déchets électroniques), produits, composants, matériaux et éléments/substances. Les indicateurs actuels ne surveillent que le niveau macroscopique (catégories de DEEE), et nos résultats indiquent que pour améliorer la récupération des matériaux et des éléments, il est nécessaire de surveiller les flux de déchets électroniques avec un niveau de détail plus fin.

L'approche multidimensionnelle présentée dans cette étude peut soutenir les politiques futures et les meilleures pratiques en matière de gestion des DEEE afin d'améliorer le suivi et la valorisation des DEEE et des matières secondaires (critiques). En changeant de paradigme, les acteurs de la gestion des DEEE ont le potentiel d'étendre leur offre de valeur au-delà du traitement des déchets et des substances dangereuses, afin de devenir des fournisseurs de matières premières secondaires (ou recyclées, MPR) de qualité.

Ces travaux de recherche ont été financés par l'Agence de l'Environnement et de la Maîtrise de l'Énergie (ADEME) et Ecologic (éco-organisme français de la filière DEEE). Les résultats de ces travaux ont l'intention de soutenir l'élaboration d'indicateurs de suivi pour les prochains cahiers des charges de la filière réglementaire.

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*"A vida é a arte do encontro, embora haja tanto desencontro pela vida."*
Samba da benção - Baden Powell e Vinícius de Moraes (1967)

As Vinicius de Moraes[1] once said: "life is the art of encounter although there is so much mismatch throughout life." This work is a result of many special encounters I had in the past years that helped me build my path.

Nicolas Perry and Carole Charbuillet thank you for supervising this work. I'm grateful for your support and for letting me lead my research.

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I have no words to express my gratitude for my beloved family and friends in Brazil and Portugal. Even miles away, your love and compassion are one of my main sources of energy for improving my work.

If I genuinely consider France my home now it is thanks to my stepfamily and friends. Nico, thank you for your support and for believing in me in the hardest moments of this journey.

[1] Vinicius de Moraes (1913-1980) was a Brazilian poet, lyricist, essayist, and playwright. He wrote lyrics for a great number of songs that became all-time bossa nova and samba classics.

My hope is that the results of this thesis can also support the improvement of e-waste management outside the European Union, especially in Brazil, which hopefully will soon find its way back to a socio-environmental development.

## Chapter 1 | Introduction

*"Enquanto eu tiver perguntas e não houver resposta continuarei a escrever.*[2] *"*
Clarice Lispector, A hora da estrela (1977)

### 1.1. Background

The population rise and new technological developments go hand in hand with an expansion of energy and raw material consumption which entails waste and emissions as by-products (Schandl et al., 2016). The growing need for materials also implies a search for solutions to ensure future material supply, since natural resources are finite and non-renewable on a human time scale entails (Hofmann et al., 2018). In the European Union (EU), the secure and sustainable supply of raw materials has a crucial aspect due to the importance of raw materials for European industry, and its reliance on imports (Løvik et al., 2018).

In this context, the circular economy is seen as a lifeline to achieving better resource management by extending and closing material cycles and reducing waste (Zeller et al., 2019). This new economic model is an alternative to the linear 'take, make, dispose' economy (Parajuly and Wenzel, 2017).

Circular economy concepts are promoted at European level through the Circular Economy Package (European Commission, 2015). This transition is seen as an opportunity to decrease resource dependence and waste, as well as to increase employment and growth (Ellen MacArthur Foundation, 2015). In France, the Energy Transition for Green Growth Act, enacted in 2015, set targets and actions to encourage circularity, among which to halve the amount of landfilled waste and recycle 60% of waste by 2025; as well as incentives for product design prioritizing the use of secondary raw materials (JORF, 2015a). In 2018, the French government published the Roadmap for the Circular Economy. Together with improvements in production and consumption, waste management is presented as an important key to closing the economy loop. Several measures and targets were defined to boost ecodesign, reuse and recycling (Ministry for an Ecological and Solidary Transition and Ministry for the Economy and Finance, 2018).

As illustrated in Figure 1-1, closing the materials loop is at the heart of the circular economy model, by highlighting that resources contained in wastes should be recovered and utilized as much as possible (Cossu and Williams, 2015). The process of reclaiming materials and components from waste is known as urban mining (Gutberlet, 2015). Waste Electrical and Electronic Equipment (WEEE) is among the key urban mining stream due to its composition and rising volumes (Koutamanis et al., 2018; Zeng et al., 2018). Other important urban mining streams highlighted in the literature are scrap vehicles and

[2] So long as I have questions to which there are no answers I shall go on writing.

construction and demolition waste. Alongside primary extraction, urban mining is an alternative source of materials (Tunsu et al., 2015).

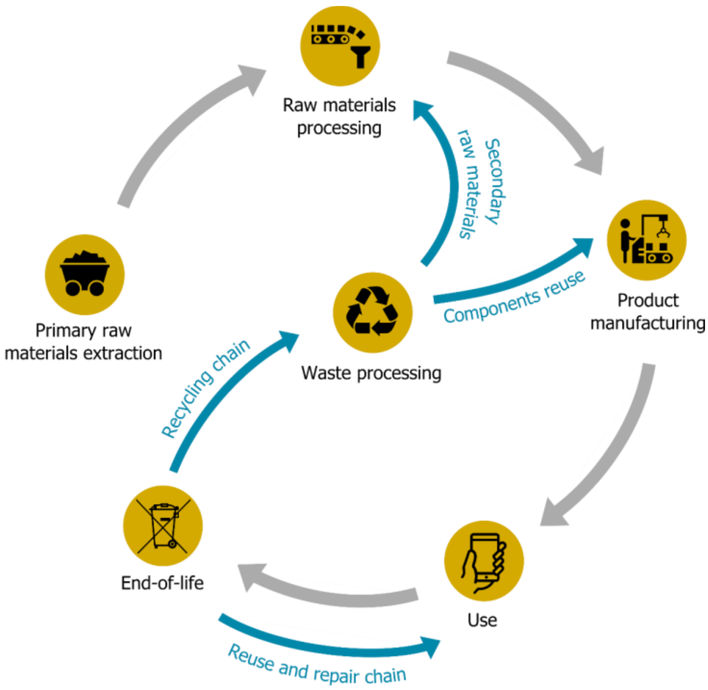


Figure 1-1. Circular economy perspective: from waste to secondary resources.

In this context, waste treatment actors have a crucial role in the setup of the circular economy through the management and redirection of waste flows into treatments to extend the life cycle of products and materials. Nevertheless, a circular economy transition requires moving beyond waste management and recycling, towards sustainable development and use of materials and products through circularity strategies (e.g. extending product lifetimes and increasing high-grade recycling) (Potting et al., 2017).

### 1.1.1. Waste: a paradigm shift

When the first discussions on implementing waste management policies started, the focus was pollution control to reduce environmental and health risks. In Europe, the development of waste policies started in the seventies with the first Waste Framework Directive 75/442/EEC (European Commission, 1975). After its publication the Directive was amended several times in order to improve efficiency and reflect the most recent issues related to waste management (Más, 2016).

In the 90s the concept of a waste hierarchy was introduced. As presented in Figure 1-2, prevention is seen as the most desirable option, followed by preparing waste for reuse, recycling and other recovery (e.g. energy recovery). Landfill occupies the bottom of the hierarchy, as this option is regarded as the least desirable for the environment (Manfredi and Goralczyk, 2013). In 2006 the

Directive 75/442/EEC was codified into one new text that replaced all the previous versions, and in 2008 the Waste Framework Directive (revised) entered into force.

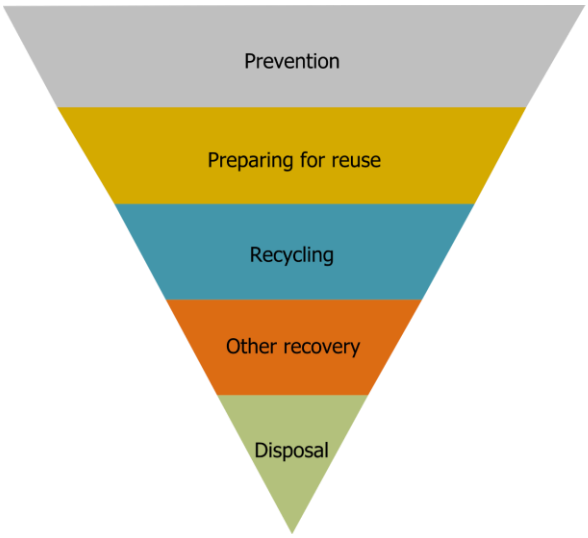


Figure 1-2. Waste hierarchy.
Source: Adapted from European Commission, 2008.

As a further part of its overall waste management strategy, the European Commission (EC) has defined several specific waste streams for priority attention (e.g. packaging waste, end-of-life vehicles, batteries and WEEE). As a consequence, a series of directives and legal acts applying to specific waste streams were released (European Parliament, 2015a).

Nowadays the EU waste management policy is built on policy strategies, directives and legal acts applying to specific waste streams. Two principles apply across all waste streams: the 'waste hierarchy' previously mentioned, and the 'polluter pays principle' (those who produce pollution should bear the costs of management and treatment). The obligation for producers to take operational or financial responsibility for the End-of-Life (EOL) phase of their products is named Extended Producer Responsibility (EPR), and it is used in several waste stream chains across the EU (European Parliament, 2015b).

In France, the EPR principle has existed in law since 1975 and is codified in Article L. 541-10 of the Environment Code. The first EPR system implemented was on household packaging in 1992, and nowadays twenty EPR chains exist for different products, at diverse stages of development (ADEME, 2017).

Each EPR systems has its particularities, but they share common principles that underlie the implementation of EPR, such as the set-up of a minimum target for reuse, recycling and recovery, regulatory financing requirements, and application of a fee when products are placed on the market. The main actors of EPR chains in France, and their roles, are detailed below (ADEME, 2015):

**Authorities**

Establish the regulatory framework for targets, allocate responsibility among actors, grant authorizations, etc., and ensure proper implementation of EPR chains, including monitoring products and waste chains. This includes different government actors that contribute to policy development and the set-up of the system, as well as the French Environment & Energy Management Agency (ADEME) that does the steering and monitoring of EPR chains.

**Compliance schemes**

Also known as take-back schemes or producer responsibility organizations. On behalf of the producers, they organize the collection and treatment of waste. These organizations are nonprofit bodies and are certified by government authorities under terms of reference (specifications) that set the guidelines and national targets that should be achieved.

**Clearinghouse**

Ensures a level playing field for all compliance schemes and uniform service levels. It is also responsible for the equalization of compliance scheme operations according to their market share. Another central mission is to place contracts with local authorities and to pay them a financial support for collection. In the WEEE EPR chain, OCAD3E (an acronym for Unified Coordinating Center for WEEE Management) is a subsidiary of the compliance systems, and it is evaluated and re-accredited every five years by the French authorities.

**Consumers (waste holders)**

They must sort and dispose their waste in accordance to the mechanisms established to collect the waste. It includes both household and professional waste holders.

**Distributors (retailers)**

Must inform consumers of the proper end-of-life management of used products and may also be required to take back end-of-life products free of charge.

**Producers**

Must participate to the waste management chain either by joining one of the compliance schemes and contributing financially, or by setting up their own system (handling waste streams). It includes manufacturers, distributors, and importers.

**Waste treatment providers**

They operate under contract to provide waste management services, including collection, transport, storage, preparation for reuse, recovery of materials, and final disposal.

As a consequence of the evolution of legal requirements and technologies, waste management has evolved progressively in the EU (WEEElogic, 2019). The recent efforts on shifting to a circular economy in the EU have highlighted the importance of turning waste into a resource (European Parliament, 2014). To achieve this goal, it is essential to improve the current EPR chains.

### 1.1.2. (W)EEE: a potential and complex stream

Electric and Electronic Equipment (EEE) are complex multi-material products in which a variety of elements are connected both by constructive and chemical connections (Van Schaik and Reuter, 2014). If a mobile phone in the 1990s used a dozen major metals, modern smartphones consist of up to 60 elements in various mixtures of metals (Nguyen et al., 2018).

Annually, millions of tons of EEE are placed on the market in the world leading to a massive generation of waste. In 2018 in France, 28.8 kg of EEE per inhabitant were placed on the market, an increase of 15% in comparison to 2012. The amount of electronic waste (e-waste) is rising globally, with an annual growth of approximately 5% (Ibanescu et al., 2018). In 2016, 44.7 million tons (Mt) of WEEE, were generated in the world. Europe is the second continent in e-waste generation (12.3 Mt), only Asia produces more (18.2 Mt) (Baldé et al., 2017).

Besides being the fastest growing waste stream, WEEE is a particularly complex one due to its material composition (Vadoudi et al., 2015). While it contains some high-value materials (such as gold, copper, rare earth elements and palladium), it also includes some toxic ones (e.g. mercury, lead and brominated flame retardants).

The regulatory scope for e-waste varies significantly between countries, as does the development level of WEEE management systems (Tansel, 2017). In the EU, WEEE management is regulated by the WEEE Directive (Directive 2012/19/EU) (European Commission, 2012). In France, it was transposed into French law, together with the RoHS Directive (Directive 2002/95/EC) (mostly by Decree 2005-829) (JORF, 2005), to regulate the composition of EEE and to manage WEEE. It was later complemented by other legislation and by some additions made to the French Environmental Code.

The WEEE Directive aims to prevent the generation of e-waste, as well as to improve the performance of the treatment operations for reuse, recycling and other forms of recovery (Wäger and Hischier, 2015). It introduces the concept of EPR which means that producers of EEE (manufacturers and importers) are accountable for the EOL of appliances that they have placed on the market. The ERP has led to the creation of compliance schemes.

Europe is the continent with the highest collection rate, due - among other reasons - to the directives, regulations and systems established for e-waste since 1990. Nevertheless, collection performance per country differs significantly. In 2016, for example, the collection rate was 67% in Sweden, 45% in France and 10% in Malta (Eurostat, 2018).

These variations are explained by the wealth, the development stage of the official schemes, as well as the lack of harmonization across the EU Member States of the methodology for gathering data to calculate the performance rates (Baldé et al., 2017; Salhofer et al., 2016).

### 1.1.3. Main drivers for e-waste treatment

Because of the e-waste composition, its treatment is driven by three main benefits/reasons: economic, environmental and public health. From an economic perspective, WEEE contains precious metals (e.g. gold, silver, and palladium) and other valuable metals (e.g. copper, aluminum, and iron) (Ibanescu et al., 2018).

From a resource perspective, recycling of e-waste, at a certain scale, decreases the pressure on the environment caused by the extraction of raw materials from mineral deposits (Meshram et al., 2019). From the outlook of sustainability, we should rationalize raw material use considering the needs of future generations. Recycling can also contribute to the security of raw material supply and progress towards a more circular economy (Mathieux et al., 2017). Nonetheless, from a public health perspective, this type of waste also contains hazardous substances that represent risks for human health and the environment, and that require specific treatment (e.g. mercury and lead) (Lixandru et al., 2017). It is known that the WEEE chain results in environmental impacts (e.g. emissions related to transport, diverse emissions, and resource consumption associated with energy consumption for reuse and recycling, etc.). However, the impacts are expected to be lower than the production of raw materials.

The main economic driving force for e-waste recycling is copper and precious metal (e.g. gold, palladium and silver), given their significant content in comparison to mineral ores (Awasthi et al., 2018; Kumar et al., 2017). For example, in certain mobile phones gold represents 0.04 % of the total weight, a concentration of 400 g/t – 200 times greater than typical gold ore (1-3 g/t), and 14 times higher than the highest known grade deposit (28 g/t) (Charles et al., 2017).

Due to its potential as a source of valuable secondary resources, WEEE has been identified as a lucrative business in both developed and developing countries (Sinha et al., 2016). Nevertheless, between the e-waste collection by the official schemes and the actual production of secondary raw materials, there is a complex chain of processes driven by economic and environmental interests (e.g. control of substances such as lead and mercury) (Valero Navazo et al., 2013). Widespread e-waste generation, combined with the existence of secondary non-official schemes, entails high collection costs and modest collection by the official schemes. Due to the complexity and heterogeneity of e-waste, depending on its type, different techniques for clean-up, dismantling, shredding, sorting and recycling must be combined in order to recover the secondary raw materials. This treatment has a significant cost when carried out in compliance with all environmental and health standards. As a consequence, despite the intrinsic value of the materials in WEEE, sometimes profitability does not properly counterbalance capital and operating costs (Cesaro et al., 2018). Some recycled materials, such as polymer resins, may be more expensive to treat than the equivalent raw materials (IEA, 2014).

### 1.1.4. Assessment of e-waste chain performance

Indicators are a means to transform complex data into key information for decision-makers, as well as non-technical specialists (Cifrian et al., 2015). As a waste management tool, indicators are used to quantify the impacts and benefits of the waste chain, and also to monitor progress over time and compare performance between one or more systems (Giljum et al., 2011). They can be used to assist in decision-making, in addition to communicating results to different stakeholders.

Currently, in the EU, e-waste chain performance is mainly assessed by technical indicators that aim to ensure system compliance with collection and recovery targets set by the WEEE Directive. In this study, indicators are classified as *technical* when they aim to quantify the e-waste flows (e.g. waste generated, collected, recycled), or measure the effectiveness of a process or waste system (e.g. collection and recycling rate).

The WEEE Directive establishes three technical indicators to monitor WEEE system efficiency: collection rate, recycling and preparation for reuse rate, and recovery rate (art. 3 of the WEEE Directive). Besides the WEEE Directive, other indicators are proposed by stakeholders in e-waste treatment (e.g. policy-makers, compliances schemes, and associations) to assess the performance of e-waste treatment in a country or to compare the performance between different countries. For example, the Raw Materials Scoreboard released by the EC compares the WEEE chain performance in the Member States with indicators based on the weight of WEEE collected (total and per household), recycled and reused, divided by the number of inhabitants (kg per capita) (Vidal-Legaz et al., 2018).

The existing indicators and targets are measured in an overall weight-based approach without considering treatment losses, and excluding flows not collected by the official schemes but captured by complementary flows (Haupt et al., 2016; Parajuly et al., 2017). WEEE not collected by the official schemes ends up in other non-documented routes, including equipment reused, e-waste disposed in the municipal waste stream, as well as undocumented exports either as (illegal) waste or as reusable items (Bigum et al., 2013; Huisman et al., 2015).

Moreover, the WEEE Directive does not set incentives for specific materials, in particular those contained in low or trace amounts, which includes Critical Raw Materials (CRMs) (Van Eygen et al., 2016). There is a lack of clear and robust operational indicators and targets covering sustainability aspects related to WEEE management (Nelen et al., 2014). Furthermore, seeing the potential of WEEE as an urban mine, there is a need for indicators that can monitor the performance of the e-waste chain as a supplier of quality secondary (critical) raw materials.

### 1.1.5. Industrial context of this thesis: focus on the French chain

The French EOL chain for professional e-waste has been operational since 2005, and 2006 for household e-waste. According to data provided by ADEME, in 2018 the e-waste chain has complied to the collection target set by the WEEE Directive (45%). However, it has not complied with the collection target fixed by the French regulation (59%) (Fangeat, 2019). More details of EU and French rates, as well as the performance of French e-waste chain are presented in **Chapter 2**.

From 2015 to 2020 three compliances schemes are approved by the public authorities for the management of WEEE in France: Ecologic, ESR[3] (formerly Eco-Systèmes and Récylum) and PV Cycle. As an approved compliance scheme, Ecologic, co-financer of this research, is faced with various challenges to develop the WEEE sector, among which:

- increasing the collection rate seeing the ambitious collection targets set by the French regulations for the following years (65% in 2019 and 2020);
- improving the recycling rate of the materials contained in the e-waste;
- developing recycling of plastics and critical raw materials;
- ensuring compliance with the French legislation requirements and the targets set by the WEEE Directive.

The approved compliance schemes must comply with the specifications provided by the Environmental Code, which presents guidelines and national targets (annexes of the Decree of 2 December 2014 on the approval procedure for household WEEE and Decree of 20 August 2015 for professional WEEE). According to these specifications, each year the compliance schemes must report to ADEME and the Ministry of the Environment (currently named as "Ministry for an Ecological and Solidary Transition") the indicators related to household and professional WEEE management (JORF, 2015b, 2014). The list of monitoring indicators requested is presented in the household and professional specifications. However, the authorities detail neither the method nor which criteria and scope should be used to calculate the indicators.

Therefore, this research can support the development of future specifications, through the suggestion of a set of indicators, including the methods to calculate them, as well as an approach for data acquisition. It can also improve knowledge regarding the output fractions of e-waste treatment (composition and secondary use), and on the assessment of the environmental impacts and benefits of the official schemes.

[3] While this thesis was being finalized, the name of the compliance scheme (on its site and database) was ESR. More information about the most recent change in the compliance scheme name can be found here: https://www.actu-environnement.com/ae/news/dechets-electrique-electronique-ecosystem-energie-recylum-recyclage-34218.php4

## 1.2. Research goal and questions

The main goal of this thesis, aligned with the needs of the partners of this research (ADEME and Ecologic) is:

***To establish a robust set of indicators covering the sustainability (technical, economic, environmental and criticality) aspects related to the collection and treatment of WEEE, in order to improve visibility on the progress of the official WEEE schemes in a circular economy.***

In this context, the general research question of this thesis is:

***Could WEEE chain performance be improved with a set of indicators that boost its assessment and monitoring?***

This PhD thesis has developed methods and indicators in order to achieve this objective and answer the general research question. The general research question was subdivided into four research questions (RQ):

**RQ1. Do the current indicators provide any information, beyond the technical performance, regarding the progress of the WEEE chain?**

This question is related to mapping and classifying the current indicators according to their dimension of analysis: technical, economic, environmental and social. A literature review of the indicators currently used by the WEEE sector, as well as published in the literature, is presented in **Chapter 3**.

**RQ2. What are the relevant indicators and methods to assess the technical performance of the WEEE chain?**

This question refers to identifying the most suitable technical indicators to support the quantification of collection and treatment performance, without limiting the assessment to an overall weight-based approach. The technical indicators selected from the literature, and some new indicators, are presented in **Chapter 4**. The feasibility of the indicators is assessed with a case study based on the screens flows in France.

**RQ3. How should the environmental and economic performance of the WEEE chain be assessed?**

This research question investigates the methods available for evaluating the environmental and economic impacts and benefits of the WEEE chain. The methods and indicators identified are presented, respectively, in **Chapters 5 and 6**. In the light of the importance of some raw materials for the EU economy, an approach based on criticality as a parameter to assess material recovery from WEEE is discussed in **Chapter 7**. A case study based on the screens flows in France is presented at the end of each chapter in order to illustrate the results obtained with the indicators selected.

**RQ.4. Based on the results of the multi-dimensional indicators, is it possible to propose improvements towards a more sustainable and circular (W)EEE chain?**

This research question focuses on the use of indicators to monitor progress over time and to support decision-making in the (W)EEE chain. This topic is presented in **Chapter 8**, along with a discussion regarding the essential parameters and the approach for data acquisition to allow the feasibility of the indicators.

This research focuses on the French WEEE system and aims to improve monitoring of the e-waste flows by the actors of the official schemes (e.g. ADEME and compliance schemes). Thus, the indicators are validated in a case-study considering data and particularities of the e-waste chain in France. The European regulatory framework is considered in the research since France transposed the WEEE Directive into national legislation, and it must comply with European targets and programs. Moreover, we believe that the results of this study could be used in other Member States, to support the evolution of WEEE chains from waste treatment towards secondary raw material providers.

This thesis is structured around the four research questions, and the approach and steps to answer them are presented in the following sections.

### 1.3. Research approach and methodologies applied

The complexity of e-waste implies a multidisciplinary approach to ensure a holistic comprehension of the challenges associated with its management. Besides the technical aspects related to the technologies and the economic impacts (costs and profits), the regulatory framework cannot be neglected in waste management. This chain also interacts with the environment; therefore, the social and environmental issues should also be considered.

The regulatory framework of e-waste in Europe, and more precisely in France, is the guideline for the discussions and for this thesis. Thus, **Chapter 2** presents the regulation background for WEEE, as well as structure of the WEEE chain in France and the different actors of the EOL chain.

While the main goal of this study is to propose a suitable set of indicators, an assessment of the existing indicators is the starting point of the research and is presented in **Chapter 3.** It also includes the method used to classify the indicators identified in the literature review, as well as the limitations and gaps of the current indicators.

Quantifying e-waste flows is crucial for calculating the performance of the chain. Thus, Material Flow Analysis (MFA) is the main methodology used to assess the technical performance of WEEE chain presented in **Chapter 4**. MFA is applied in this study to track the mass balance of flows, products, components, materials, and substances in e-waste. In order to estimate the flows throughout all end-of-life stages, different areas of expertise are required such as waste management and process engineering. This last one encompasses the knowledge of different engineering processes required for handling the waste to obtain the secondary raw materials. Some of these processes are described in section 4.4. These processes allow the recovery of different polymers and metals from heterogeneous e-waste flows composed of complex multi-materials products.

Life Cycle Assessment (LCA) is the main methodology used to assess the environmental impacts of e-waste. A review of the LCA studies, followed by a case study applied to screens recycling in France is presented in **Chapter 5**. The scope and methodological choices for assessing WEEE chain impact and benefits with LCA, and the effectiveness and difficulties of using the results to support decision-making in WEEE management are discussed in section 5.6. Finally, section 5.7 present environmental indicators to report the environmental impacts and benefits of WEEE chain.

As previously mentioned, the value of the secondary raw materials is one of the drivers for the WEEE chain. Thus, an assessment of the WEEE chain potential from the perspective of the intrinsic value (also known as market value) of the materials is presented in **Chapter 6** (section 6.3.2). However, before the secondary raw materials are ready to be used in a new product, several treatment steps must be implemented, which imply treatment costs. From the outlook of the compliance schemes, based on cost analysis of the EOL stages, the costs and revenues associated with e-waste treatment in France are converted into economic indicators in section 6.3.1.

The criticality of raw materials is one of the new parameters presented in this study to assess the recovery performance of materials from e-waste (section 7.5). It is based on the criticality assessment developed by the European Commission (EC) (Blengini et al., 2017). The indicators developed in **Chapter 7**, and part of the indicators presented in **Chapter 4**, were developed during a five-month research period at the Joint Research Centre (JRC) of the EC (Directorate Sustainable Resources - Unit D.3 Land Resources).

The set of indicators developed in the thesis, as well as the discussions regarding the potential ways they could support different actors of the WEEE chain, are presented in **Chapter 8**. This discussion intends to support Ecologic and ADEME in the development of the specifications for the compliance schemes that will be published in 2021, as presented in section 1.1.5. Lastly, a summary of the methods and results of the thesis in French is presented in **Chapter 9**.

Information regarding WEEE treatment and secondary raw material availability is useful, for example, for EU and national policy-makers, compliance schemes, recyclers, and designers. Thus, this research is relevant to a broad community of readers besides the scientific community in the field and could encourage future improvements in the WEEE legislation as well as research in the field.

## 1.4. Thesis outline

The structure of this thesis follows the research questions and the approach described in the previous sections. The chapters are organized into six areas, and Figure 1-3 illustrates the link between the chapters and the research questions.

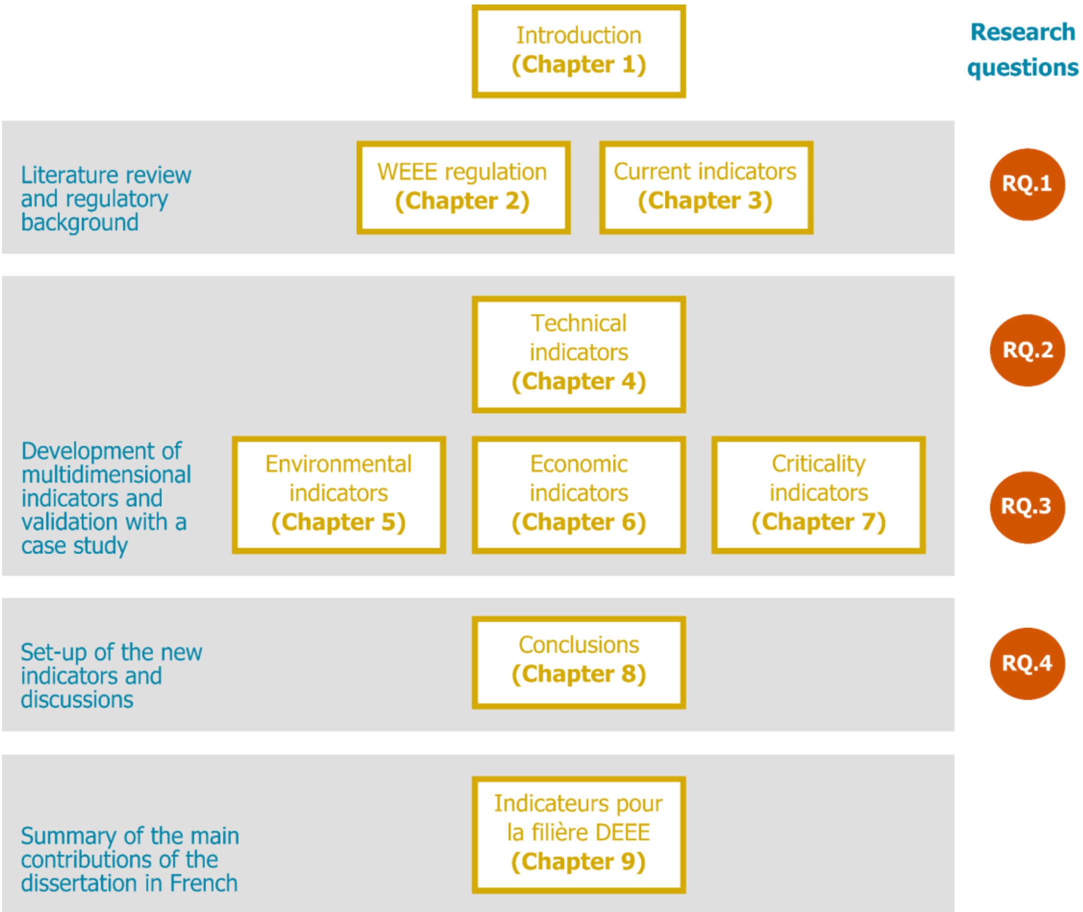


Figure 1-3. Structure of this thesis.

- regulatory background (**Chapter 2**);
- assessment of the current indicators (**Chapter 3**);
- WEEE flows mapping and characterization (**Chapter 4**);
- development of multidimensional indicators and experimentation: technical (**Chapter 4**), economic (**Chapter 6**), environmental (**Chapter 5)**, and based on the criticality (**Chapter 7**);
- set-up of the new indicators and discussions (**Chapter 8**); and
- summary of the main contributions of the dissertation in French (**Chapter 9**).

Following the presentation of the indicators in **Chapters 4, 5, 6** and **7**, a case study focused on the treatment of screens in France is presented to validate the indicators. This organization aims to facilitate the comprehension of the contribution of this dissertation and also it follows the research questions.

## 1.5. Terminology and definitions

In this section, the most important terms and definitions used in the dissertation are presented clarify aspects discussed in the following sections. This glossary does not intend to be exhaustive of all relevant terms in the field.

Collection | the gathering of waste, including the preliminary sorting and preliminary storage of waste, for the purposes of transport to a waste treatment facility (European Commission, 2008).

Collection rate | according to the WEEE Directive, it is calculated based on the total weight of waste collected by the official schemes, expressed as the percentage of the overall weight of EEE placed on the market in the three preceding years. It is alternatively expressed as a percentage of the waste generated.

Complementary flows | all waste flows that are not reported at a national level by the official compliance systems. A certain portion of these flows are exported, incinerated or landfilled. The term also includes non-compliant treatment, like recycling with other waste streams, for instance, with mixed metal scrap (Huisman et al., 2016).

Compliance schemes | collective entity set up by producers or through legislation which becomes responsible for meeting the recovery and recycling obligations of the individual producers (European Commission, 2014). It is also known as take-back systems and producer responsibility organizations (PRO).

Circular economy | global economic model that decouples economic growth and development from the consumption of finite resources. It is restorative by design, and aims to keep products, components and materials at their highest utility and value, at all times (Ellen MacArthur Foundation; & Granta Design, 2015).

Critical Raw Materials (CRMs) | according to the methodology developed by the EC, a material is considered as critical for the EU if it has high economic importance to the economy of the EU and its supply is vulnerable to disruption (Deloitte Sustainability et al., 2017).

Downcycling or non-functional recycling | process converting materials into new materials of lesser quality and reduced functionality (Ellen MacArthur Foundation; & Granta Design, 2015).

Disposal | means any operation which is not recovery, even where the operation has as a secondary consequence the reclamation of substances or energy (European Commission, 2008).

Economic importance | in the EC methodology, it is calculated based on the importance of a given material in the EU end-use applications and the performance of available substitutes in these applications (Deloitte Sustainability et al., 2017).

Electrical and Electronic Equipment (EEE) | equipment which is dependent on electric currents or electromagnetic fields in order to work properly and equipment for the generation, transfer and measurement of such currents and fields and designed for use with a voltage rating not exceeding 1,000 volts for alternating current and 1,500 volts for direct current (European Commission, 2012).

Element | fundamental substance. It cannot be separated into smaller substances by physical or chemical methods (e.g. iron metal – chemical element with symbol Fe).

End-of-Life (EOL) | is defined as the part of the life-cycle of a certain product starting with the point of disposal by a customer (this can be a consumer, organization or business) till the point where materials are re-entering production chains. Remanufacturing and reuse also fall under the end-of-life chain (Huisman, 2003).

End-of-life recycling input rate ($EOL_{RIR}$) | measures, for a given raw material, how much of its input into the production system comes from recycling of "old scrap" i.e. scrap from end-of-life products. The $EOL_{RIR}$ does not take into account scrap that originates from manufacturing processes ("new scrap") (Eurostat, 2016).

End-of-waste | result of treatment whereby the resulting fractions are no longer classified as waste (European Standardization Organizations, 2014).

Extended Producer Responsibility (EPR) | an environmental policy approach in which a producer's responsibility for a product is extended to the post-consumer stage of a product's life cycle (European Commission, 2014).

Functional recycling| is the portion of EOL recycling in which the material in a discarded product is separated and sorted to obtain recyclates. Recyclates obtained by functional recycling are used for the same functions and applications as when obtained from primary sources. As opposed to recyclates generated from non-functional recycling which substitute other raw materials, and therefore do not contribute directly to the total supply of the initial raw material (Talens Peiró et al., 2018).

Fractions | streams of multiple materials with one or more target materials to be recovered at secondary processes (recyclates) or as (residue) streams treated by final waste processors (Huisman, 2003).

Indicator | is a value representing the performance of a system or organization. It gives an idea of its state or level of compliance and is often assessed by comparing the result with an agreed standard or target.

Life Cycle Assessment (LCA) | compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle (ISO, 2006).

Linear economy | consists of 'take, make, dispose' industrial processes and associated lifestyles resulting in a depletion of finite reserves. Virgin materials are used to create products that end up in landfills or incinerators (Ellen MacArthur Foundation; & Granta Design, 2015).

Material | composed of two or more elements with a given proportion. It can be separated into simpler substances and elements by chemical methods (e.g. steel alloys).

Material Flow Analysis (MFA) | method to quantify flows and stocks of materials or substances in a system.

Mechanical treatment | the sorting and shredding of (streams of) discarded consumer electronics followed by separation of materials into fractions by processes like magnets, Eddy-Current separation or air tables (Huisman, 2003).

New scrap | the scrap generated from processing and manufacturing processes and it is also sometimes regarded as pre-consumer scrap. It has a known composition, normally high purity, and origin, and can be often recycled within the processing facility (Talens Peiró et al., 2018).

Old scrap | the amount of material contained in products that have reached their EOL, also regarded as post-consumer scrap. It is often mixed with other materials such as plastics or alloys, therefore its recycling requires further detailed processing for proper recovery (Talens Peiró et al., 2018).

Primary raw materials | virgin materials, natural inorganic or organic substance, such as metallic ores, industrial minerals, construction materials or energy fuels, used for the first time (Deloitte Sustainability et al., 2017).

Raw material | natural or processed resources which are used as an input to a production operation for subsequent transformation into semi-finished and finished goods. Primary raw materials, as opposed to semi-finished products, are extracted directly from the planet and can be traded with no, or very little, further processing (Deloitte Sustainability et al., 2017).

Re-use or reuse | any operation by which products or components that are not waste are used again for the same purpose for which they were conceived (European Commission, 2008).

Recovery | any operation whose the principal result is waste serving a useful purpose by replacing other materials which would otherwise have been used to fulfil a particular function, or waste being prepared to fulfil that function, in the plant or in the wider economy (European Commission, 2008).

Recovery rate | according to the WEEE Directive, it is calculated based on the total weight of WEEE that enters the recovery facility, expressed as the percentage of the weight collected by the official schemes. It should be calculated per WEEE category.

Recycling | any recovery operation by which waste materials are reprocessed into products, materials or substances, whether for the original or other purposes. It includes the reprocessing of organic material, but does not include energy recovery nor the reprocessing into materials that are to be used as fuels or for backfilling operations (European Commission, 2008).

Recycling and prepared for re-use rate | according to the WEEE Directive, it is calculated based on the total weight of WEEE that enters the recycling/preparing for reuse facility, expressed as the percentage of the weight collected by the official schemes. It should be calculated per WEEE category.

Reported collection or registered flows | quantities of WEEE reported to national registers and Eurostat WEEE database (Huisman et al., 2016).

Scavenging | removal of valuable components outside the official schemes, only considering reuse or material value in e.g. compressors from temperature exchange equipment, hard disks, memory and other small IT components (Huisman et al., 2016).

Secondary raw materials | defined as materials produced from other sources than primary. Secondary raw materials can also be obtained from the recycling of raw (i.e. primary) materials. Examples: steel or aluminum scrap (Deloitte Sustainability et al., 2017).

Substance | matter having an invariant chemical composition and distinct properties.

Supply risk | in the EC methodology, it is calculated based on factors that measure the risk of a disruption in supply of a specific material (Deloitte Sustainability et al., 2017).

Sustainability | the notion of sustainability varies considerably, and there are numerous definitions in the literature (Van de Kerk and Manuel, 2008). In this study we consider, from an anthropocentric point of view, that sustainability means development that meets the needs of the present generation without compromising the ability of future generations to meet their needs. It covers three aspects: (1) environment and resources depletion; (2) economic; and (3) social.

Upcycling | denotes a process of converting materials into new materials of higher quality and increased functionality (Ellen MacArthur Foundation; & Granta Design, 2015).

Urban mining | process of reclaiming compounds and elements from any kind of anthropogenic stocks, including buildings, infrastructure, industries and products (in or out of use) (Cossu and Williams, 2015).

Waste generated | corresponds to the total mass of discarded products (waste) as a result of consumption within the territory of that Member State in a given reporting year, prior to any activity (collection, preparation for reuse, treatment, recovery (including recycling) or export) after discarding. Waste arising from private, business and industrial sectors (Huisman et al., 2016).

Waste management | means the collection, transport, recovery and disposal of waste, including the supervision of such operations and the after-care of disposal sites, and including actions taken as a dealer or broker (European Commission, 2008).

Waste Electrical and Electronic Equipment (WEEE) | electrical and electronic equipment which is waste, including all components, sub-assemblies and consumables which are part of the product at the time of discarding (European Commission, 2012).

WEEE chain | activities and actors performing various functions in the treatment of e-waste.

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## Chapter 2 | WEEE regulation

*"A distância mais curta entre dois pontos*
*pode ser a linha reta, mas é nos caminhos curvos*
*que se encontram as melhores coisas.*[4] *"*
Lygia Fagundes Telles, Ciranda de Pedra (1954)

### 2.1 EU Directives related to (W)EEE

The process of developing waste policies in Europe began in the seventies when the first Waste Framework Directive (75/442/EEC) was published. The process continued further and expanded into several Directives specific to the different waste streams. In 1990, the European Commission (EC) initiated the Priority Waste Streams Program focused on six different streams. Waste Electrical and Electronic Equipment (WEEE) was selected among these main waste streams because of the fast growth of technological innovation, the burden it brought to municipal authorities and its complex composition. In 1992 the Basel Convention identified e-waste as hazardous, and developed a framework for controls on the trans-boundary movement of such waste (Widmer et al., 2005).

The first version of the WEEE Directive (Directive 2002/96/EC) came into force in 2002, parallel with the RoHS Directive (Directive 2002/95/EC). The RoHS Directive restricts the use of certain hazardous substances (e.g. lead and mercury concentration higher than 0.1%) in Electrical and Electronic Equipment (EEE) to increase the protection of human health and facilitate environmentally-sound recovery and disposal of WEEE (Friege, 2012; Kumar et al., 2017). By December 31, 2006, the Member States had to ensure the compliance with WEEE Directive targets, and the hazardous substances were banned from July 1, 2006. Following the increase in (W)EEE generation and its complexity, both WEEE and RoHS Directives were revised in 2012 and 2011 respectively and became effective two years later.

Seeing the importance of improving the ecodesign of energy-using products, which include some EEE, in 2005 the EC published the Ecodesign Directive (Directive 2005/32/EC), revised in 2009. The Ecodesign Directive sets minimum mandatory requirements for energy-using products.

Figure 2-1 presents the simplified life cycle of EEE and its interaction with the three (W)EEE Directives aforementioned. Taking a closer look at WEEE Directive, the main subject of this work, its scope is focused on the collection and treatment of WEEE by the official schemes.

[4] The shortest distance between two points may be a straight line, but it's in the curving paths that the best things are found.

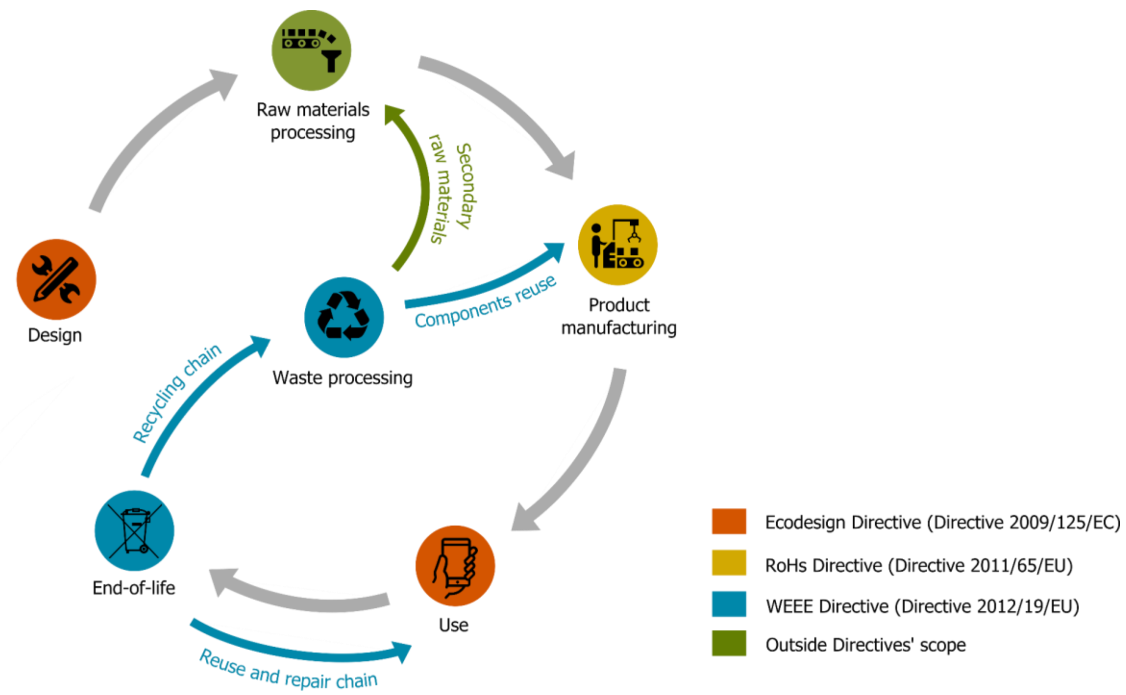


Figure 2-1. EEE life cycle and (W)EEE Directives.

The collection is comprised of storage, transport and regrouping of e-waste prior to treatment. The WEEE treatment can be divided into three main steps: (I) pre-treatment (including clean up, dismantling, shredding and sorting); (II) disposal of non-recyclable fractions; (III) recovery and production of secondary raw materials. These steps are detailed in **Chapter 4**, but it is important to remark that nowadays the focus and scope of the WEEE Directive and monitoring are on the first two steps: until the *end-of-waste status* is achieved for the recoverable fractions or disposal of non-recoverable fractions. In the third step (recovery and production of secondary raw materials materials), the fractions are already seen as *products* (instead of *waste*), which results in a lower monitoring level by the compliance schemes in comparison to the previous steps.

This chapter presents the European Union (EU) and French regulation on WEEE, with a focus on the current targets and indicators imposed by the regulations.

## 2.2 WEEE Directive

### 2.2.1. Principles and goals

The first version of the WEEE Directive had as objectives: to (I) prevent the generation of e-waste; (II) reduce the amount of e-waste by means of reuse, recycling, and other forms of recovery; and (III) improve the environmental performance of all operators involved in the life cycle of electrical and electronic equipment (e.g. producers, distributors, consumers and in particular those operators directly involved in the treatment of WEEE) (European Commission, 2002).

The principle of Extended Producer Responsibility (EPR) is the core of the WEEE Directive. It sets strict obligations for producers (including manufacturers, distributors, and importers) to take operational or financial responsibility for the end-of-life phase of their products. In 70% of the EU countries, for the management of household WEEE, the EEE producers can fulfill their obligation either by an individual or collective systems known as compliance schemes, take-back schemes or producer responsibility organization, as presented in Figure 2-2.

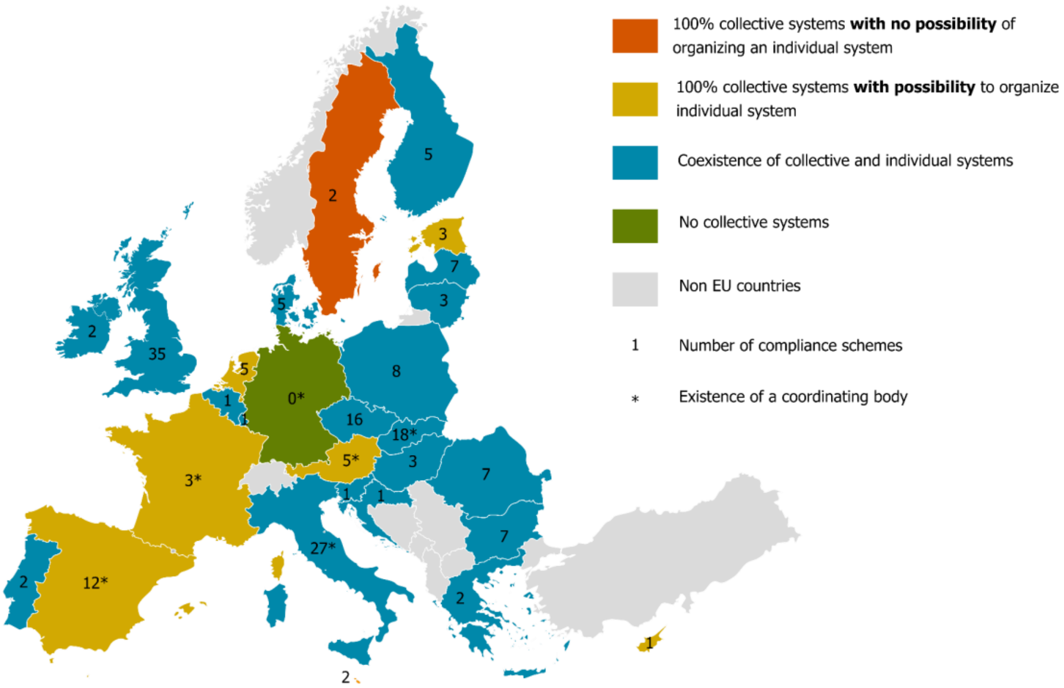


Figure 2-2. Overview of the European WEEE organization.
Source: adapted from Hestin et al., 2016; Horta Arduin et al., 2019.

Compliance schemes, on behalf of their members, are the management bodies in charge of organizing the collection, recovery treatment and disposal of e-waste. Despite the changes in the patterns of collection and disposal brought about by the WEEE Directive, in December 2008, the EC proposed a revision of the Directive in order to tackle the rapid increase of WEEE.

The WEEE Directive 2012/19/EU entered into force on August 2012 and became effective on February 2014. The WEEE Directive (recast) aims to protect the environment and human health by preventing or reducing the adverse impacts of the generation and management of WEEE and by reducing overall impacts of resource use and improving the efficiency of such use (European Commission, 2012). Table 2-1 presents the WEEE categories according to WEEE Directive recast:

Table 2-1. E-waste categories according to WEEE Directive.

| **WEEE categories From August 12, 2012 to August 14, 2018** | **WEEE categories From August 15, 2018** |
|---|---|
| 1. Large household appliances | I. Temperature exchange equipment |
| 2. Small household appliances | II. Screens, monitors, and equipment containing screens having a surface greater than 100 $cm^2$ |
| 3. IT Equipment and telecommunications | III. Lamps |
| 4. Consumer equipment | IV. Large equipment (any external dimension more than 50 cm) |
| 5. Lighting equipment | V. Small equipment (no external dimension more than 50 cm) |
| 6. Electrical and electronic tools | VI. Small IT and telecommunication equipment (no external dimension more than 50 cm) |
| 7. Toys, leisure and sports | |
| 8. Medical devices | |
| 9. Monitoring and control instruments | |
| 10. Automatic dispensers | |

Source: adapted from European Commission, 2012.

Considering the heterogeneity of (W)EEE, the categories aim to regroup the different types of devices. They are also used to set different targets considering the treatment specificity of each category of WEEE.

The major changes presented by the WEEE Directive recast were (Ibanescu et al., 2018):

- revision of WEEE categories from 2018;
- presentation of an alternative method for calculating collection rate based on the waste generated;
- introduction of new collection and recovery targets;
- clarification of the dual character of equipment used both in and outside private households.

### 2.2.2. Indicators and targets

The WEEE Directive (recast) establishes three weight-based indicators to monitor WEEE systems efficiency: **collection rate**, **recycling and preparation for reuse rate**, and **recovery rate**. The scope defined by the Directive is focused on the treatment performance of the e-waste collected by the compliance schemes.

**Collection Rate**

The revision of the collection rate target and its calculation are among the main changes in the WEEE Directive recast. The first WEEE Directive required Member States to collect at least 4kg per capita of WEEE from households. From 2016 to 2018, the minimum collection rate was 45% calculated based on the total weight of WEEE collected, expressed as the percentage of the overall weight of EEE placed on the market (POM) in the three preceding years. From 2019, collection rate can be calculated based on volumes of EEE POM, or WEEE generated (Favot et al., 2016). By 2019, the minimum collection rate will be 65% of the EEE POM, or 85% of WEEE generated (European Commission, 2012).

The POM approach does not consider the life span of the devices (most EEE have a life span longer than three years). Moreover, especially for certain categories of equipment that change significantly due to new technologies (e.g. mobile phones, TVs and computers), the e-waste collected is very different from the more recent devices placed on the market, in terms both of weight (e.g. cathode ray tubes are heavier than flat panel displays) and composition. Consequently, once it is a weight-based indicator, a high collection rate result does not reflect necessarily that the collection schemes are collecting many devices in terms of pieces.

The WEEE generated approach considers the amount of waste leaving the stock once discarded, taking into account the life span of electronic equipment (Huisman et al., 2017). It includes waste collected by the official compliance schemes, as well as those captured by complementary flows. As presented by Wang (2014), different methods for quantifying e-waste generation are discussed in the current e-waste research and literature. In order to have a common methodology for calculating WEEE generated in the EU, the EC published the WEEE Calculation Tool[5] (based on Magalini et al., 2015). This methodology is detailed in **Chapter 4** and is adopted in this thesis.

So far, the POM metric is the conventional approach to communicate and compare the collection rate between the Member State (EUROSTAT, 2017), together with the collection of e-waste per capita (Vidal-Legaz et al., 2018). Figure 2-3 presents the collection results in the EU Member States in 2016 based on data from Eurostat[6]. The collection rate is calculated as the ratio of the amount of collected WEEE in 2016 in relation to the average amount of EEE placed on the market in the three preceding years (2013-2015).

[5] WEEE Calculation Tool is available in: http://ec.europa.eu/environment/waste/weee/data_en.htm

[6] Eurostat is the statistical office of the European Union. It provides statistics at European level that enable comparisons between countries and regions in different topics, among which WEEE. Information about WEEE is available in: https://ec.europa.eu/eurostat/statistics-explained/index.php/Waste_statistics_-_electrical_and_electronic_equipment

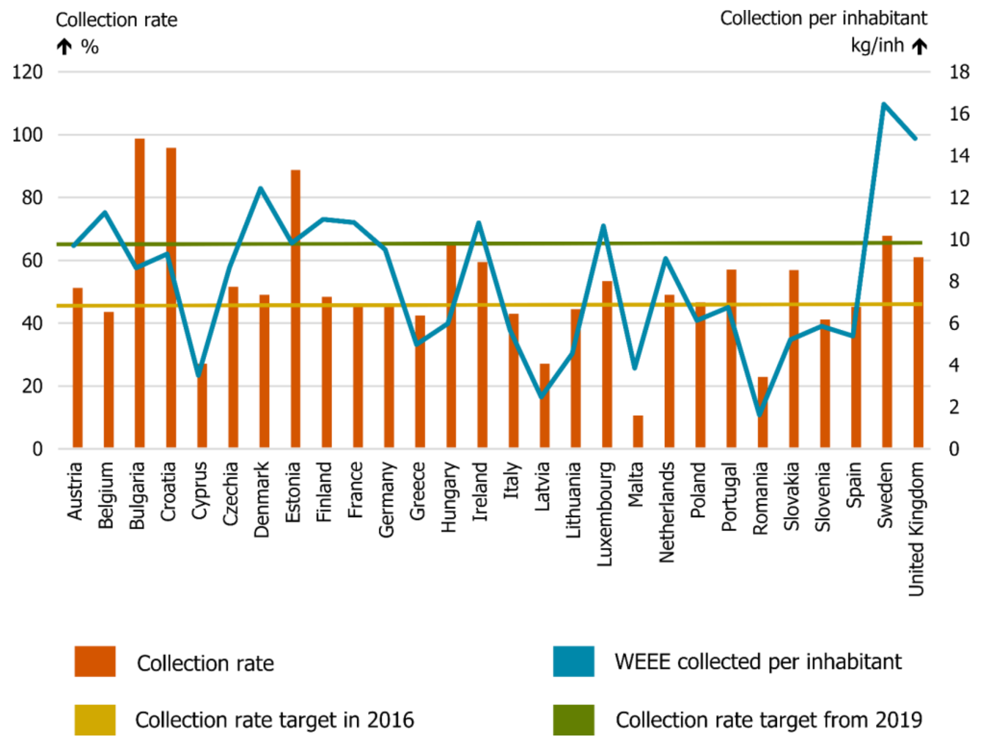


Figure 2-3. Collection rate and e-waste collected per capita in the EU Member States in 2016. Source: based on data from Eurostat, 2018.

In 2016, most of EU countries complied with the 45% collection target. Factors such as sorting errors, plundering, alternative systems of collection and treatment are some of the reasons for WEEE to be diverted from WEEE official collection schemes. As it can be noticed in Figure 2-3, there are significant differences between Member States performance. These variations are explained by the different development stage of the collection schemes across the Member States, one-off collection campaigns performed during the reference year used in the calculations, and also for lack of harmonization across the Member States in the methodologies to retrieve or determine POM information. For example, Bulgaria had a high collection rate in 2016 due to many additional collection campaigns organized, but this result is expected to decrease in subsequent years (Eurostat, 2018).

It is important to remark that the actual collection target is based on the overall WEEE collected. In 2015 the EC performed a study (Magalini et al., 2015) that, among others, assessed the possibility of setting individual collection targets for one or more WEEE categories. The EC concluded that, at that stage, it was inappropriate to set collection targets per WEEE category. According to the study, while setting individual collection targets may bring some economic, environmental and social benefits, it is difficult to draw conclusions about the feasibility of setting such targets at EU level (European Commission, 2017). Moreover, based on the performance of the last years, the 2019 target will probably not be achieved by most of the EU countries.

### Recycling (and preparation for reuse) Rate

From 2015 the recycling rate includes preparation for reuse. The WEEE Directive establishes that the recycling and preparation for reuse rate should be calculated, for each WEEE category, by dividing the overall weight that enters the recycling or the preparation for reuse facility by the overall weight of e-waste collected, expressed as a percentage. Thus, neither the flows not collected by the official schemes, nor the recycling losses are included in the scope.

Although the waste management hierarchy indicates waste reduction and reuse as the preferred options (Manfredi and Goralczyk, 2013), recycling is the main treatment in Europe (for further details see section 4.4). Waste recycling has an economic interest in terms of material recovery, unlike reuse, which represents a (small but real) decrease in market share for producers (Merot, 2015). Still, this develops local jobs and the social and solidarity economy (Defalvard and Deniard, 2016). Nowadays reuse is still modest, and not well documented by the official schemes. Thus, reporting recycling and reuse in the same indicator, besides not allowing a clear overview of treatment performance separately, does not support an increase in reuse practices.

### Recovery Rate

As described in section 1.5 (**Chapter 1**), recovery includes any operation where waste is used to replace other materials (e.g. use as fuel or other means to generate energy, as secondary raw materials, among others). The recovery rate is calculated by dividing the weight of the WEEE that enters the recovery facility, by the weight of all separately collected WEEE for each category, expressed as a percentage.

### Treatment targets fixed by the WEEE Directive

Table 2-2 and Table 2-3 present the recycling and reuse, and the recovery targets, respectively, for the former categories from 13 August 2012 to 14 August 2018, and for the new categories in force since 15 August 2018.

Table 2-2. WEEE Directive recovery target per category from August 2012 to August 2018.

| WEEE categories | Targets from 13 August 2012 until 14 August 2015 | Targets from 15 August 2015 until 14 August 2018 |
|---|---|---|
| 1. Large household appliances | • 80 % shall be recovered, and<br>• 75 % shall be recycled; | • 85 % shall be recovered, and<br>• 80 % shall be prepared for reuse and recycled; |
| 2. Small household appliances | • 70 % shall be recovered, and<br>• 50 % shall be recycled; | • 75 % shall be recovered, and<br>• 55 % shall be prepared for reuse and recycled; |
| 3. IT Equipment and telecommunications | • 75 % shall be recovered, and<br>• 65 % shall be recycled; | • 80 % shall be recovered, and<br>• 70 % shall be prepared for reuse and recycled; |
| 4. Consumer equipment | • 75 % shall be recovered, and<br>• 65 % shall be recycled; | • 80 % shall be recovered, and<br>• 70 % shall be prepared for reuse and recycled; |
| 5. Lighting equipment | • 70 % shall be recovered, and<br>• 50 % shall be recycled; | • 75 % shall be recovered, and<br>• 55 % shall be prepared for reuse and recycled; |
| 6. Electrical and electronic tools | • 70 % shall be recovered, and<br>• 50 % shall be recycled; | • 75 % shall be recovered, and<br>• 55 % shall be prepared for reuse and recycled; |
| 7. Toys, leisure and sports | • 70 % shall be recovered, and<br>• 50 % shall be recycled; | • 75 % shall be recovered, and<br>• 55 % shall be prepared for reuse and recycled; |
| 8. Medical devices | • 70 % shall be recovered, and<br>• 50 % shall be recycled; | • 75 % shall be recovered, and<br>• 55 % shall be prepared for reuse and recycled; |
| 9. Medical devices | • 70 % shall be recovered, and<br>• 50 % shall be recycled; | • 75 % shall be recovered, and<br>• 55 % shall be prepared for reuse and recycled; |
| 10. Monitoring instruments and control | • 70 % shall be recovered, and<br>• 50 % shall be recycled; | • 75 % shall be recovered, and<br>• 55 % shall be prepared for reuse and recycled; |
| 11. Automatic dispensers | • 80 % shall be recovered, and<br>• 75 % shall be recycled; | • 85 % shall be recovered, and<br>• 80 % shall be prepared for reuse and recycled. |

Source: adapted from European Commission, 2012.

Table 2-3. WEEE Directive recovery target per category from August 2018.

| WEEE categories | Targets from August 15, 2018 |
|---|---|
| I. Temperature exchange equipment | • 85 % shall be recovered, and<br>• 80 % shall be prepared for reuse and recycled; |
| II. Screens, monitors, and equipment containing screens having a surface greater than 100 $cm^2$ | • 80 % shall be recovered, and<br>• 70 % shall be prepared for reuse and recycled; |
| III. Lamps | • 80 % shall be recycled; |
| IV. Large equipment (any external dimension more than 50 cm) | • 85 % shall be recovered, and<br>• 80 % shall be prepared for reuse and recycled; |
| V. Small equipment (no external dimension more than 50 cm) | • 75 % shall be recovered, and<br>• 55 % shall be prepared for reuse and recycled; |
| VI. Small IT and telecommunication equipment (no external dimension more than 50 cm) | • 75 % shall be recovered, and<br>• 55 % shall be prepared for reuse and recycled. |

Source: adapted from European Commission, 2012.

As illustrated in Figure 2-4, for most of the (W)EEE, the recovery and recycling targets from 2019 do not represent a significant increase compared to the recovery targets in force until 2018.

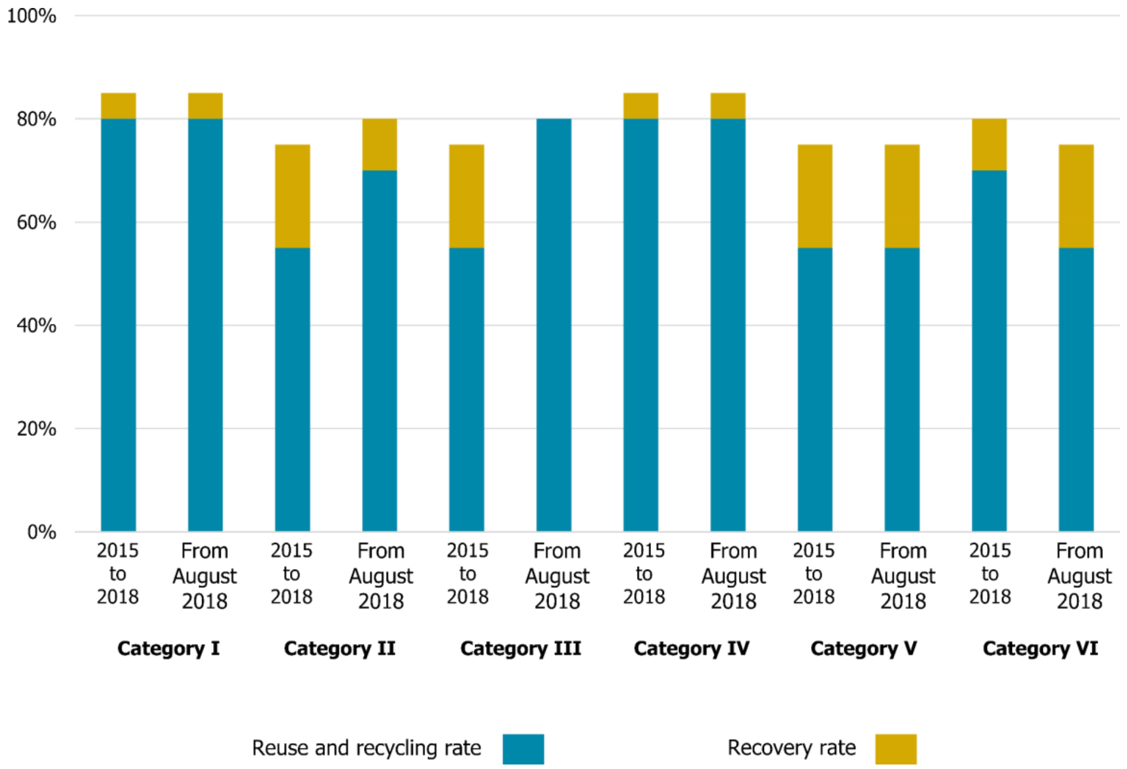


Figure 2-4. Targets evolution in the recent years.

For example, the targets for large equipment (formerly classified as category 1, and currently category IV) and small equipment (formerly classified as category 2, and currently category V) remained the same. From 2018, lamps (formerly classified as category 5, and currently category III) have a target for recycling that is only 5% higher than the previous recycling and reuse target. Information Technology (IT) devices (formerly comprised in categories 3 and 4) are now divided into two categories: screens (category II) and small IT (category VI) with different recovery and recycling targets.

### 2.2.3. Open Scope

According to the WEEE Directive, from 15 August 2018, the "open scope" is applicable, enlarging the list of different equipment managed by the official schemes. Now, any equipment which falls under definition of EEE (described in Article 3 and presented in the terminology and definition in **Chapter 1**), must be treated by the official schemes, except for those clearly mentioned explicitly in Article 2, sections 3 and 4 (e.g. equipment used for military purposes and equipment designed to be sent into space).

The open scope is a turning point in the WEEE chain, since some products have been affected by the regulation and must now comply with the official schemes (Ecologic, 2018). In its last public report, ADEME identified four new families of equipment concerned by the open scope (Deprouw et al., 2018):

- Household lighting: the support where the lamp (bulb) is fixed. Previously, only the lamps and professional lighting were included in category 5 (lighting equipment);
- Printer cartridges: previously managed under another agreement, cartridges containing electrical parts and requiring electrical power to operate are now considered as EEE;
- Generator sets: combination of an electrical generator and an engine mounted that produces electrical power;
- Electrical equipment: sockets, switches, circuit breakers, etc.

The open scope may also help engaging producers of new products (e.g. electronic cigarettes) that were not complying with their EPR obligations (Monier et al., 2014).

## 2.3 Transposition and implementation in France

The French implementation process of the WEEE Directive was combined with the RoHS Directive after an extensive process of consultation which engaged several stakeholders. WEEE Directive (recast) was transposed by the French Decree 2014-928 (articles R. 543-172 to R. 543-206 of the Environmental Code). It complements the Decree 2005-829 of 20 July 2005 transposing Directive 2002/96/EC, and Decree 2012-617 of 2 May 2012.

### 2.3.1 WEEE policy and legislation

Following the premises of the European WEEE Directive, the system created in France is based on the principle of EPR, and the responsibilities assigned to EEE producers led to the creation of compliance schemes. Since the first transposition of the WEEE Directive, e-waste system management in France has been organized in two flows: household and professional equipment. The French EOL chain has been operational since 2005 for professional e-waste, and since 2006 for the household e-waste.

Subsequent to the changes in WEEE Directive recast, the definition of household equipment was also updated in the French Environmental Code (article R543-173). Household e-waste is defined as *"WEEE from households, and commercial, industrial, institutional and other origins, that because of their nature and quantity, are similar to those of households. Equipment that was likely to be used by both household and other types of users are considered as household".*

Every five years the compliance schemes must be approved by the Ministry of Environment. The first agreement period was from 2005 to 2009. To be approved, the compliance schemes must demonstrate that they have the technical and financial capacity to meet the requirements of specifications included in the Ministerial Orders for household and professional WEEE (JORF, 2015b, 2014).

In 2019, three compliance schemes have been approved by the public authorities for the management of WEEE in France, as presented in Table 2-4 for household and professional WEEE. They operate in a free market to provide a service of collection and treatment, under the supervision of a coordinating center (called OCAD3E and further explained in section 2.3.2).

Although the compliance schemes in France are non-profit organizations, there is evident competition between those that are approved by the government to treat similar WEEE categories. In terms of market share, ESR is nowadays the biggest compliance scheme in France. However, the market share between the approved compliance schemes changes over the years and depends on the type of (W)EEE assessed (household and professional) as well as the e-waste categories. Figure 2-5 presents the market share of the compliance schemes in France from 2012 to 2018 based on the overall weight of EEE placed on the market. As it can be noticed, in the previous years, other compliance schemes than the three current compliances schemes operated in France (e.g. ERP and Recydent).

Moreover, when comparing the market share of the different compliance schemes and their scope of work over the years, we can identify a tendency of having fewer compliance schemes working with a more significant number of WEEE categories. For example, in 2012, Recydent was approved by the government to treat equipment of the dental sector included in the former categories 6 and 8. From 2013, its actives were absorbed by Recylum. In turn, from 2018, Recylum and Eco-Systèmes merged and became ESR.

Table 2-4. Compliance schemes currently approved for household and professional WEEE in France.

| WEEE categories | Household WEEE | Professional WEEE |
|---|---|---|
| I. Temperature exchange equipment | • Ecologic<br>• ESR (formerly Eco-Systèmes and currently changing to Ecosystem) | • Ecologic<br>• ESR (formerly Récylum and currently changing to Ecosystem) |
| II. Screens, monitors, and equipment containing screens having a surface greater than 100 cm$^2$ | • Ecologic<br>• ESR | • Ecologic |
| III. Lamps | • ESR (formerly Récylum and in and currently changing to Ecosystem) | • ESR |
| IV. Large equipment (any external dimension more than 50 cm) | • Ecologic<br>• ESR | • Ecologic<br>• ESR |
| V. Small equipment (no external dimension more than 50 cm) | • Ecologic<br>• ESR | • Ecologic<br>• ESR |
| VI. Small IT and telecommunication equipment (no external dimension more than 50 cm) | • Ecologic<br>• ESR | • Ecologic |
| VII. Photovoltaic panels | • PV Cycle | |

Source: adapted from Deprouw et al., 2018.

The French legislation adopted the WEEE categories according to Directive 2012/19/EU, and a new (W)EEE category applied to the photovoltaic panels (PV) was added to the European classification. In the former classification (11 categories), photovoltaic panels were classified as category 12, and at the current classification as category VII (see Table 2-4). In the last ADEME report (Deprouw et al., 2018), three categories were included in the monitoring of professional WEEE:

- Category 12: installation equipment for the low-voltage electrical power and communication network;
- Category 13: energy storage and conversion production equipment;
- Category 14: professional printing cartridges.

Regarding the collection targets, the French legislation has adopted the same targets fixed by the WEEE Directive (article R543-172-2 of the Environmental Code), and the calculation based on the average of EEE POM in the three previous years. In 2018, the collection rate in France was 49%, thus in compliance with the European target. Nevertheless, seeing the results of the past years, it will be a greater challenge to achieve the target from 2019 (65%).

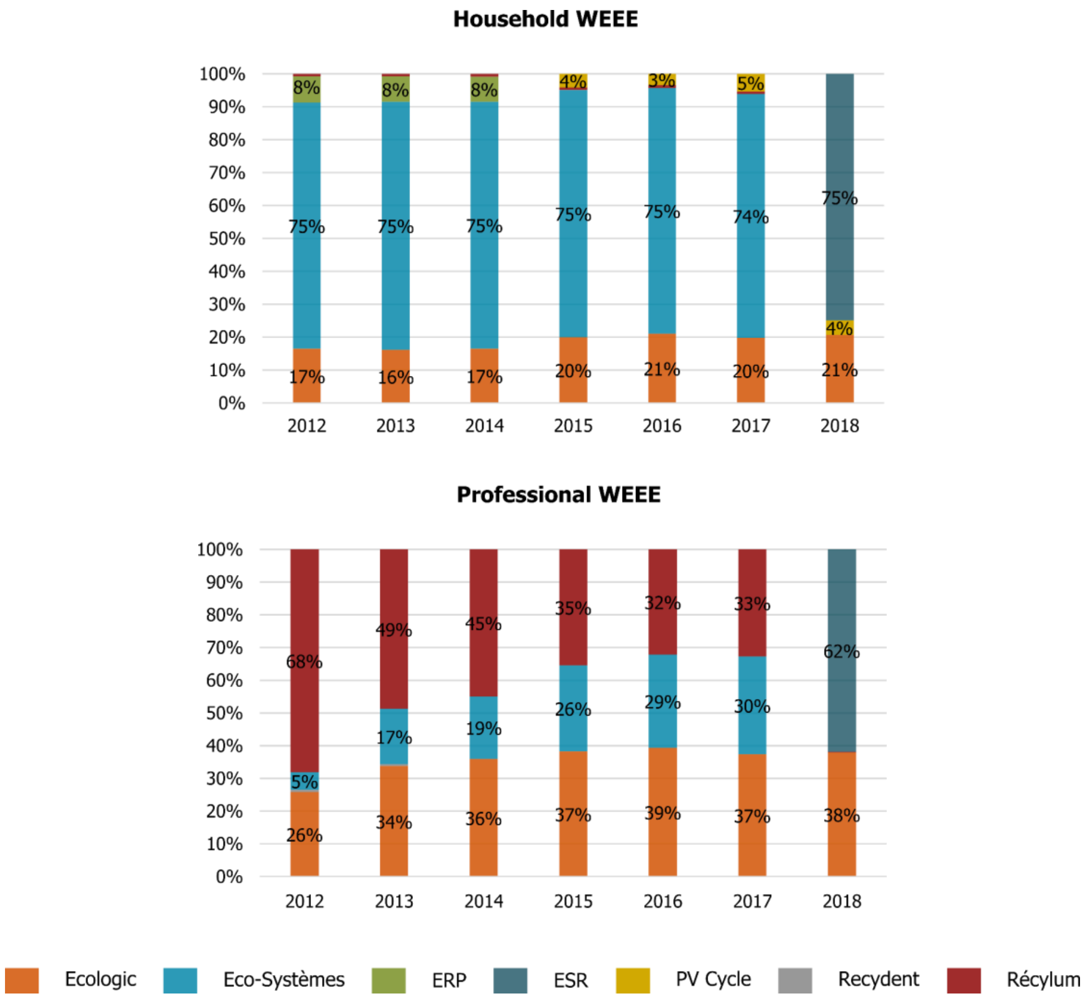


Figure 2-5. Market share of the approved compliance schemes in France (2012-2018).
Source : based on data from Fangeat, 2019.

In 2018, France has complied with the recovery targets set for the former 12 WEEE categories, as presented in Table 2-5. For this calculation, the CRT glass stored from 2014 to 2018 by ESR and disposed of in 2018 was considered to have been eliminated over the years.

From 2019, for some categories, the French legislation has set different household and professional recovery targets. According to French legislation, for both household and professional e-waste, category VI has a recovery rate 5% higher than the WEEE Directive target (80%) and a reuse and recycling rate 15% higher than the European target (70%). For household waste, the French legislation has set 85% recovery rate and 80% reuse and recycling rate for photovoltaic panels (category VII).

Table 2-5. Recovery performance per category in France (2018).

| WEEE categories | Indicators | Targets | French results in 2018 |
|---|---|---|---|
| 1. Large household appliances | Recovery rate | 85% | 90% |
| | Recycling and reuse rate | 80% | 80% |
| 2. Small household appliances | Recovery rate | 75% | 84% |
| | Recycling and reuse rate | 55% | 77% |
| 3. IT Equipment and telecommunications | Recovery rate | 80% | 80% |
| | Recycling and reuse rate | 70% | 73% |
| 4. Consumer equipment | Recovery rate | 80% | 89% |
| | Recycling and reuse rate | 70% | 80% |
| 5. Lighting equipment | Recovery rate | 75% | 91% |
| | Recycling and reuse rate | 55% | 86% |
| 6. Electrical and electronic tools | Recovery rate | 75% | 87% |
| | Recycling and reuse rate | 55% | 80% |
| 7. Toys, leisure and sports | Recovery rate | 75% | 84% |
| | Recycling and reuse rate | 55% | 77% |
| 8. Medical devices | Recovery rate | 75% | 87% |
| | Recycling and reuse rate | 55% | 82% |
| 9. Monitoring instruments and control | Recovery rate | 75% | 89% |
| | Recycling and reuse rate | 55% | 78% |
| 10. Automatic dispensers | Recovery rate | 85% | 92% |
| | Recycling and reuse rate | 80% | 90% |
| 11. Photovoltaic panels | Recovery rate | 80% | 97% |
| | Recycling and reuse rate | 70% | 90% |

Source: based on data from Fangeat, 2019.

Besides the indicators set by the WEEE Directive, according to the specifications included in the Ministerial Orders for household and professional WEEE, the French compliance schemes must report other performance indicators to ADEME and the Ministry of the Environment. The indicators applied to both household and professional WEEE are listed below (JORF, 2015b, 2014), and are further discussed in **Chapter 3**.

- Number of producers member of one of the compliance schemes approved by the government under Articles R.543-196 and R.543-197 of the Environmental Code;

- Market shares of the compliance schemes;
- Quantities of WEEE collected and treated by the compliance schemes;
- Reuse, depollution, recycling and recovery of WEEE;
- Employment and integration in the e-waste official chain;
- Environmental impacts in the context of the official WEEE chain;
- Indicators regarding the income and expenses of the compliance schemes.

### 2.3.2 WEEE official schemes

Figure 2-6 presents the French e-waste chain framework and the interactions between the different actors. For household WEEE, producers are responsible for the collection and treatment of the e-waste. They must fulfill these obligations either by joining one of the compliance schemes already approved by the authorities, or by setting up an individual one, which must also be approved by the authorities (no individual scheme has been approved to date).

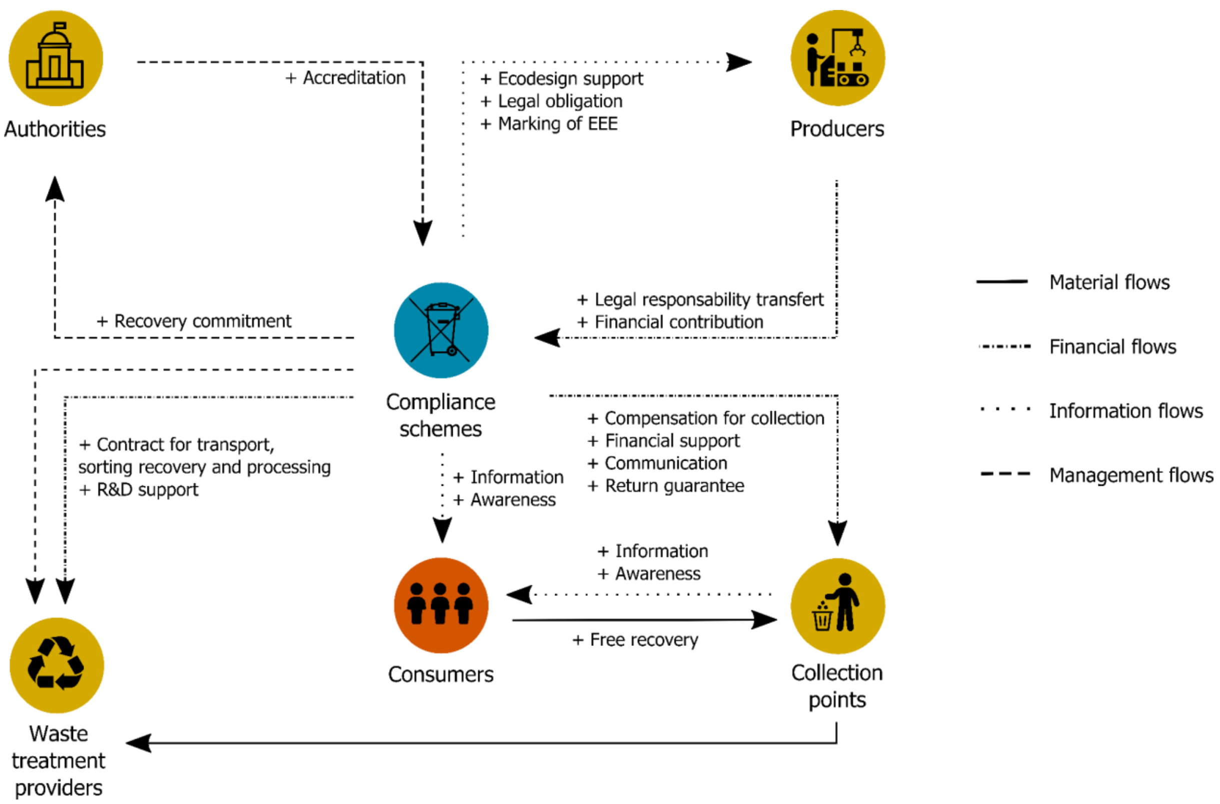


Figure 2-6. EPR framework in France.
Source: adapted from Deprouw et al., 2018; Horta Arduin et al., 2019.

Regarding professional WEEE, producers are responsible for end-of-life EEE placed on the market after 13 August 2005, or of older equipment in the case where it has been the object of replacement. Therefore, owners of professional EEE placed on the market before this date are responsible for its end-of-life. Similar to household equipment, the producer can either join one of the producer compliance schemes already approved by the authorities or implement an individual one. In 2017, 78% of the professional EEE placed on the market were under contract with one of the approved compliance schemes.

Any producer part of a compliance scheme must provide for the collection of WEEE and contribute to it by paying a financial contribution. Figure 2-7 presents the financial flows of the WEEE chain in France. Since the beginning, France chose the traditional clearing house model. This system is based on a visible fee which has been attributed to all EEE placed on the market since 2005. The fee is a representation of the cost of the treatment of WEEE and comes reflected in the sale prices of those products.

The municipalities receive compensation for their collection efforts by the French EPR compliance organizations. They are not involved in the EPR governance, although these authorities have to establish and organize household and hazardous waste management plans (Bahers and Kim, 2018).

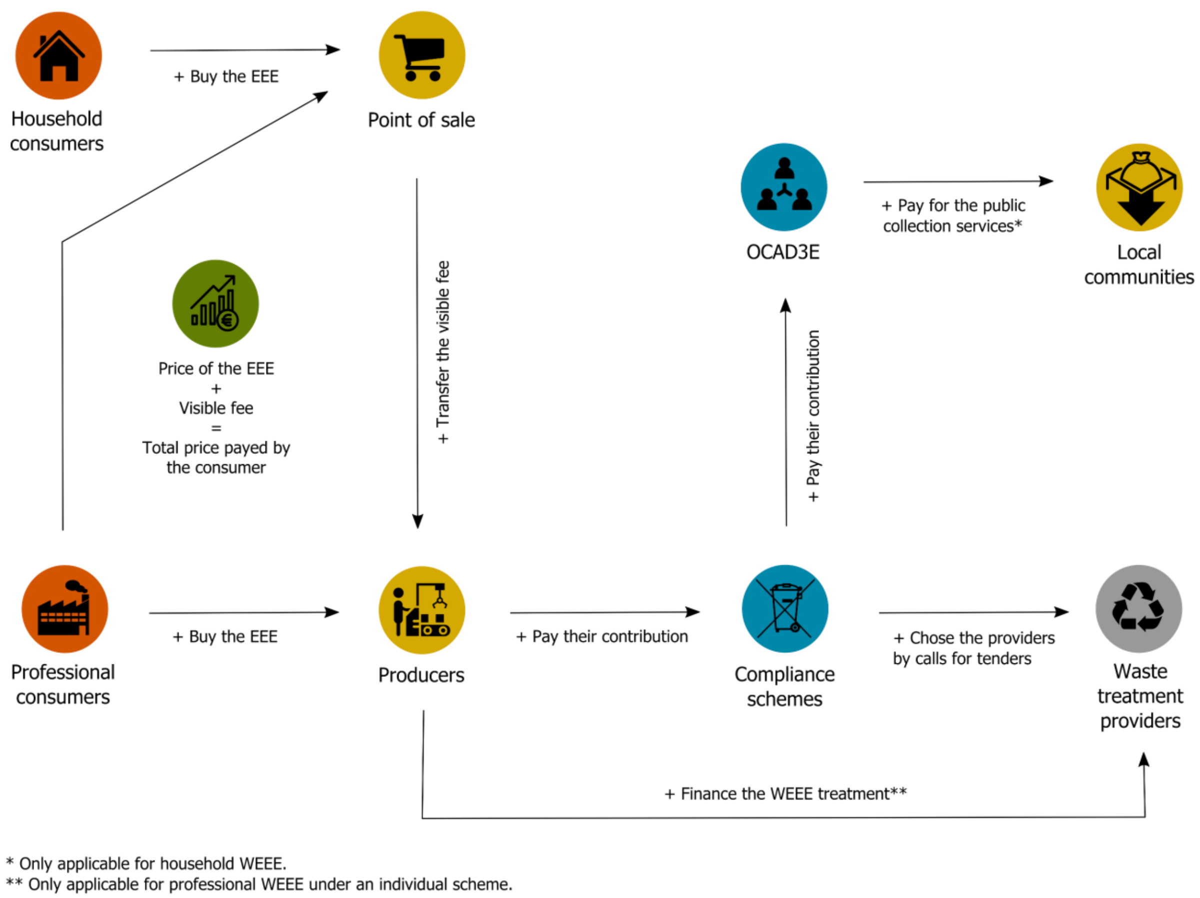


Figure 2-7. E-waste chain financial flows.

Source: data from Deprouw et al., 2018.

Since 2012, as part of a strategy to avoid free riders, waste treatment providers must have a contract with the compliance schemes. The compliance schemes choose waste treatment providers through calls for tenders to motivate competition, as well as to stimulate mutual monitoring.

Coordination and management of the interactions between the compliance schemes and local authorities is made by an association named OCAD3E (see section 1.1.1). One of the central missions of this clearing house is to place contracts with municipalities and to pay the financial support they offer for collection.

Since 2016 the obligation to have a contract with the compliance schemes was extended to transit operators, also known as brokers, and regrouping operators. The aim is to provide legal security for the traceability of hazardous waste when using private operators. Moreover, in order to discourage WEEE scavenging and support the official schemes, since 2011 the French public authorities have prohibited the payment in cash when purchasing metals (Micheaux, 2017).

In France, household WEEE collection and treatment is organized in six waste streams: large household cooling appliances, large household appliances non-cold (except for cooling appliances), screens, other small appliances, lamps and photovoltaic panels (included in 2015) (Movilla, 2016). Table 2-6 presents the correlation between the waste streams and the WEEE categories.

Table 2-6. E-waste streams in France.

| Waste streams | Former WEEE categories | Current WEEE categories |
|---|---|---|
| Large household cooling appliances | 1. Large household appliances | I. Temperature exchange equipment |
| Large household appliances non-cold (except for cooling appliances) | 2. Small household appliances | IV. Large equipment (any external dimension more than 50 cm) |
| Screens | 3. IT Equipment and telecommunications<br>4. Consumer equipment | II. Screens, monitors, and equipment containing screens having a surface greater than 100 $cm^2$ |
| Other small appliances | 6. Electrical and electronic tools<br>7.Toys, leisure and sports<br>8. Medical devices<br>9. Monitoring instruments and control<br>10. Automatic dispensers | VI. Small IT and telecommunication equipment (no external dimension more than 50 cm)<br>V. Small equipment (no external dimension more than 50 cm) |
| Lamps | 5. Lighting equipment | III. Lamps |
| Photovoltaic panels | 11. Photovoltaic panels | VII. Photovoltaic panels |

As it can be noticed, the categories in force since August 2018 are closer to waste streams adopted for WEEE collection and treatment in France. Every year, the French compliance schemes perform characterization campaigns to estimate the amount of equipment per category collected through each waste stream.

The compliance schemes have organized four collection channels for the household WEEE in France (Deprouw et al., 2018; Vadoudi et al., 2015):

- Separate collection set up with the support of the local authorities, e.g. municipal waste collection centers and door-to-door collection. It is the main collection channel, and in 2017, it represented 60% of the total mass of household e-waste collected;
- Retailers' collection points at stores, or the delivery, in exchange or not for a purchased product;
- Social enterprises that prepare the devices for reuse (e.g. associations, Emmaüs France, Envie Network);
- Other collections organized by the compliance schemes, e.g. in front of residences, mobile waste collection points, enterprises, and scrap dealers.

The collection of professional WEEE has more variable waste flows in comparison to household flows. Thus, it has led to the development of specific channels, such as on-site collection on request, adapted logistics, internet services, among other ones (Monier et al., 2016).

Except for the flows collected by social enterprises, that are usually prepared for reuse, the e-waste collected in the different channels is oriented towards the recycling chain. After collection, the WEEE is transported to a regrouping center in order to be weighed separately according to the waste streams previously mentioned, and then stored. After a certain volume is reached, the WEEE is transported to a treatment center.

## 2.4 Conclusions

The WEEE (Waste Electrical and Electronic Equipment) Directive, together with the RoHS Directive (Hazardous Substances) and the Basel Convention, were important milestones in e-waste management. Besides its importance in Europe, they have helped the development of a legal and technical framework in several countries.

The French e-waste take-back system has been developed according to the transposition of the WEEE and RoHS Directives into national law. Some choices made by the French legislators (e.g. legal provisions and targets for household and professional e-waste and addition of a new e-waste category for PV) have expanded and improved certain concepts and instruments. French choices involved focus on regulatory actions, the involvement of municipalities and operational efforts.

Since the initial stages of the national regulation, clear and steady policies have been adopted. This has led to a positive level of awareness and commitment from the stakeholders involved in the WEEE management system, which is reflected in the establishment of a clear set of roles and responsibilities, as well as a structure for the e-waste take-back.

The WEEE Directive targets a higher collection rate for e-waste in the coming years (65%) that may be challenging to achieve for most of Member States, including France. Therefore, to ensure an increase in collection, it is necessary to improve the knowledge and control of the WEEE flows. Thus, it is important to enlarge the scope of the indicators to monitor treatment losses and flows not collected by the official schemes and captured by complementary flows better. This thesis aims to provide elements to respond to this demand.

Taking a closer look at the goals of the WEEE Directive, it can be noticed an evolution between the first version and the recast that goes beyond the introduction of stepped collection and recovery targets. The goal of WEEE Directive recast includes the concepts of improvement of WEEE management and resources use. Nevertheless, once the current indicators were based on an overall weight-based approach, they only allow a limited overview of secondary raw materials recovery. Thus, to achieve the current targets, WEEE chain actors can focus their efforts on higher weight materials that have well-established recovery processes (e.g. iron, copper and aluminum). **Chapter 3** details other indicators identified on the ground (e.g. used by ADEME, Ecologic, etc.) as well as indicators suggested in the literature.

This PhD thesis aims to expand the current monitoring indicators from a perspective of circular economy and resource efficiency. The core of this proposition is the inclusion of other dimensions than simply technical performance, the expansion of the indicators' scope and on the shift from an overall weight-based approach to the monitoring of some target secondary raw materials. Indeed, due to the high value (in resources) of WEEE products, we want to foster the recycling of elements present in low or trace amounts, but that are important from economic and environmental point of view (e.g. precious metals as gold and palladium; and critical raw materials as cobalt and neodymium). Moreover, aiming to support the reuse of equipment and components, separate targets for recycling and reuse should be established. The following chapters present in detail the different proposals suggested in this work.

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## Chapter 3 | WEEE performance indicators: current approaches from the WEEE chain and literature

*"Há um certo gosto em pensar sozinho.*
*É ato individual, como nascer e morrer.*[7] *"*
Carlos Drummond de Andrade, Passeios na ilha:
divagações sobre a vida literária e outras matérias (1975)

### 3.1 Indicators to assess the performance of WEEE schemes

Performance indicators are widely used in many fields to assess and compare products, companies and scenarios. Therefore, authors of different fields have studied performance indicators from a theoretical perspective, as well as proposed new indicators or frameworks to integrate a set of indicators (Del-Río-Ortega et al., 2013). According to the set of criteria "CREAM", good performance indicators must be (Kusek and Rist, 2004):

- Clear: precise and unambiguous;
- Relevant: appropriate to the subject at hand;
- Economic: available at a reasonable cost;
- Adequate: provide a sufficient basis to assess performance;
- Monitorable: amenable to independent validation.

Indicators typically represent phenomena which are difficult to quantify, or for which units of measurement do not exist (Franklin-Johnson et al., 2016). Once they capture a complex reality into a single number, this entails simplifications and assumptions, which may link them to a specific context (Nelen et al., 2014). We identified in the literature other relevant aspects that should be considered when developing indicators (Van de Kerk & Manuel, 2008; Dahl, 2012; Cifrian et al., 2015):

- Indicators must be independent of each other and must not overlap;
- Data must be reliable, recent and regularly updated;
- Indicators must be easily understandable, in order to ensure ease of use;
- The calculation model must be clear and validated;
- The number of indicators must be limited;
- The set of indicators should provide a good overview of the current situation regarding sustainability and indicate the gap between the current situation and the ideal situation.

In the European Union's (EU) action plan for the circular economy, the implementation of a set of reliable indicators is identified as fundamental in order to assess progress towards a more circular economy and to assess the effectiveness of action at EU and national levels (European Commission, 2015).

[7] There is a certain pleasure in thinking alone. It's an individual act, like being born and dying.

In another report published by the European Commission (EC) on Extended Producer Responsibility (EPR) programs in force in the various Member States, one of the conclusions was the need for better data to improve performance monitoring and strategic decision-making (Monier et al., 2014). According to this report, performance indicators should be developed to address the concept of optimization, including different dimensions: environmental; economic; quality of processing operations; export monitoring, etc.

According to Forti et al., (2018), indicators arising from the measurement framework should capture the most essential aspects of a country's performance of e-waste management. As presented in the previous chapters, the WEEE chain already uses indicators to assess the system's performance (e.g. WEEE Directive indicators). This chapter presents an overview of indicators currently used in the EU and France to monitor WEEE performance (section 3.2), and indicators identified in the literature (section 3.3). The indicators' assessment is divided according to the following classification:

**Technical indicators**

Quantify the e-waste flows (e.g. waste generated, collected, recycled), or measure the effectiveness of a process or waste system (e.g. collection and recycling rate).

**Economic indicators**

Measure certain dimensions of economic activity, such as the costs and/or revenues of the WEEE chain (e.g. quantity collected in euros with the visible fee, and cost per tons of WEEE collected).

**Environmental indicators**

Assess the impacts on the environment due to e-waste management activities, and/or the environmental benefits of the WEEE chain. This includes the application of a method (e.g. life cycle assessment) to quantify potential environmental impacts, resulting in indicators such as global warming, resource depletion and human toxicity; and indicators based on life cycle inventory interpretation (e.g. energy consumption and emissions into the air). It also includes indicators related to the circularity and criticality of materials.

**Social indicators**

Describe the impact of the supply chain on human well-being, including on employees of the e-waste chain, and other stakeholders, such as the community.

## 3.2 Indicators currently used in the WEEE sector

In order to identify existing indicators to monitor the development of WEEE chain, national and European regulations were verified, as well as studies developed by WEEE treatment stakeholders. The following sources were considered: WEEE Directive 2012/19/EU (European Parliament, 2012), the specifications for the approval of household and professional compliance schemes in France (JORF, 2015, 2014), WEEE Forum website and public reports (WEEE Forum, 2016), internal and public reports from the French Environment & Energy Management Agency (ADEME) (e.g. Deprouw et al., 2018; Fangeat, 2019), Ecologic annual reports (e.g. Ecologic, 2018a, 2018b, 2018c), also two reports released by the European Commission (Monier et al., 2014; Vidal-Legaz et al., 2018).

Considering that the organization of the sector in France is divided into household and professional WEEE, at the national level the indicators used by the WEEE actors follow this classification. Most of the indicators identified aim to measure the technical performance of the sector. However, other dimensions such as economic, environmental and social dimensions have been identified. Table 3-1 and Table 3-2 summarize the indicators currently used by the WEEE chain.

The most recurrent indicators are those required by the WEEE Directive (collection rate, recycling and reuse rate, and recovery rate), or that contain intermediate information to calculate the indicators required by the regulation. For example, the quantities placed on the market, and the quantities of e-waste collected, are intermediate information for calculating the collection rate. The quantities of equipment placed on the market are commonly used for monitoring the supply chain, and the results are reported by category and/or total.

Regarding the quantities treated, results are expressed by waste stream, category, type of treatment (equipment reuse, components reuse, recycling, energy recovery or disposal), place of treatment (country), and/or, in the case of professional WEEE, by type of system (individual or compliance schemes). The composition of the WEEE treated (household), and its depollution (household and professional) are also national indicators.

It is important to remark that in France the treatment rates for household WEEE are quantified per waste stream. Consequently, there are no indicators monitoring collection and treatment per category. In the annual report released by ADEME, the WEEE Directive indicators (collection rate, recycling and reuse rate and recovery rate) are communicated per category and include both household and professional. To perform the breakdown from the waste streams into categories, ADEME uses data from the compliance schemes' characterization campaigns – which is not publicly disclosed. Other Member States also perform the collection and treatment of WEEE categories in different waste streams. For example, in Sweden, the household collection is based on four waste streams: fridges and freezers, electrical and electronic goods, large white goods and straight fluorescent tubes (Román, 2012).

The WEEE Forum monitors and systematizes key figures of WEEE chain performance in Europe but the data is available only to its members. On the WEEE Forum website there are some public reports[8], including a technical paper on the current recycling rate calculation according to the WEEE Directive. According to this document, the "weight of the input waste entering the final recycling

[8] Available in : https://weee-forum.org/publications-papers/

process" should not be considered equivalent to the "weight of municipal waste recycled" (WEEE Forum, 2016). They suggest the recycling rate is calculated according to EN 50625-1 (European Standardization Organizations, 2014).

Table 3-1. Technical indicators currently used to assess the WEEE chain performance.

| **Thematic Area** | **Type of WEEE** | **Indicator** | **Unit of measure** | **Source** |
|---|---|---|---|---|
| Follow-up of placed on the market | H/P | Number of producers members of the compliance schemes | Dimensionless | ADEME Ecologic |
| | H/P | Quantities of equipment placed on the market (total and per categories) | Dimensionless | ADEME Ecologic |
| | H/P | Market shares of compliance schemes (total and per categories) | Percentage | ADEME |
| Collection set-up | H | Number of agreements signed with local authorities | Dimensionless | ADEME |
| | H | Population covered by the collection system | Inhabitants | ADEME |
| | H/P | Number of collection points per type of collection channel | Dimensionless | ADEME |
| Collection | H/P | Quantity of WEEE collected (per waste stream, categories and collection channel) | Tons and kg/inhabitant* | ADEME Scoreboard* |
| | H | Quantity collected per department and collection channel | Tons and kg/inhabitant | ADEME |
| | H | Collection rate (total, per compliance scheme and per type of collection point) | Percentage | ADEME WEEE Directive |
| Treatment | H/P | Quantity treated (total, per waste stream, per category and per type of treatment) | Tons | ADEME |
| | P | Share of WEEE treated by compliance schemes and individual systems | Percentage | ADEME |
| | H/P | Quantity of WEEE recycled or prepared for reuse | kg/inhabitant | Scoreboard |
| | H/P | Recycling and reuse rate | Percentage | ADEME WEEE Directive |
| | H/P | Recovery rate (total, per waste stream) | Percentage | ADEME WEEE Directive Ecologic |
| | H/P | Quantities of products and substances extracted (total, per compliance scheme) | Tons | ADEME Ecologic |
| | H/P* | WEEE composition per type of fraction (ferrous, non-ferrous, plastics, hazardous, glass, PCB, shredder residues, mineral fraction, others) | Percentage | ADEME Ecologic* |
| | P | Quantities of used EEE exported (total and per categories) | Tons | ADEME |

H: Household; P: Professional; *: applicable only to the source or indicator highlighted with the asterisk.

WEEE is among the twenty-six sets of indicators proposed by the Raw Materials Scoreboard (Vidal-Legaz et al., 2018). The indicators are calculated based on the data available in Eurostat (Eurostat, 2018), and a comparison between the amount of WEEE collected and amount treated by the Member States, in kilos per inhabitant, is presented.

Regarding the economic indicators, the WEEE Directive does not propose indicators and targets. The only economic indicator available in the annual report published by ADEME is the quantity collected (in millions of euros) with the visible fee, an indicator also reported by Ecologic in its annual report. Nevertheless, based on data reported by the compliance schemes, ADEME monitors other economic indicators detailed in Table 3-2 that are not disclosed in the public report. Further information on the internal economic indicators monitored by ADEME is presented in Chapter 6.

Table 3-2. Economic indicators currently used to assess the WEEE chain performance.

| **Thematic Area** | **Type of WEEE** | **Indicator** | **Unit of measure** | **Source** |
|---|---|---|---|---|
| Revenues | H | Visible fee | k€ | ADEME Ecologic |
| | H/P | Fractions selling | k€ | ADEME** |
| | H/P | Products | k€ | ADEME** |
| | H/P | Other revenues | k€ | ADEME** |
| | H/P | Total revenues | k€ | ADEME** |
| Expenses | H/P | Operational costs | k€ | ADEME** |
| | H/P | Net operational costs | k€ | ADEME** |
| | H/P | Support to collection actors (except communication) | k€ | ADEME** |
| | H/P | Communication | k€ | ADEME** |
| | H/P | R&D | k€ | ADEME** |
| | H/P | Provisions | k€ | ADEME** |
| | H/P | Operational fees | k€ | ADEME** |
| | H/P | Taxes | k€ | ADEME** |
| | H/P | Total expenses | k€ | ADEME** |
| Cost-effectiveness of the EPR | H/P | Total fees of the system | k€ | Monier et al., 2014 |
| | H/P | Fees per EEE POM | €/tons | |
| | H/P | Fees per EEE collected | €/tons | |
| | H/P | Fees per inhabitant per year | €/inhabitants | |

H: Household; P: Professional; **: available in ADEME internal report.

In the European Commission report for EPR guidance, we identified four cost-effectiveness indicators that are presented in Table 3-2 (Monier et al., 2014). As discussed in Chapter 2, according to the specifications for the accreditation of household and professional compliance schemes, environmental, and social indicators must be annually reported, besides the technical and economic indicators. However, neither the indicators and nor the methods are specified. The other references consulted do not include environmental and social indicators.

This work aims to provide a dashboard with the relevant indicators for ADEME and Ecologic to monitor and report WEEE chain performance. Thus, in addition to the indicators aforementioned, with

Ecologic and ADEME we identified four aspects that could be monitored by indicators in order to improve information on the WEEE chain performance:

- EEE not officially declared as placed on the market (grey market);
- average lifetime of EEE;
- avoided disposal i.e. e-waste not sent for disposal because it is recovered by the official WEEE chain.
- number of complaints related to theft of WEEE in waste collection centers.

## 3.3 Indicators published in the literature

Table 3-3 presents the keywords selected to perform the literature review on indicators for monitoring EoL chain performance. The search was carried out in different databases such as Science Direct, Scopus, Google Scholar, etc.

After reading the summaries of the articles and reports, we selected only those whose field of study was the development and/or application of indicators to analyze waste management chain or systems. Studies on the development of sustainability indicators to analyze and compare the overall performance of different countries (not specifically on waste) and those that propose indicators to analyze the design and/or use of products were not included in the analysis.

Table 3-3. Review of performance indicators for WEEE chain: selected keywords.

| **English** | **French** |
|---|---|
| Indicators | *Indicateurs* |
| Waste electric and electronic equipment | *Déchets d'équipements électriques et électroniques* |
| WEEE | *DEEE* |
| Performance assessment | *Évaluation de la performance* |

Although the initial research focused on indicators applied to the WEEE sector, some studies related to other waste streams, such as household waste, were not excluded from the assessment because they present interesting indicators that can be applied to other waste streams (e.g. Ak & Braida, 2015; Cifrian, Andres, & Viguri, 2015; Rigamonti, Sterpi, & Grosso, 2016). Thirty-two studies were selected (see Table 3-4), totaling more than 250 indicators.

Table 3-4. Type of indicators used by the studies identified in the literature.

| Studies | Type of indicators | | | |
|---|---|---|---|---|
| | Technical | Economic | Environmental | Social |
| Forti et al., 2018 | X | | | |
| De Oliveira Neto et al., 2017 | | X | X | |
| Parajuly et al., 2017 | X | | | |
| Baxter et al., 2016 | | | X | |
| Căilean and Teodosiu, 2016 | X | | X | |
| Favot et al., 2016 | X | X | | |
| Franklin-Johnson et al., 2016 | X | | | |
| Fornasiero et al., 2016 | | X | X | |
| Haupt et al., 2016 | X | | | |
| Li et al., 2016 | X | X | | |
| Morris and Metternicht, 2016 | X | X | X | X |
| Rigamonti et al., 2016 | | X | X | |
| Sinha et al., 2016 | X | | | |
| Souza et al., 2016 | | X | | X |
| Van Eygen et al., 2016 | X | | X | |
| Zeng and Li, 2016 | X | | | |
| Souza et al., 2015 | | X | | X |
| Ak and Braida, 2015 | X | X | X | X |
| Barletta et al., 2015 | X | X | X | X |
| Cifrian et al., 2015 | X | X | | X |
| Cucchiella et al., 2015 | X | X | | |
| Umair et al., 2015 | | | | X |
| Wang, 2014 | X | X | X | |
| Lu et al., 2014 | | | X | X |
| Nelen et al., 2014 | X | X | X | |
| Manfredi and Goralczyk, 2013 | | | X | |
| Gossart and Huisman, 2011 | X | X | | X |
| Wen et al., 2009 | X | | | |
| Gregory and Kirchain, 2008 | | X | | |
| Widmer et al., 2005 | X | X | X | X |
| Sinha-Khetiwal et al., 2005 | X | | X | X |
| Huisman, 2003 | X | X | X | |

In order to facilitate the analysis and comparison of the proposed indicators, the following aspects were assessed in each work:

- Overall objective of the indicators;

- Type of waste – e.g. WEEE, municipal solid waste (MSW);
- Type of indicator proposed: qualitative or quantitative;
- Country or region of the study;
- Dimension analyzed by the indicator: technical, economic, social and/or environmental;
- Method and calculation scope.

As illustrated in Figure 3-1, most of the studies assessed one or more technical indicators (69%). More than 80% of the indicators are quantitative, and most articles proposed between 1 and 5 indicators (56%).

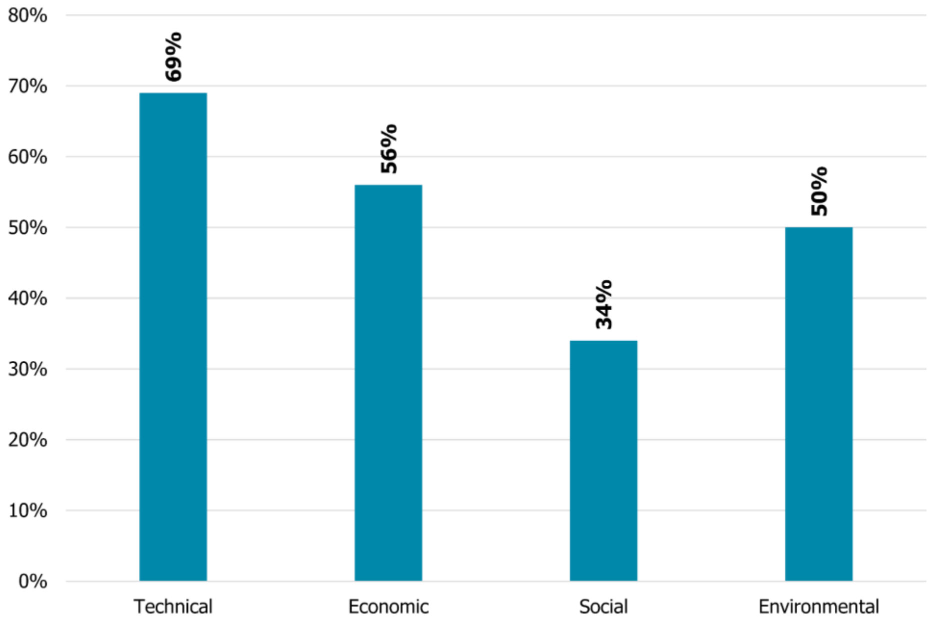


Figure 3-1. Analysis dimension of the indicators identified in the literature.

Globally, the studies developed and/or applied indicators to analyze the performance of waste management. The use of indicators to measure the circularity of materials and to support decision-making in waste management are more recent applications (after 2014).

Except for studies that did not specifically identify their location, the indicators were developed to a specific country or region or were validated with a case study in one or more countries. Figure 3-2 summarizes the amount of studies per countries or regions. Some authors used the indicators to compare the waste management performance of different countries.

Before, or in parallel, to the development or application of the indicators, some authors used Material Flow Analysis (MFA) to quantify the flow of materials contained in the system under study. In order to obtain a single result on the performance of the system, some studies developed aggregation indices (e.g. Ak and Braida, 2015; Cifrian et al., 2015; Nelen et al., 2014). According to Gossart and Huisman (2011), having an aggregated indicator evaluating the performance of the solutions adopted in a given country to solve the e-waste problem could help engage citizens and policy-makers in this major challenge.

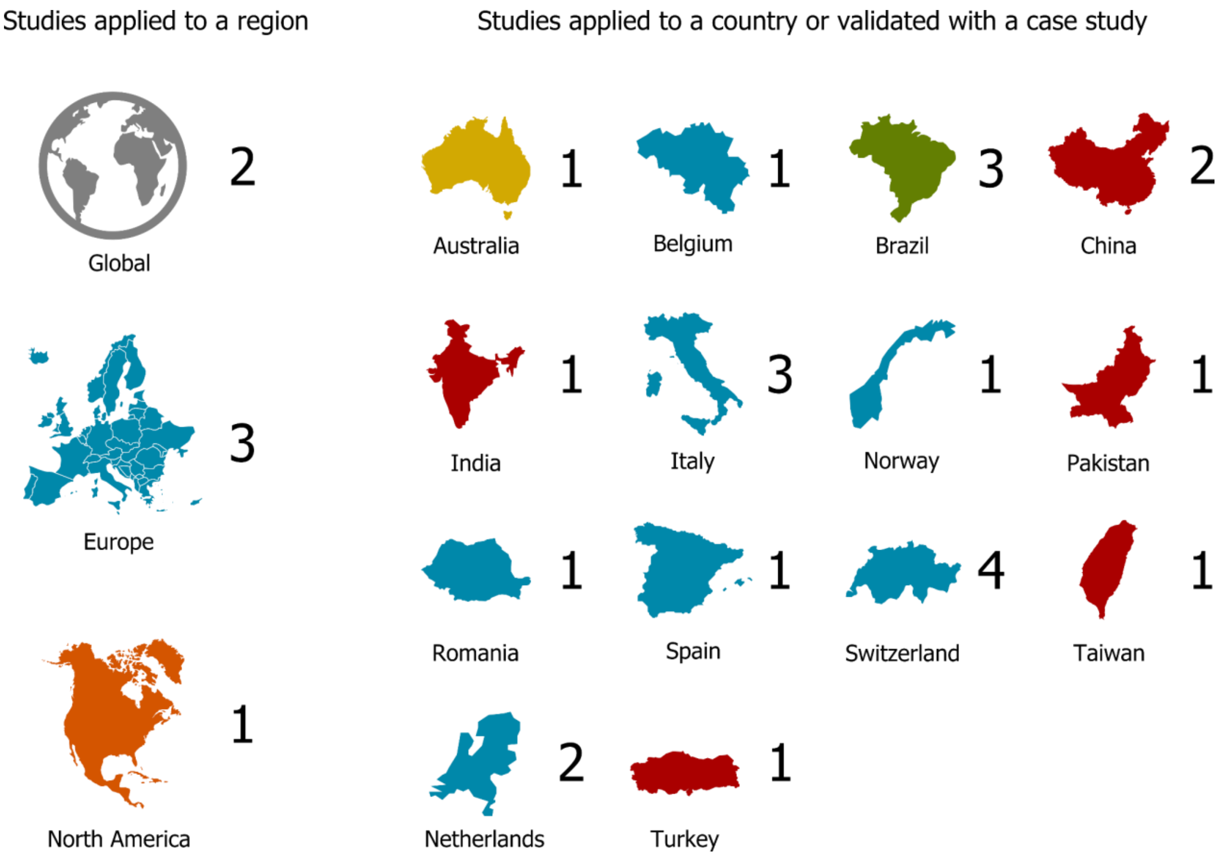


Figure 3-2. Origin of the studies identified in the literature.

**Technical indicators**

Most of the indicators are related to e-waste quantification (generation, collection and/or treatment), but some studies focused on the quantities of electrical and electronic equipment (EEE) placed on the market (Favot, Veit, & Massarutto, 2016; Parajuly et al., 2017), or its life span. Franklin-Johnson et al. (2016) developed the longevity indicator which seeks to quantify the length of time for which a material is retained in a product system including different types of treatment (reuse, recycling, etc.). Among other indicators for WEEE treatment in Australia, Morris et Metternicht (2016) proposed monitoring the increase/decrease in WEEE life span.

Aiming to assess and compare the performance of WEEE management in different countries, Forti et al., (2018) present the methodology and data required to calculate four technical indicators: total EEE placed on the market (kg/inh), total e-waste generated (kg/inh), e-waste formally collected (kg/inh) and collection rate (ratio between WEEE collected and generated, in percentage). Regarding the collection rate, other studies calculated it based on the ratio between the amount collected and average weight of EEE placed on the market in the preceding years (e.g. Favot et al., 2016; Wen et al., 2009).

Some studies calculated recycling rates according to the WEEE Directive (e.g. Wen et al., 2009), and others discussed the limitations of these indicators. According to Parajuly et al. (2017), the WEEE

Directive method to calculate recycling rate does not reflect the true material recycling rate as it excludes the complementary flows and the losses that occur in the WEEE treatment chain. Haupt et al (2016) stated that the current recycling rate hides improvement potentials in waste management besides not indicating how much secondary material becomes available.

Some authors presented new approaches to recycling rate indicators by considering the weights of materials actually recycled at the end of the process compared to those entering the treatment facility (Van Eygen et al., 2016). In addition, one study proposed that the recycling rate be calculated separately according to the type of recycling: open-loop recycling and closed-loop recycling (Haupt et al., 2016b). This proposal is grounded on the strong probability that in an open-loop recycling the recycled material will be used in a process with lower quality requirements (downcycling).

Some authors also discussed the overall weight-based targets fixed by the WEEE Directive. According to Van Eygen et al. (2016), to achieve the targets, this approach allows recyclers and compliance schemes to focus on materials that are present in large quantities in the waste stream (e.g. ferrous metals), rather than on critical materials.

Given the importance of e-waste actors in the system management, indicators to quantify them were also identified. These indicators can be regrouped in two groups: collection (e.g. number of collection points) and treatment (e.g. number of recycling sites and landfills). Souza et al. (2015, 2016) proposed indicators to quantify new companies in the WEEE sector, including those that offer innovative treatment processes.

Several articles proposed indicators to quantify waste, by typology and/or type of treatment (e.g. reused, recycled, incinerated and landfilled). WEEE quantification is done according to the total weight of waste, per waste stream, per inhabitant and/or category (Barletta et al., 2015; Parajuly et al., 2017; Rigamonti et al., 2016; Van Eygen et al., 2016). Usually, the scope of the assessment is limited to the official schemes. However, some authors proposed indicators to evaluate the quantities treated by complementary flows (Souza et al., 2016, 2015). Indicators for exported WEEE flows were also proposed (Gossart & Huisman, 2011; Morris & Metternicht, 2016). Seeing the potential of WEEE as a source of secondary raw materials, Parajuly et al. (2017) developed an indicator to quantify the secondary resources available in WEEE that can be recovered through recycling.

**Economic indicators**

The economic indicators can be classified into three groups: costs, profits, and economic efficiency (balance between costs and revenues). According to Wang (2014), economic indicators can account for market value disparities between different materials and support the identification of recycling priorities.

Rigamonti et al. (2016) assessed the ratio between the collection, treatment and disposal costs and the quantity of waste collected. Favot et al. (2016) proposed a similar indicator based on the ratio between the total costs and the WEEE treated. Other authors have proposed individual indicators to quantify the costs associated with each stage of WEEE treatment: collection, regrouping, processing, management, etc. (Gregory and Kirchain, 2008; Li et al., 2016; Lu et al., 2014; Souza et al., 2015).

Gregory and Kirchain (2008) proposed an indicator to quantify the revenues obtained with the visible fee. As previously mentioned, this indicator is currently declared by the compliance schemes and ADEME in France. Souza et al. (2015), proposed an indicator to estimate the increase in product prices due to the reverse logistics costs. Widmer et al. (2005), suggested an indicator to measure the financial support for ecodesign.

Some articles estimated the financial benefits of WEEE chain. De Oliveira Neto et al. (2017) estimated the investments made in infrastructure, machinery, labor force and logistics costs, to calculate the economic advantage of WEEE treatment. The costs of acquiring raw materials to manufacture polymer components for EEE are also estimated and compared to the cost of recycled polymers.

Cucchiella et al. (2015) performed an economic assessment of the potential revenues of fourteen electronic products on the base of current and future disposed volumes in Europe. The recovery economic potential is defined as the product between the WEEE's material composition (g/unit product) and the related market prices (€/kg).

Nelen et al. (2014) proposed an indicator to measure material cycle closure based on the ratio of the market price of the output fraction after shredding and sorting (e.g. ferrous metals and PCB) and the market price of the corresponding material used in the original application. Wang (2014) proposes the profit per input product (€/kg), and the efficiency of the waste treatment as indicators for evaluating the performance of e-waste treatment.

**Environmental indicators**

Among the studies that proposed environmental indicators, the majority used Life Cycle Assessment (LCA). Since most studies based on LCA considered different impact categories, they account for around 80% of the environmental indicators identified in the literature. A detailed assessment of the LCA studies of WEEE is presented in **Chapter 5**.

Huisman (2003) developed the quotes for environmentally weighted recyclability (QWERTY) approach focused on the determination of environmentally weighted recycling scores rather than weight-based recycling scores. It was the first study to assess WEEE treatment from an environmental point of view. The approach is based on the net environmental value recovered over the total environmental value of a product. The environmental values are derived from LCA studies.

Indicators based on calculation methods other than LCA can be classified into four groups: pollution, resources, material circularity and material criticality. The indicators related to criticality are presented in detail in **Chapter 7**.

Sinha et al. (2016) proposed two indicators related to the material circularity. The loop leakage indicator determines the resource fraction leaving the product system and indicates to what extent the loop is closed and metals are preserved in the system. The loop efficiency indicator determines how efficiently the resources are utilized in the system without being stocked.

### Social indicators

Most of the studies that included social indicators focused on WEEE chain employees, but some authors also proposed indicators to assess the impacts of WEEE in society. Some studies performed social life cycle assessment (S-LCA) for WEEE treatment formal and/or informal conditions. For example, the S-LCA indicators suggested by Lu et al. (2014) are number of work hours, employment, housing and education. Table 3-5 presents the impact categories and the corresponding indicators identified by Souza et al. (2015) in a study performed in Brazil.

Table 3-5. Social impact categories and indicators for WEEE treatment.

| Social impact categories | Suggested indicators |
|---|---|
| Social inclusion | Number of WEEE workers and relatives provided with social and psychological assistance; |
| Formal and informal employment and generation of income with opportunity to professional development | Number of new WEEE workers that come from such groups: women; informality; prison; slums; alcoholism; drugs addition; crime; physical and mental disabilities;<br>Number of WEEE workers (formal and informal) per occupation;<br>Average income of WEEE workers (formal and informal) per occupation;<br>Number of workers that undertook professional training and refresher courses; |
| Risks and working conditions | Number of WEEE workers (formal and informal) working in adequate conditions (equipment, protection, training);<br>Occurrence of job accidents and diseases directly related to risks of the WEEE chain; |
| Access to healthcare, education, environmental education and digital inclusion | Number of WEEE workers and their relatives provided with health insurance;<br>Number of WEEE workers and their relatives per level of education;<br>Number of individuals (workers, relatives and community) benefited by the WEEE chain with digital inclusion and environmental education. |

Source: Souza et al., 2015.

Other studies proposed indicators related to job creation by the sector and the employees' education level (Cifrian et al., 2015; Sinha-Khetiwal et al., 2005; Umair et al., 2015; Widmer et al., 2005). Concerning working conditions, the studies proposed indicators related to the number of work hours, the presence of forced labor and child labor, and job turnover (Barletta et al., 2015; Umair et al., 2015).

The employees' health, the number of accidents and average income are considered in different studies (Barletta et al., 2015; Sinha-Khetiwal et al., 2005; Souza et al., 2016, 2015; Umair et al., 2015;

Widmer et al., 2005). Regarding the incomes, none of the studies correlated the revenues with other social data such as gender.

Some studies used indicators related to the level of commitment, education and awareness of the population (Ak and Braida, 2015; Gossart and Huisman, 2011; Umair et al., 2015; Widmer et al., 2005). Widmer et al. (2015) also proposed an indicator to identify the existence of research and development for WEEE management and treatment.

### 3.4 Gaps identified and needs for further development

The indicators currently used to communicate the performance of the WEEE chain in the EU and France are limited to WEEE Directive targets (overall weight-based indicators) and focus on technical performance. As presented in section 3.3, different studies recently discussed the importance of reviewing collection and treatment rates calculation methods.

Until 2019, the complementary flows were not included in the scope of the WEEE Directive indicators. As presented in **Chapter 2**, from 2019, the Member States can report their collection rate based on the average of EEE placed on the market in three previous years, or in the waste generated approach. Some of the studies identified in the literature considered the waste generated approach to calculate the collection rate. This approach is interesting because in addition to providing data on the official collection rate, it is possible to quantify the complementary flows.

Regarding the treatment rates, the current calculation by the WEEE Directive does not support the transition towards a circular economy. Firstly, recycling and reuse rates should be assessed separately. Moreover, a more accurate method, including treatment losses, should be used to calculate the recycling rates to encourage the development of high-quality material recycling.

As pointed out in some studies, the targets based on overall-weight do not incentivize the recycling of materials present in lower quantities. Material recycling efficiencies could be based not only on the total amount of materials sent to secondary material processing, but also specifically on the amount of target elements (Huisman, 2003; Nelen et al., 2014). Thus, monitoring collection and recycling of some strategic elements (e.g. critical raw materials), and, eventually, setting treatment targets could boost secondary raw material recovery in the EU.

The quantity collected in euros with the visible fee is the only economic indicator currently reported by the WEEE chain in France. Some costs and revenues are monitored internally, but they provide limited data on economic performance. For example, there is no monitoring in terms of costs per treatment of each WEEE category (€/tons of WEEE treated). We identified in the literature some interesting indicators that estimate the cost-effectiveness and the benefits of the WEEE schemes.

Although no environmental indicators are currently reported by the WEEE chain, in the literature, more than 50% of the studies included at least one environmental indicator. LCA is the most prevalent method, but the feasibility of using LCA indicators to monitor and report the impacts of the WEEE chain should be verified. According to Wang (2014), using the environmental impact per kilo of e-waste to assess the performance of e-waste treatment is challenging due to the high data demand, and the complicated modeling and calculation of impact assessment.

Social impacts of the WEEE chain are not the focus of this thesis, but the monitoring and reporting of the official schemes should include social indicators. Differently to some of the studies identified, France has an official WEEE scheme, established labor laws (e.g. a fixed number of work hours, and child and forced labor are forbidden) and a public health system. Thus, some of the indicators identified in the literature are not suitable to assess the official schemes in France. Nevertheless, certain aspects related to the working conditions (e.g. income, level of education, equal opportunities for men and women) could be addressed by social indicators.

Some indicators identified in the literature could provide interesting information on the treatment performance of the WEEE chain, as well as on its potential as a secondary raw material supplier. However, not all the studies in the literature assessed the feasibility of applying the suggested indicators considering the data currently tracked by the compliance schemes. Apart from developing new indicators, it is necessary to improve data monitoring and transparency.

## 3.5 Conclusions

In this chapter, an overview of the current indicators used to monitor WEEE performance in the EU and France is presented (section 3.2). This assessment allowed us to reply to the first research question of this thesis presented in **Chapter 1** (section 1.2) regarding the type of information provided by the current indicators.

A review of the indicators proposed by different authors in the scientific literature is presented in section 3.3, and the gaps identified are discussed in section 3.4. These sections provided essential insights for subsequent chapters, in which a set of indicators is proposed and validated in a case study in France.

The main issues retained from the literature review for the WEEE chain technical performance, are the need for indicators beyond the weight-based approach, and that provide more precise information on the WEEE chain performance. As suggested by Zeng et al. (2017), future e-waste management should be concerned from macroscopic (e.g. overall collection, recycling per category) to microscopic scales (e.g. type of products POM and collected, materials and elements recovered, collection performance per department/region in a country). A new set of technical indicators is proposed in **Chapter 4**.

Regarding the economic performance, although the compliance schemes and ADEME monitor the financial balance of the WEEE chain, there is an absence of indicators that allow an overview of the expenses and profits per WEEE collected and treated. We identified in the literature review some interesting indicators that could improve the current monitoring. This aspect is explored in **Chapter 6**.

Seeing the importance of including the environmental impacts and benefits of WEEE chain, in the past years some authors suggested indicators mostly based on LCA. Inspired by studies as Huisman (2003) in **Chapter 5,** we explore the potential of LCA to provide environmental indicators for the WEEE chain. A more recent approach identified in the literature review is related to the recovery of critical raw materials (Nelen et al., 2014; Van Eygen et al., 2016). This aspect is addressed in **Chapter 7**.

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## Chapter 4 | Assessment of WEEE flows in France

*"O real não está na saída nem na chegada:*
*ele se dispõe para a gente é no meio da travessia.*[9] *"*
Guimarães Rosa, O Grande Sertão Veredas (1956)

### 4.1 Assessment of WEEE chain technical performance

As discussed in the previous chapters, the current indicators used by the Waste Electrical and Electronic Equipment (WEEE) chain are focused on monitoring its technical performance. These indicators have helped to improve e-waste collection and treatment since its introduction following the first version of the WEEE Directive in the early 2000s. However, its scope is focused on e-waste management simply based on the overall weight of WEEE. To move towards a circular economy, there is a clear need for indicators that can provide information on the potential of the WEEE chain as a supplier of secondary raw materials.

This chapter proposes an expansion of the current indicators used in WEEE legislation (section 4.5). Before presenting the new indicators, three parameters relevant to the approach proposed in this thesis are discussed:

- WEEE flows quantification (section 4.2): the importance of monitoring the waste generated and also the complementary flows in addition to the official collection schemes is explored, as well as a methodology to quantify WEEE flows;
- WEEE composition (section 4.3): aiming to quantify the recovery of secondary raw materials, the need to identify (W)EEE composition is stated;
- WEEE treatment (section 4.4): the e-waste recycling (main treatment in France nowadays) steps are detailed.

The technical indicators are validated with a case study applied to WEEE category II (section 4.6), followed by a discussion on the results (section 4.7) and the conclusion of the chapter (section 4.8).

### 4.2 Quantifying WEEE flows

Information regarding WEEE flows and secondary raw material availability is useful at various levels of decision making, including for European Union (EU) and national policymakers, compliance schemes, recyclers, and designers. In this study, the e-waste flows are assessed using Material Flow Analysis (MFA).

[9] The truth is not in the setting out or in the arriving: it comes to us in the middle of the journey.

MFA is a systematic assessment of the flows and stocks of materials and substances within a system boundary (Brunner and Rechberger, 2004). The system boundary is defined in space and time and can consist of geographical borders (e.g. region or country) or practical limits (e.g. private households) (Van Straalen et al., 2015). The law of conservation of matter is its fundamental principle: inputs into a system must equal the outputs (Zeller et al., 2019). The system boundaries of our study are presented in Figure 4-1, and they cover the processes from e-waste generation in France to the recycling of secondary raw materials.

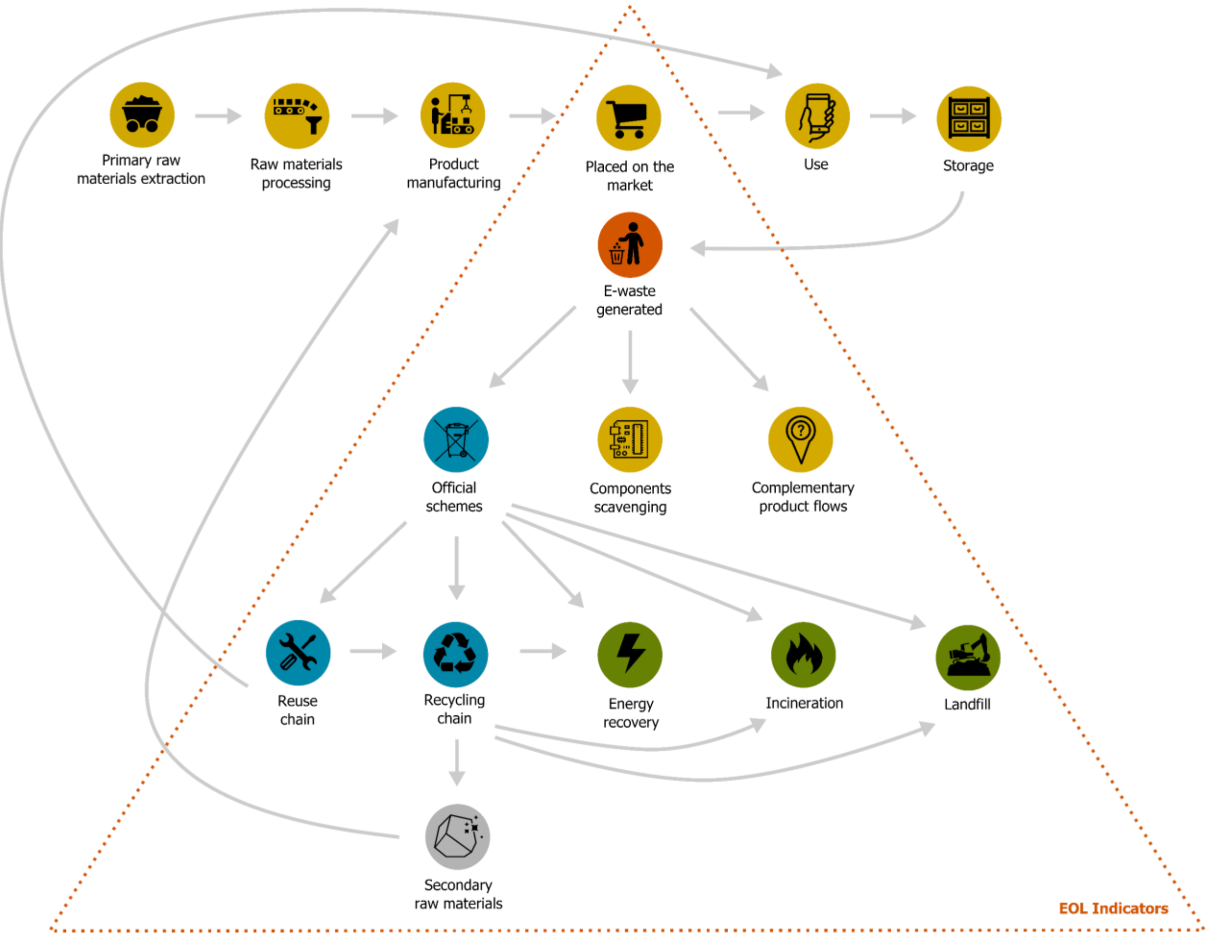



Figure 4-1. Boundaries of EOL indicators.

In waste management, MFA is often used for performance evaluation of current systems and to support decision makers (Allesch and Brunner, 2015; Turner et al., 2016). Through balancing inputs and outputs, the flows of wastes become visible, and their sources can be identified, allowing improvements to be proposed. Thus, MFA is used in the literature to study the route of e-waste flow from the collection, into treatment sites, or disposal areas and also to support the development of indicators (Kiddee et al., 2013).

Afterwards, depending on the scope of the indicators proposed in the study, the system boundaries are adapted (e.g. to calculate the collection rate, the scope may comprise only collection by the official schemes or the complementary flows). The following sub-sections present the methodology to estimate the flows and the performance across the different stages of the e-waste treatment.

Data quality is crucial to ensure a consistent interpretation of the results. As detailed in the subsequent sections, the sources of data in this study are literature data, technical reports, and primary data from Ecologic, ADEME, and companies in charge of collecting and treating WEEE in France.

### 4.2.1. WEEE generation

Evaluating the collection performance necessitates quantifying the total amount of e-waste generated, besides the quantities of e-waste collected and treated by the official schemes. Different methods for quantifying the generation of electronic waste are discussed in the current research and literature on electronic waste (Wang, 2014).

Aiming for a common approach for calculating WEEE generated across the Member States, the European Commission (EC) (based on Magalini et al., 2015) has developed the WEEE Calculation Tool. This tool is an important step towards the adoption by the Member States of a waste generated approach based on a standardized method. Seeing that we intend to propose indicators that can be adopted by the WEEE chain stakeholders (e.g. policymakers and compliance schemes), the WEEE Calculation Tool is used in this thesis to estimate the waste generation. The tool personalized for each Member State, as well as the user manual, can be downloaded from the EC website[10].

The methodology for calculating the WEEE generated using the WEEE Calculation Tool is based on two parameters (Baldé et al., 2017):

- amount of EEE Placed On the Market (POM) in the preceding years;
- corresponding product life span.

The tool already has pre-filled data on the quantities of EEE POM until 2015, based on the apparent consumption methodology. The apparent consumption uses available statistical data of domestic production (PRODCOM statistics) and imports and exports (CN codes[11]) to quantify POM (Magalini et al., 2015). The calculation routines used by the tool were developed by Statistics Netherlands (Van Straalen et al., 2016) and its script, as well as the data extracted from Eurostat databases, are published in open-source format[12]. The users can manually update the POM data after 2015.

As presented in **Chapter 2**, the WEEE Directive has set 6 categories of (W)EEE (see Table 2-1). The categories regroup diverse types of products. In order to be able to calculate the life span per product, the WEEE Calculation Tool converts the POM input data per WEEE category into POM per UNU-

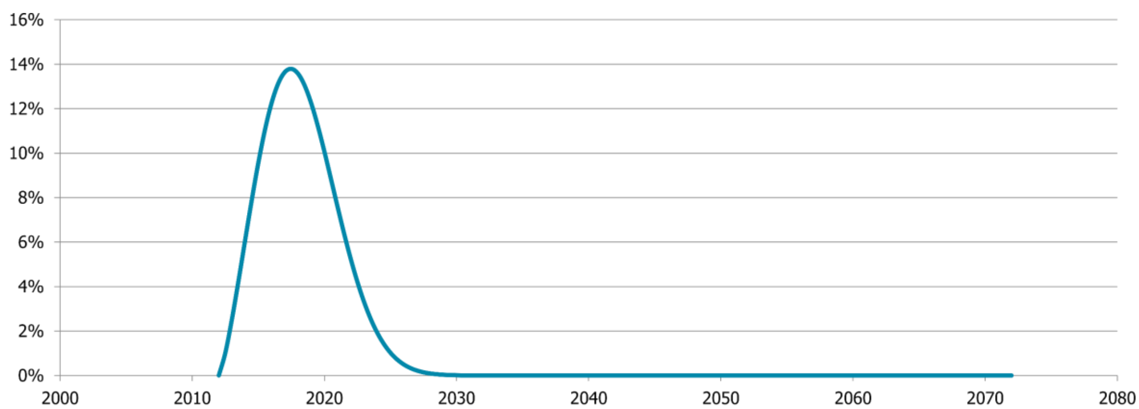
[10] WEEE Calculation Tool is available in: http://ec.europa.eu/environment/waste/weee/data_en.htm
[11] Combined Nomenclature (CN) is the EU's eight-digit coding system. More information is available in: https://trade.ec.europa.eu/tradehelp/eu-product-classification-system
[12] More information about "Waste over Time" can be found here: https://www.cbs.nl/en-gb/our-services/methods/statistical-methods/throughput/throughput/waste-over-time-script

Keys. UNU-Keys is a classification developed by the United Nations University (UNU) to categorize different 'baskets' of WEEE products according to composition and life span properties. The UNU-Keys classification divides different types of WEEE items (about 900 products, clustered into 660 main product types, clustered in subkeys with common compositions, sizes and life spans) into 54 UNU-Keys (Baldé et al., 2015).

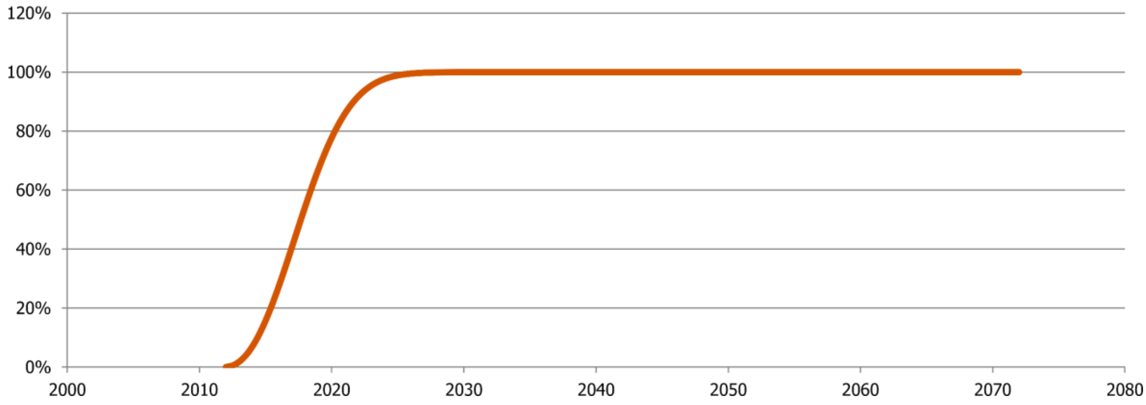
The Weibull distribution function is used to estimate the life span profile of the different UNU-Keys, defined by a time-varying shape parameter (α) and a scale parameter (β). The parameters were defined for each UNU-key based on multiple country studies (Baldé et al., 2017; Forti et al., 2018; Magalini et al., 2015). The Weibull distribution is considered to be the most suitable probability function to describe discarding behavior for EEE (Forti et al., 2018).

This function gives the probability that equipment placed on the market in year *n* appears in WEEE arisings *x* years later (year *n+x*) (Monier et al., 2013b). In this method, the life span for most UNU-Keys is considered constant over time. Exceptions are made for a few UNU-Keys where life span changes dramatically (such as CRT Monitors and TVs) (Magalini et al., 2015). These methodological choices undoubtedly impact the results, and probably do not capture all the factors that may influence the life span (technical, social and cultural), but as previously mentioned, it is the most suitable to estimate life span profile.

Figure 4-2 presents the two life span graphs of flat panel display monitors (UNU-Key 0309) generated by the WEEE Calculation Tool. The first graph (a) shows the percentage of products placed on the market in the reference year and being discarded in each of the following years. It shows that 14% of the monitors placed on the market in 2012 will be discarded in 2020. The second graph (b) shows the cumulative amount, expressed as a percentage, of products discarded. In 2020, 73% of the monitors placed on the market in 2012 will be discarded, and 17% will be still in use. The tool also provides the average lifetime of the flat panel display monitors POM in 2012 estimated as 6.14 years.




a. WEEE generated per year in percentage of POM




b. WEEE generated per year in percentage of POM (cumulative)

Figure 4-2. Life span of flat panel displays in 2012 in France according to WEEE Calculation Tool. Source: adapted from Baldé et al., 2017.

The parameters defined in the tool consider the different socio-economic conditions throughout the Member States. The countries are regrouped into three clusters (stratum) based on their purchasing power (Magalini et al., 2015). France, together with other countries such as Belgium, the Netherlands, and Sweden, is classified in the first stratum in terms of purchasing power. Consequently, the results for the same reference year and UNU-Key may be different, depending on the Member State. For example, the average lifetime of the flat panel display monitors placed on the market in Italy in 2012 (classified in the second stratum together with countries as Portugal and Spain) is estimated at 6.54 years. The different life span is due to the distinct scale parameters set for both countries in the WEEE Calculation Tool.

### 4.2.2. Collection by the official schemes

As presented in **Chapter 3**, the collection rate by the official schemes is among the indicators used to assess the performance of the official schemes. Nowadays it is mostly presented as the amount collected per inhabitant (kg/inh), or as a collection rate based on the total weight of WEEE collected, expressed as the percentage of the overall weight of EEE placed on the market in the three preceding years. Figure 4-3 presents the collection rate by the official schemes in France, as well as the respective targets fixed by the Directive from 2012 to 2018, and the national targets set by the specifications included in the Ministerial Orders for household and professional WEEE (JORF, 2015, 2014).

For household e-waste, France has complied with the target based on the weight (in kilograms) collected per inhabitant (practiced from 2012 to 2015), as well as with the target based on the total weight of WEEE collected, expressed as the percentage of the overall weight of EEE placed on the market (POM) (in practice from 2016). As discussed in **Chapter 2**, from 2019, the collection rate can be calculated based on volumes of EEE POM, or WEEE generated.

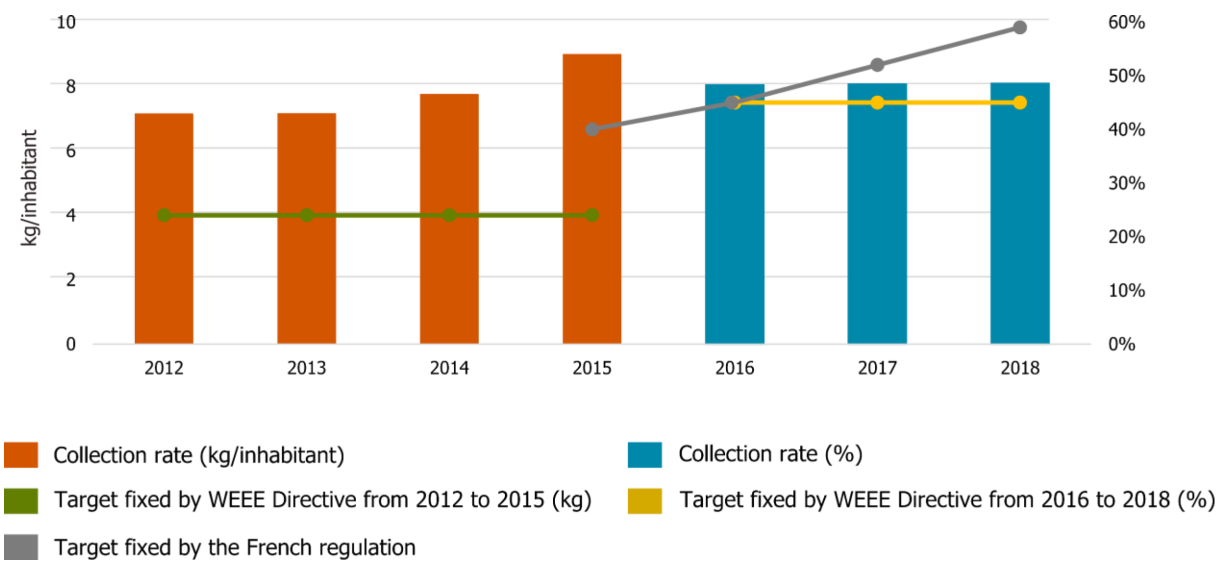



Figure 4-3. Collection rate from 2012 to 2018 in France and WEEE Directive targets. Source: based on data from Fangeat, 2019.

Regardless of the approach used to calculate the collection rate, the information regarding the amount of WEEE collected (in kg or tons) is essential for the assessment. Currently, the national registers only track the total amount of WEEE collected and treated per category and/or waste streams by the official schemes. Usually, there is no detailed assessment in terms of breakdown of the flow per type/category of products or their clustering per UNU-key. This limits the usefulness of the information, not allowing monitoring in terms of products, materials and elements collected by the official schemes.

#### 4.2.3. Complementary flows and scavenging

In 2012, only one-third of WEEE discarded in the EU was collected by the official schemes (Huisman et al., 2015). WEEE not collected by the official schemes ends up in other non-documented and unofficial routes (complementary flows), including equipment reused, e-waste disposed in the municipal waste stream, as well as undocumented exports either as (illegal) waste or as reusable items (Bigum et al., 2013; Huisman et al., 2015; Vidal-Legaz et al., 2018).

E-waste wrongly disposed of with the household waste has a little chance of separation. It goes to landfills or incineration, which results in toxic leaching or harmful emissions (Kumar et al., 2017). It is also known that e-waste is illegally exported to countries with low enforcement laws. Countries in Africa and Asia have been identified as recurring destinations of e-waste illegally transported from European and North-American countries (Li et al., 2013).

Another relevant source of flows not documented by the official schemes is components scavenging (Huisman et al., 2017). Scavenging is a consequence of plundering in the e-waste collection points, and of components removed by the consumers before disposing of the e-waste. The components are scavenged for: (I) elements with high economic value (e.g. copper in cables and precious metals in printed circuit boards); (II) potential revenues that can be obtained when re-selling components for reuse purposes; and (III) cheaper treatment in countries where compliance with EU WEEE legislation is not required or not enforced (Magalini and Huisman, 2018). Table 4-1 presents the scavenging level estimated per category (in percentage) according to a study performed by the European Electronics Recyclers Association (EERA).

Table 4-1. Component scavenging in WEEE.

| Category | Component | Scavenging level |
|---|---|---|
| I. Temperature exchange equipment | Cables | 22% |
| | Compressors | 22% |
| | Casings | 7% |
| | Other parts | 24% |
| II. Screens, monitors, and equipment containing screens having a surface greater than 100 $cm^2$ | Cables | 30% |
| | Copper and iron motors/coils | 8% |
| | Circuit boards | 5% |
| | Drives | 32% |
| | Batteries | 15% |
| | Other parts | 15% |
| IV. Large equipment | Cables | 11% |
| | Copper and iron motors/coils | 10% |
| | Casings | 2% |
| | Other parts | 3% |
| VI. Small IT and telecommunication equipment | Cables | 16% |
| | Circuit boards | 14% |
| | Batteries | 1% |
| | Other parts | 15% |

Source: based on Magalini and Huisman, 2018.

From an environmental perspective, the existence of complementary flows is alarming because it is known that the WEEE devices are treated in conditions that represent a risk to the environment and human health. For example, a fridge inappropriately treated may release ozone-depleting substances (e.g. CFC’s) contained in the refrigeration system.

Nowadays, there is no official tracking of complementary flows, and this issue is not widely discussed in the WEEE chain. A study performed in France based on the POM approach, estimated that in 2012 the official schemes collected 30% of the e-waste, whereas 30% were dissipated in complementary flows identified by the study and 40% were captured by other undocumented flows (Monier et al., 2013b). Regarding the complementary flows identified by the study, it was estimated that 13% was treated in mixed scrap metal in France, 8% exported in mixed scrap metal, 4% disposed in the municipal waste stream and, 4% disposed in bulky waste containers. This figure has probably changed in the past years as a result of the compliance schemes’ efforts to improve collection (which was increased by 15% in 5 years), but it gives an idea of the complexity of the scenario, on the lack of recent official data and the need for additional indicators that can monitor e-waste flows. Nevertheless, the total amount of complementary flows (products and components scavenged) can be indirectly calculated as the difference between the quantity of e-waste generated, and the quantity collected by the official schemes (for further details see section 4.5).

## 4.3 (W)EEE composition

Metal content and the total amounts of elements in e-waste indicate the volume and grade of potential secondary resources (Oguchi et al., 2011). To track target elements in e-waste flows, data about the composition of waste flows and products is essential. The composition of waste flows involves classifying the different types of equipment generated and collected per waste stream/category. In turn, the composition of (W)EEE includes its characterization per type of products (e.g. UNU-Keys), components, materials and/or elements.

The composition of (W)EEE depends on producers, functionalities of the devices and changes in technologies. Currently, producers do not report EEE composition, and the compliance schemes usually do not sample e-waste on a regular basis. Moreover, the e-waste streams include a mix very different and heterogeneous of products, so sampling is rarely representative.

Annually, the French compliance schemes perform a composition assessment per waste stream, and the average composition of e-waste is reported in the annual report released by ADEME, as presented in Table 4-2.

Table 4-2. Composition of household e-waste treated in France (2017).

| Output fractions | Quantity of e-waste treated (t) | Percentage of e-waste treated (%) |
|---|---|---|
| Printed circuit boards | 6,260 | 0.9% |
| Ferrous metals | 346,272 | 51% |
| Non-ferrous metals | 52,712 | 7.8% |
| Plastics (mix) | 115,291 | 17% |
| Mineral fraction | 17,910 | 2.6% |
| Shredding residues | 92,975 | 13.7% |
| Glass | 38,926 | 5.7% |
| Others | 8,094 | 1.2% |
| Total | 678,409 | 100% |

Source: based on data from Deprouw et al., 2018.

The results allow a rough estimation of the output fractions from shredding and sorting processes. However, the sorting processes are not 100% efficient, and afterwards, there are further treatments to recover target elements (e.g. iron, copper and gold). As discussed in **Chapter 2**, the compliance schemes focus on monitoring the first steps of e-waste chain to ensure end-of-waste fulfillment. Until they track the recovery of the elements in the output fractions, it will not be possible to use primary data from the e-waste chain to monitor collection and recycling performance per target element.

In the case study presented in section 4.6, data from the ProSUM project is used to identify the composition of screens. The core of this system lies in the representation of waste flow "f" (e.g. screens category) as a regrouping of products "p" (e.g. TVs and laptops) that are the sum of their constituent components "c" (e.g. hard drives and batteries), materials "m" (e.g. alloys) and elements "e" (e.g. copper), as represented in Figure 4-4. Details regarding the methodologies used to characterize the e-waste composition (sampling, sample preparation and chemical analysis) are presented in the project's report (Huisman et al., 2017).

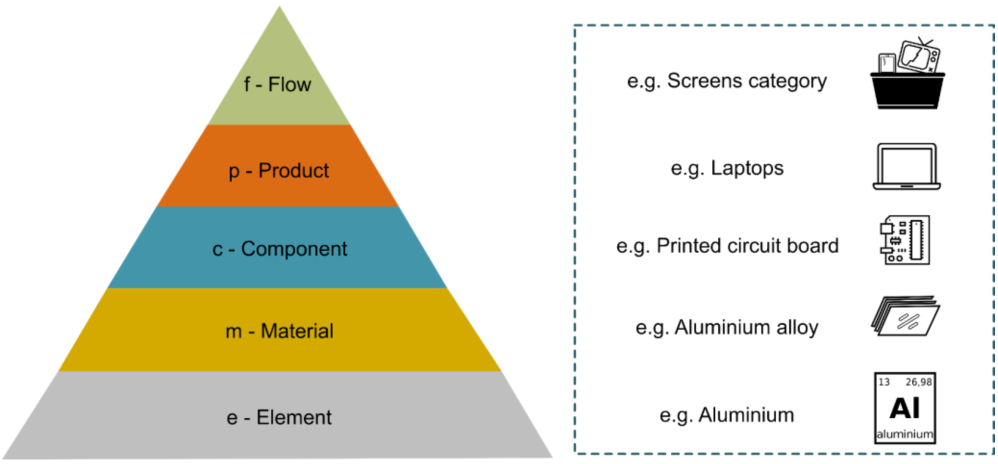



Figure 4-4. ProSUM classification system.
Source: adapted from Huisman et al., 2017.

The advantage of this classification is that it allows a uniform description of all composition parameters, from the share of products (e.g. per UNU-key) to the waste flow (p-f) and the metal content in a certain alloy (e-m) used in certain components (m-e). This information is necessary to determinate the recoverability of elements based on the technological processes available. Often, WEEE composition data is provided as the content of the elements in flows or products (e.g. quantity of antimony or gold in tablets). However, the recoverability of the elements is influenced by their content in materials and components. For example, antimony can be recovered in processes based on complex lead/copper/nickel metallurgy with high efficiency (>80%), but antimony in plastics and cathode-ray tube (CRT) glass is generally not recycled (Chancerel et al., 2015; Hagelüken, 2006).

Seeing the complexity of e-waste composition, in the case study we decided to concentrate on a few elements present in WEEE category II. The selection of the target elements was made after a first screening of screen composition with data available in the Urban Mine Platform[13]. Besides elements

[13] More details on the Urban Mining Platform is available in: www.urbanmineplatform.eu

present in higher terms by weight, we selected elements that have a high market value, as well as some Critical Raw Materials (CRMs) that are present in smaller quantities (for further details on CRM see **Chapter 7**). The following 11 elements were selected, of which 6 are included in the criticality list (see 7.2.2) released by the European Commission in 2017 (European Commission, 2017):

- Non-critical raw materials: aluminum (Al), gold (Au), silver (Ag), copper (Cu) and lithium (Li);
- Critical raw materials: antimony (Sb), cobalt (Co), indium (In), magnesium (Mg), neodymium (Nd) and palladium (Pd).

Lithium was considered in this study even if it has a rather average intrinsic value (3 times higher than copper but 2,000 times lower than gold) and is not considered a CRM in the EC list because its economic importance is slightly below the limit (≥ 2.8). However, lithium supply risk is above the limit considered in the criticality zone (≥ 1), and with its rapidly growing consumption, the economic importance result may change in the future criticality assessment (for further details on criticality methodology see section 7.2).

The targeted elements are present in different components of screens. Aluminum and magnesium are present in several different types of alloys in screen casings. Among others, PCBs contain precious metals (Ag, Au and Pd), as well as base metals like aluminum and copper. Copper is also present in cables and CRT screens' motors. Antimony is mostly found in plastic components (applied as a synergist for brominated flame retardants), and CRT glass. Lithium and cobalt are mainly present in batteries in tablets and laptops. Indium and neodymium are mainly present in flat panel displays (FPD), in thin-film-transistor (TFT), in liquid-crystal displays (LCD) and magnets in drives.

## 4.4 WEEE treatment: focus on recycling

In the European regulatory hierarchy of waste treatment, prevention is seen as the most desirable option, followed by preparing for reuse and recycling (European Commission, 2008; Manfredi and Goralczyk, 2013). Nonetheless, only a relatively small proportion of the WEEE is reused (Cole et al., 2019).

In France, since the set-up of the WEEE schemes, the main treatment is recycling (Les Amis de la Terre France, 2016). As can be noticed in Figure 4-5 for all six collection waste streams, recycling is the main treatment. Reuse is still modest and not well documented by the official schemes.

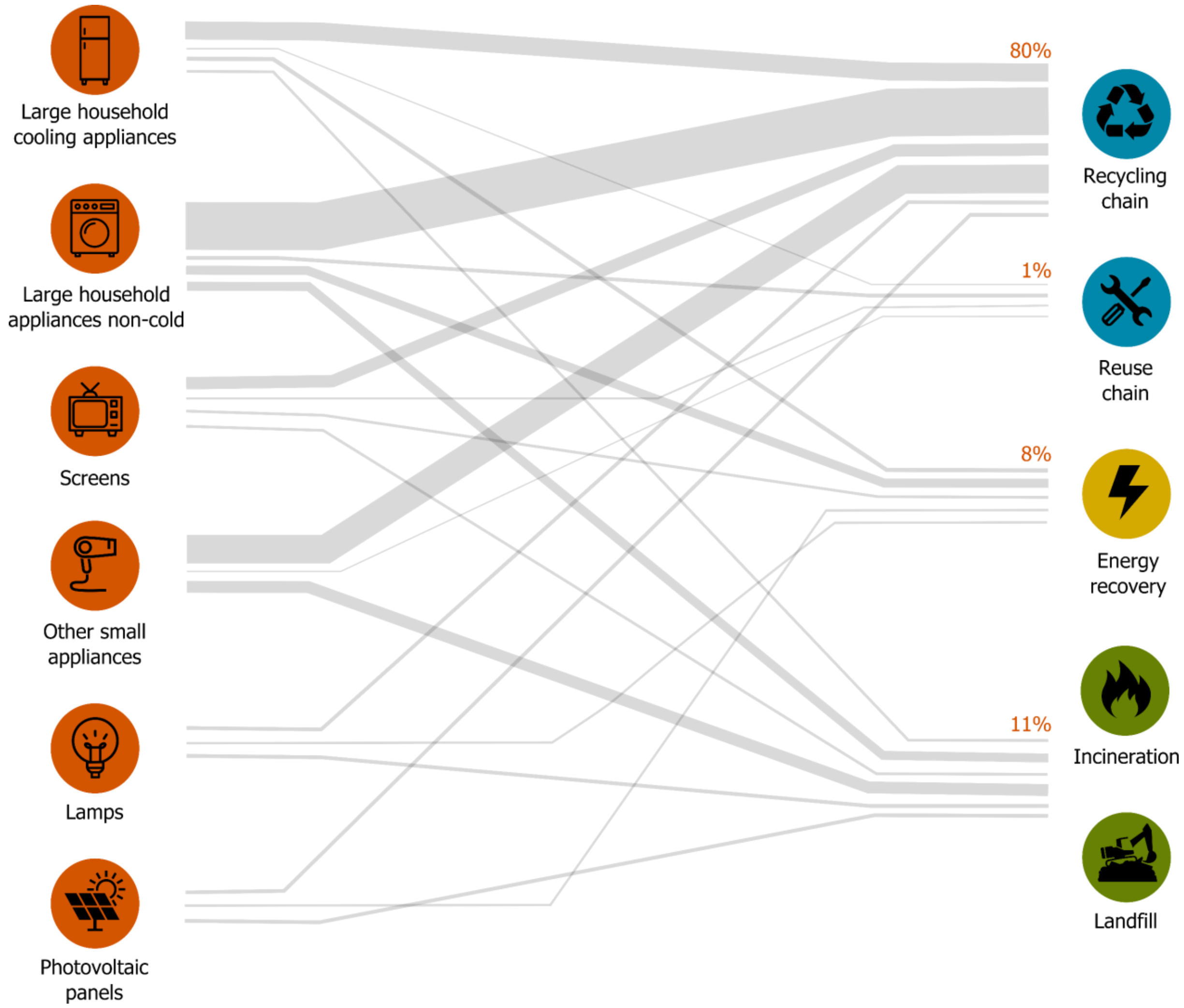



Figure 4-5. Household WEEE officially collect in France per type of treatment (2017).
Source: based on Deprouw et al., 2018.

One of the reasons for having low amounts of EEE officially destined to reuse is a consequence of the systems' set-up. As presented in **Chapter 2**, the collection of household WEEE is divided into four collection channels (municipal waste collection centers, retailers' collection points, social enterprises, and other collections organized by the compliance schemes). The only channel that collects EEE for reuse is the social enterprises, and in recent years it only accounted for less than 4% of the total weight of household (W)EEE collected. Even if part of the WEEE collected by the other channels could be reused, the devices are not tested before entering the recycling chain process. Many consumers are not aware of the collection system operation and discard WEEE that could be potentially reused in other collection channels.

Regarding professional e-waste in France, a higher amount is reused, according to the national registers. In 2017, 3.1% of professional e-waste treated by the official schemes was reused, in comparison to 1% of the household (Deprouw et al., 2018). The higher reuse for professional WEEE is a consequence of reuse contracts established by some companies. According to the same report, IT and telecommunication equipment (category VI) are replaced before their end-of-life, allowing further reuse.

The WEEE Directive refers to an aggregated target for the "WEEE prepared for reuse and recycled" per WEEE category. Aiming to promote reuse, some Member States such as Spain and Belgium introduced separate quantitative targets (Vidal-Legaz et al., 2018). Figure 4-6 presents, for the ten former categories, the disaggregated results of reuse and recycling rate in the EU in 2015, as well as the number of countries complying with the WEEE Directive target. The results represent the average percentages for the EU-28, excluding Cyprus, Malta, and Romania due to the absence of data when the survey was carried out (July 2018).

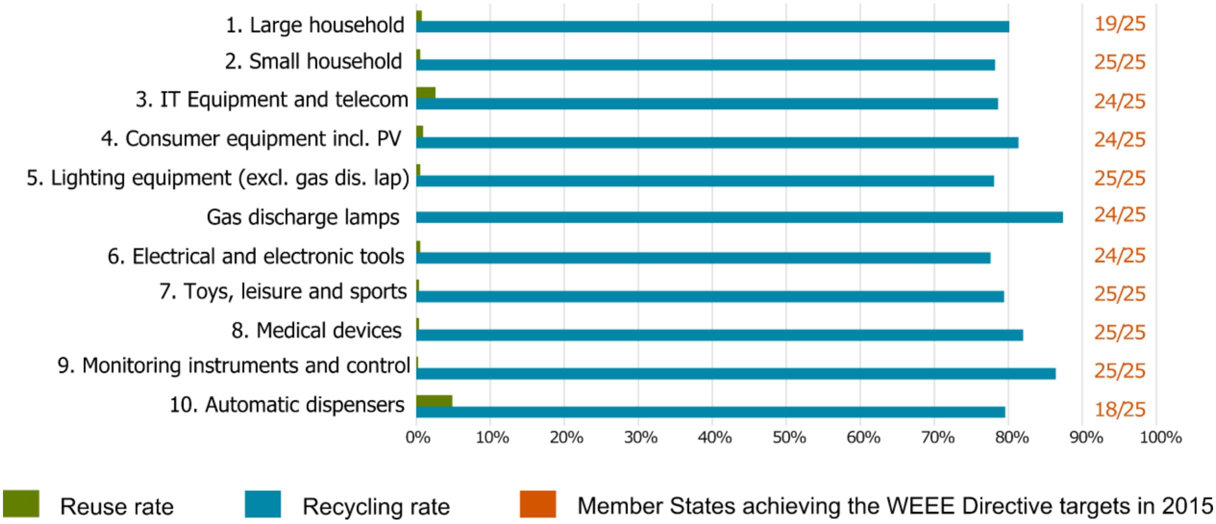



Figure 4-6. Reuse rate and recycling rate (in percentage) per WEEE category and number of Member States achieving the targets of the WEEE Directive.
Source: adapted from Vidal-Legaz et al., 2018.

As discussed in **Chapter 2**, recycling and reuse rate is calculated, for each WEEE category, by dividing the overall weight that enters the recycling/preparation for reuse facility by the overall weight of e-waste collected, expressed as a percentage. As can be noticed, recycling rates are generally high for all categories, ranging from 78% for formerly category 6 (electrical and electronic tools) to more than 87% for gas discharge lamps. Reuse has low results in all categories. The highest results for reuse are for automatic dispensers (formerly category 10), and IT and communication equipment (formerly category 3): 5% and 3% respectively – which corresponds to the current categories V (large equipment) and VI (small IT and telecommunication equipment).

Between the e-waste collection by the official schemes and the actual production of secondary raw materials, there is a complex recycling chain driven by economic and environmental interests (Mathieux et al., 2008; Valero Navazo et al., 2013). This chain includes several stakeholders in charge of transport, storage, clean-up, shredding, sorting, disposal of the non-recyclable fraction and preparation of the secondary raw materials (Tesfaye et al., 2017).

After the collection, the WEEE is transported to a regrouping center to be separately weighed per collection stream (large household cooling appliances, large household appliances non-cold, screens, other small appliances, lamps, and photovoltaic panels). After achieving a certain volume or weight, the WEEE is transported to a treatment center where it undergoes different treatments depending on the type of WEEE, until the sorting of recyclable and non-recyclable fractions.

Recycling of complex, multi-material consumer products demands an extended network of different types of processes to recover the wide range of materials present (Van schaik and Reuter, 2012). The more materials to be recovered, the more extraction steps are usually carried out (Ilankoon et al., 2018). Figure 4-7 summarizes the main treatment steps presented in more detail below.

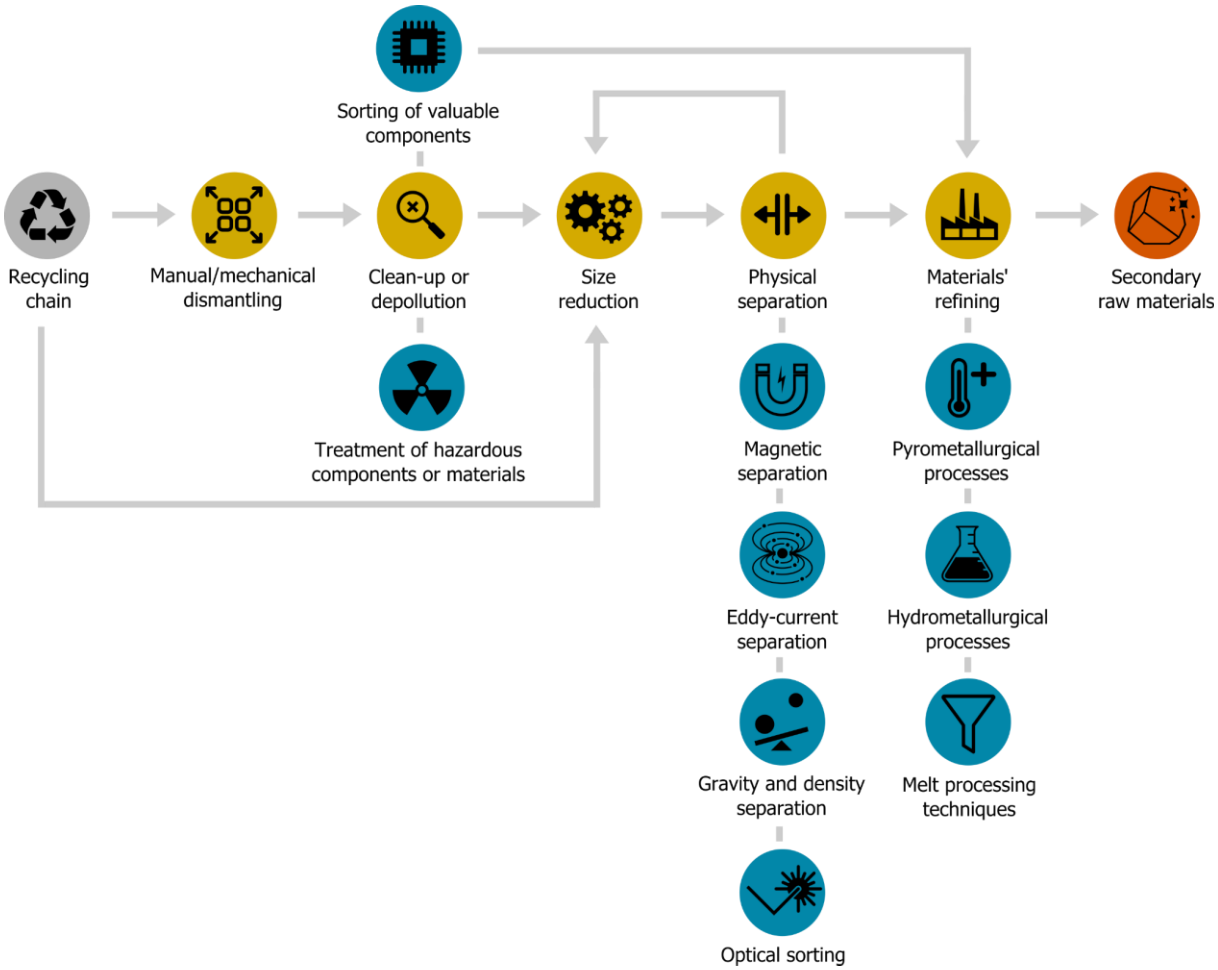



Figure 4-7. Schematic flowsheet of recovery of secondary raw materials from e-waste.
Source: based on data from Batinic et al., 2018; Menad, 2016; Tuncuk et al., 2012.

Pre-processing of e-waste is one of the most important steps in the recycling chain (Tesfaye et al., 2017). Its efficiency affects the overall recovery of metals and polymers from WEEE (Chancerel et al., 2011). Usually, it starts with manual and/or mechanical dismantling of the e-waste. This step aims to upgrade valuable substances contained in the e-waste while eliminating hazardous components (Cui and Zhang, 2008). Manual sorting of high-value components (e.g. PCBs and drives) and large plastic and metallic components (e.g. TVs and laptops housing) increases the resource efficiency (in terms of the quantity and quality of recoverable materials), however, it entails higher labour costs (Ardente et al., 2014; Tecchio et al., 2018).

The WEEE Directive requires this selective treatment, during which certain substances, mixtures and components must be removed (this step is known as depollution, de-pollution or clean-up) before e-waste can undergo any further form of treatment (European Commission, 2012). Table 4-3 summarizes the materials and components that must be removed from any separately collected WEEE according to Annex VII of the WEEE Directive. Depending on the type of element/component, depollution is prior to (e.g. cables) or after (e.g. printed wiring boards) the dismantling.

Depollution must not damage or destroy components in such a way that hazardous substances are released into the environment (European Standardization Organizations, 2014). Besides the dangers highlighted in the table, some of the components also contain valuable and/or critical raw materials. As suggested by Huisman et al., 2008, some requirements highlighted in Table 4-3 (requirements 9, 10 and 12) from the safety outlook, are superfluous either by the existence of other legislation or as a result of product changes, or by reasons of being prematurely over-prescriptive. In this context, the redundant items could be potentially removed in a future revision of the WEEE Directive.

Dismantling and clean-up are performed for most of the waste streams, but some e-waste, such as those comprising the "other small appliances" waste stream, are sent directly to size reduction. In the step known as size reduction, shredders and/or hammer mills are used to fragment, grind and rip the waste. These processes pursue the liberation of metals and their separation from the non-metallic fraction (Marra et al., 2018). The extent of size reduction depends on the recovery processes that are implemented in the following steps.

E-waste is then separated based on physical and physicochemical properties using techniques like those used in the mineral dressing (Tesfaye et al., 2017). Physical sorting techniques enable the separation or isolation of different output fractions for suitable treatment or disposal (Menad, 2016). Depending on the types of fractions to be separated, different technologies can be used to sort them. Moreover, size reduction can be necessary at different stages (before and after certain sorting processes), as represented by the arrow connecting "physical separation" to "size reduction" in Figure 4-7.

Separation of ferrous metals (iron, steel, nickel, etc.) is performed by magnetic separation, where low-intensity magnetic drum separators are widely used for this purpose. This step is usually followed by separation of non-ferrous metals flow (aluminum, copper, etc.) from other materials (mainly plastic, glass, etc.), by using electric conductivity-based separation techniques, as eddy current separation and electrostatic separation (e.g. roll-type corona-electrostatic separator).

Table 4-3. Materials and components of WEEE that must be removed before further treatment.

| Materials and components to be removed | Description |
|---|---|
| 1. Asbestos waste and components which contain asbestos | Asbestos was widely used in consumer products (e.g. electrical heating devices) due to its heat and chemical resistance. It can cause serious health diseases thus should be treated separately |
| 2. Batteries | Batteries contain heavy metals such as lead, mercury and cadmium that are dangerous to human health and the environment |
| 3. Capacitors containing polychlorinated biphenyls | Polychlorinated biphenyls were commonly used as heat stabilizers in cables and electronic components. They are classified as persistent organic pollutants, and require specific treatment |
| 4. Cathode ray tubes (CRT) | CRT contain lead in the cone glass and fluorescent coating cover in the glass panel |
| 5. Chlorofluorocarbons (CFC), hydrochlorofluorocarbons (HCFC) or hydrofluorocarbons (HFC), hydrocarbons (HC) | Used as refrigerants in refrigeration and air conditioning equipment. Have a high global warming potential, thus should be properly extracted and destroyed or recycled |
| 6. Components containing mercury, such as switches or backlighting lamps | Mercury is used in thermostats, sensors, relays and switches (e.g. on printed circuit boards and in measuring equipment and discharge lamps); and also, in medical equipment, data transmission and telecommunication |
| 7. Components containing radioactive substances except for components that are below the exemption thresholds | Radioactive waste is hazardous to human health and the environment, and must treated selectively |
| 8. Components containing refractory ceramic fibers | Ceramic fibers are used as insulation materials in some appliances (e.g. toasters, self-cleaning ovens), because of their ability to withstand high temperatures. They can cause serious health diseases thus should be treated separately |
| 9. Electrolyte capacitors containing substances of concern (height > 25 mm, diameter > 25 mm or proportionately similar volume) | No substances of concern have been identified other than the very remote possibility of polychlorinated biphenyls that is already covered by entry #3 |
| 10. External electric cables | The possible substances of concern in external cables are also present in internal cables, which are not required to be removed |
| 11. Gas discharge lamps | Mercury must be removed |
| 12. Liquid crystal displays of a surface greater than 100 $cm^2$ and all those back-lighted with gas discharge lamps | The possible substance of concern is mercury present in backlight in some LCD units, that is already covered by the entry #6 |
| 13. Plastic containing brominated flame retardants | During incineration/combustion of the plastics flame retardants can produce toxic components |
| 14. Printed circuit boards (PCBs) with surface greater than 10 $cm^2$ | Cadmium occurs in certain components in PCBs, such as SMD chip resistors, infrared detectors and semiconductors |
| 15. Toner cartridges, liquid and paste, as well as color toner | Toner powder and consumables containing it are classified as hazardous waste and as such, must be separately treated |

Source: based on Cui and Forssberg, 2003; European Parliament, 2012; Huisman et al., 2008.

Sensor and/or optical based sorting techniques are also applied in certain treatment facilities. Usually, optical sorting is used in the advanced stage of the sorting process, in order to separate plastics further (e.g. separation of brominated from nonbrominated plastics and types of polymers) and shredded printed circuit boards. The plastic mix that does not contain restricted brominated flame retardants (BFRs) above the threshold values (2,000 ppm of bromine) also can be sorted by optical techniques, followed by advanced recycling processes.

From the economic point of view, physical separation processes benefit from low capital and operating costs. Nevertheless, the processes are not 100% effective as physical properties can be affected by particle size or adhesion to surfaces, and the losses can be significant, from 10 to 35% (Tuncuk et al., 2012; Valero Navazo et al., 2013). For example, aluminum-laminated plastics are mostly found in the aluminum flow; and metal screws are regularly embedded in plastic elements that pollute the ferrous metals output fraction flow (Horta Arduin et al., 2019).

In the refining stages of the recycling chain, recovery of the metals is achieved by metallurgical processes similar to the primary production of metals. Consequently, the recovery of precious metals from e-waste is done by major primary metal producers (e.g.Boliden (Sweden), Umicore (Belgium), Aurubis (Germany) and Brixlegg (Austria), Xstrata (Canada), and Dowa (Japan) (Valero Navazo et al., 2013). The fractions are melted (pyrometallurgical processes) and/or dissolved (hydrometallurgical processes) and further sorted based on their chemical and metallurgical properties (Van schaik and Reuter, 2012).

The main pyrometallurgical methods employed for the recovery of metals from e-waste are smelting, combustion, pyrolysis, and molten salt processes (Ebin and Isik, 2016). During pyrometallurgical treatments, components and output fractions are heated and melted in furnaces at temperatures above 1,500°C to remove the plastics, whereas the metals are concentrated in a molten bath. The latter are cast as anodes, which is further refined using electrometallurgical processes for obtaining high purity metals (copper). Slurry residues derived from this refining phase are rich in precious metals and are then processed for their recovery. Umicore recycling process continues with leaching and electrowinning (Marra et al., 2018; Zhang and Xu, 2016).

Hydrometallurgy, also known as leaching, consists of the selective dissolution of metals. The isolation of metals of interest is conducted through solvent extraction, adsorption, and ion exchange processes. It involves the use of aqueous chemicals to dissolve the metal, which is then recovered from the solution by electrolysis or precipitation (Tesfaye et al., 2017; Valero Navazo et al., 2013).

Modern pyrometallurgical and hydrometallurgical refineries achieve above 95% recovery of gold present in the e-waste fraction or component and are capable of recovering numerous metals in addition to precious metals and copper (Charles et al., 2017). Nevertheless, recovering precious and less common metals may not be economically feasible (Oguchi et al., 2011).

Regarding plastic recycling, the first step is the separation of the individual polymers, which also results in non-recyclable fractions (European Electronics Recyclers Association, 2018). Mechanical recycling per type of plastic is an effective alternative only if a high-purity product (i.e. the grade of product be higher than 96%) is achieved via a separation process. For example, a triboelectric separator can be employed to collect a PS-rich positively charged fraction, and a PE-PVC negatively charged fraction. Subsequently, the PE-PVC fraction can then be separated using an air table, taking advantage of differences in specific gravities (Dodbiba et al., 2008). After sorting, plastics are subjected to melt processing techniques such as extrusion, especially for making pellets and injection molding depending on the potential secondary use (Sahajwalla and Gaikwad, 2018).

The quality of recycled plastics depends both on the product input and on the recycling process (Wagner et al., 2019). The main difficulties identified in e-waste plastics recycling are: (1) presence of brominated flame retardants; (2) sorting limitations due to black plastics; (3) presence of plastic additives; (4) significant ranges of polymers together with a lack of monitoring of the treatment inputs and limited technologies to sort and recycle them (Buekens and Yang, 2014; Deloitte et al., 2015; Ma et al., 2016; Sahajwalla and Gaikwad, 2018). Therefore, the recycled material often has a lower quality and functionality than the original material (downcycling). Additionally, the polymer recycling processes may be more expensive than the equivalent raw material extraction and manufacturing procedures (IEA, 2014).

### 4.5 Expansion of the current technical indicators

After the assessment of the current indicators and their limitations (**Chapter 3**), in this section supplementary indicators are proposed to improve technical performance monitoring based on the three parameters discussed in the previous sections of this chapter: (1) WEEE flows; (2) composition of (W)EEE; and (3) treatment performance. The supplementary indicators are presented in the following subsections, and their feasibility is assessed in a case study focused on screens in France presented in section 4.6.

#### 4.5.1. Follow up of EEE placed on the market

Nowadays, the EEE placed on the market in France is reported based on the overall weight and number of pieces placed on the market per (W)EEE categories. It does not include an assessment per type of product or UNU-Keys. We believe that an indicator per category is suitable for an overall POM follow-up, however it is important to improve POM tracing to enable tracking of the changes in equipment type that will impact the end-of-life. It could support, for example, the development of recycling techniques in order to treat the e-waste that will arrive in the following years.

We suggest a tighter tracking and assessment of EEE POM in France in order to better capture changes in technology that impact the treatment of this EEE when it becomes waste. The current follow-up by ADEME is based on Harmonized System (HS) four-digit coding system, and not in the CN eight-digit coding system used by the WEEE Calculation Tool to convert the POM input data per WEEE category into POM per UNU-Keys (see section 4.2.1). Thus, we suggest changing the monitoring in France, in accordance with the European approach above mentioned.

Besides quantifying the EEE placed on the market in terms of weight and pieces, we propose to report it as a ratio of the number of inhabitants. This information allows EEE consumption to be quantified per inhabitant, which can be compared with EEE POM in other countries that have higher or lower population density, as well as with the quantity of WEEE treated per inhabitant. In this context, Equation 4-1 presents an additional indicator to report EEE placed on the market.

$$POM_f = \frac{\sum_{i=1}^{n} WM_{fi}}{N_{inh}}$$

$POM_f$ EEE placed on the market per inhabitant per WEEE category (weight or number of pieces/inh/year);

$n$ number of different UNU-Keys related to the WEEE category;

$WM$ amount of EEE placed on the market (weight or number of pieces);

$N_{inh}$ population of the country or region in a given year.

Equation 4-1. EEE placed on the market per inhabitant.

### 4.5.2. Collection rate based on the WEEE generated approach

As previously discussed, according to WEEE Directive, from 2019, the collection rate can be calculated with both methodologies (POM and WEEE generated). Seeing the importance of assessing the performance of the official schemes in comparison to actual waste volumes, including the complementary flows, we suggest the adoption of waste generated approach in France.

The actual target fixed by the WEEE Directive is based on the overall WEEE collected. Currently, in France, the collection rate is also calculated and reported for the household e-waste per collection waste stream, and per category for professional e-waste. For better monitoring of collection performance, as well as to facilitate comparison with other Member States that have different collection waste streams, we suggest calculating and reporting the collection rate for household and professional e-waste per category (Equation 4-2).

$$CR_f = \frac{\sum_{i=1}^{n1} WC_{fi}}{\sum_{i=1}^{n1} WG_{fi}}$$

$CR_f$ collection rate of the WEEE category;

$n1$ number of different UNU-Keys related to the WEEE category;

$WC$ weight of e-waste collected by the official schemes;

$WG$ weight of e-waste generated.

Equation 4-2. Collection rate based on the WEEE generated approach.

### 4.5.3. Collection rate per target element

In addition to the collection rate calculated based on the overall weight, we propose to calculate the collection rate per target element (Equation 4-3). This indicator aims to highlight that the performances per element can be significantly different from the overall weight and among elements, due to variances in the type of e-waste generated and collected by the official schemes.

$$CR_e = \frac{WC_e}{WG_e}$$

Where:

$$WC_e = \sum_{i=1}^{n1} WC_{ei} \times \sum_{j=1}^{n2} CH_{eij} \times (1 - SG_J)$$

$$WG_e = \sum_{i=1}^{n1} WG_{ei} \times \sum_{j=1}^{n2} CH_{eij}$$

| | |
|---|---|
| $CR_e$ | collection rate of the target element in the WEEE category; |
| $WC_e$ | weight of target element collected; |
| $WG_e$ | weight of target element generated; |
| $n1$ | number of different UNU-Keys related to the WEEE category; |
| $WC$ | weight of e-waste collected by the official schemes; |
| $n2$ | number of components and materials present in the different products; |
| $CH_e$ | element content per component and materials in the products; |
| $SG$ | scavenging rate per component; |
| $WG$ | weight of e-waste generated. |

Equation 4-3. Collection rate per target element.

### 4.5.4. French departments complying with collection targets

As presented in **Chapter 3**, the quantity (in tons) collected per department is monitored by ADEME and is published in its annual report. In the last version of the Scoreboard published by the JRC, together with the data of recycling and reuse rates, we find the number of Member States complying with the WEEE Directive targets. Inspired by these reports, we suggest defining the rate of French departments complying with the collection target set by the European regulation (Equation 4-4).

$$CR_d = \frac{DC}{DT}$$

$CR_d$ rate of departments complying with the collection target;

$DC$ number of French departments with collection target equal or higher than the collection target set by the WEEE Directive;

$DT$ total number of departments in France.

Equation 4-4. French departments complying with collection targets.

Similar to the previous indicator, the rate of departments complying with the collection target aims to provide more detailed information on the collection performance in France. This approach aims to move from the current overall macroscopic rate to specific rates, providing detailed information to improve official chain monitoring.

### 4.5.5. Clean-up efficiency

Monitoring of clean-up performance is an important criterion to facilitate continuous improvement of the treatment process (European Standardization Organizations, 2014). Even if the Directive imposes the removal of some components before further treatment (see section 4.4), it does not establish indicators and/or targets related to the clean-up. In the report released annually by ADEME, the total amount of the components highlighted in the Directive are presented, but it is not possible to monitor clean-up efficiency based only in the total quantities removed. Moreover, only the overall weight is reported (e.g. 1,295 t of batteries were removed from household e-waste in France in 2017), thus is not possible to identify the share removed per WEEE category.

After the WEEE Directive recast, the European Commission requested the European Standardization Organizations to develop European standards (ENs) for the collection, logistics and treatment (CENELEC, 2017). The clean-up is addressed by the standards and technical specifications (TS) presented in Table 4-4, and target values are clearly defined for batteries and capacitors, and limit values for concentrations of hazardous substances to be achieved by the end of the treatment processes.

The development of standards is an important step in the establishment of methods and protocols to improve monitoring of e-waste treatment performance. However, at present, standards adoption is not compulsory. The current data reported by ADEME and the compliance schemes does not allow the proposed monitoring indicators to be calculated. Moreover, we identified in discussion with Ecologic that some stakeholders of the e-waste chain in France contest the target values fixed by the technical specifications. Thus, we believe that further studies should be performed to support implementing compulsory clean-up targets in the Member States based on current performance and on stimulating progressive improvement.

Table 4-4. Standards and technical specifications (TS) related to e-waste depollution.

| Type of WEEE | Reference | Title |
|---|---|---|
| General | EN 50625-1 | Collection, logistics & treatment requirements for WEEE - Part 1: General treatment requirements |
| | TS 50625-3-1 | Collection, logistics & treatment requirements for WEEE - Part 3-1: Specification for depollution |
| Lamps | EN 50625-2-1 | Collection, logistics and treatment requirements for WEEE - Part 2-1: Treatment requirements for lamps |
| | TS 50625-3-2 | Collection, logistics & treatment requirements for WEEE - Part 3-2: Specification for depollution – Lamps |
| Displays | EN 50625-2-2 | Collection, logistics & treatment requirements for WEEE - Part 2-2: Treatment requirements for WEEE containing CRTs and flat panel displays |
| | TS 50625-3-3 | Collection, logistics & treatment requirements for WEEE - Part 3-3: Specification for depollution - WEEE containing CRTs and flat panel displays |
| Temperature exchange equipment | EN 50625-2-3 | Collection, logistics & treatment requirements for WEEE - Part 2-3: Treatment requirements for temperature exchange equipment |
| | TS 50625-3-4 | Collection, logistics & treatment requirements for WEEE - Part 3-4: Specification for depollution - Temperature exchange equipment |
| Large household cooling appliances | EN 50574-1 | Collection, logistics & treatment requirements for end-of-life household appliances containing volatile fluorocarbons or volatile hydrocarbons |
| | TS in development | - |
| Photovoltaic panels | EN 50625-2-4 | Collection, logistics & treatment requirements for WEEE - Part 2-4: Treatment requirements for photovoltaic panels |
| | TS 50625-3-5 | Collection, logistics & treatment requirements for WEEE - Part 3-5: Specification for depollution - photovoltaic panels |

Taking a closer look at the presence of brominated flame retardants (BFR), in order to comply with the strict safety regulations for EEE (Dupont et al., 2016), a significant quantity is applied to plastics, entailing recycling difficulties. As previously mentioned, currently only the overall quantity of plastics containing brominated flame retardants is reported by the compliance schemes.

According to EN 50625-1 - Annex A, plastic fractions extracted from waste streams consisting of temperature exchange equipment and large household appliances shall be deemed free of BFRs and may be recycled. Conversely, plastic fractions from other types of appliances shall be considered to contain brominated flame retardants (BFRs). Thus, the operators must separate fractions with bromine concentration higher than 2000 mg/kg, or expected to be higher than 2000 mg/kg, or if it is not declared (Hennebert and Filella, 2018).

The content of plastics in the e-waste generated in France has doubled from 2000 to 2015 (Huisman et al., 2017). The European Electronics Recyclers Association (EERA) estimated that in the average mix of WEEE plastics in the EU, only 5 to 10 % consists of plastics with BFR (European Electronics Recyclers Association, 2018). Thus, a significant quantity could potentially be recycled.

Aiming to boost recycling of plastics from WEEE and ensure compliance with the European recovery targets, it is crucial to improve materials tracking. Thus, tracking the amount of plastics that could be potentially recycled by the official chain is an interesting piece of information. In this context, we suggest including an indicator that evaluates the proportion of plastics that may contain BFRs in the total share of plastics treated by the official schemes per category (Equation 4-5). The data required to calculate this indicator is already individually monitored by the compliance schemes.

$$PBR_f = \frac{WBr}{WP}$$

| | |
|---|---|
| $PBR_f$ | rate of plastics that may contain brominated flame retardants in the WEEE category; |
| $WBr$ | weight of plastics that may contain brominated flame retardants treated by the official schemes; |
| $WP$ | overall weight of plastic fractions in WEEE collected by the official schemes. |

Equation 4-5. Rate of plastics that may contain brominated flame retardants in WEEE collected.

### 4.5.6. Reuse and recycling: need for individual indicators and targets

As discussed in section 4.4, to promote reuse, it is important to track recycling and reuse rates separately. Thus, we suggest splitting the "recycling and reuse rate" indicator proposed by the WEEE Directive (Equation 4-6 and Equation 4-7).

Currently, the system boundaries for WEEE Directive indicators are from the collection by the official schemes until the end-of-waste status is achieved for the recoverable fractions or disposal of non-recoverable fractions. For these two indicators we suggest considering the same system boundary of the WEEE Directive.

$$RuR_f = \frac{\sum_{i=1}^{n1} WRu_i}{\sum_{i=1}^{n1} WC_i}$$

| | |
|---|---|
| $RuR_f$ | reuse rate of the WEEE category; |
| $n1$ | number of different UNU-Keys related to the WEEE category; |
| $WRu$ | weight of e-waste reused; |
| $WC$ | weight of e-waste collected by the official schemes. |

Equation 4-6. Reuse rate by the official schemes per WEEE category.

$$RR_f = \frac{\sum_{i=1}^{n1} WR_i}{\sum_{i=1}^{n1} WC_i}$$

$RR_f$ recycling rate of the WEEE category;

$n1$ number of different UNU-Keys related to the WEEE category;

$WR$ weight of fractions sent to recycling facilities;

$WC$ weight of e-waste collected by the official schemes.

Equation 4-7. Recycling rate by the official schemes per WEEE category.

### 4.5.7. Treatment rate based on the waste generated approach

According to EN 50625-1 – Annex C, recycling and recovery rates determination shall start with the untreated WEEE, and end when the end-of-waste status for fractions is achieved, or with the final recovery or disposal of fractions. Thus, following the arguments previously presented for the indicator "collection rate based on the WEEE generated approach" (see section 4.5.2), we suggest calculating the rate per type of treatment by the official schemes, taking into account a wider system boundary. As presented in Equation 4-8, we suggest an assessment based on the WEEE generated, instead of focusing only on the e-waste collected by the official schemes. Differently to the previous indicator (section 4.5.6), this approach allows estimation of the flows deviated into complementary flows.

$$TR_f = \frac{\sum_{i=1}^{n1} WT_i}{\sum_{i=1}^{n1} WG_i}$$

$TR_f$ treatment rate (recycling, reuse, energy recovery or disposal) of the WEEE category;

$n1$ number of different UNU-Keys related to the WEEE category;

$WT$ weight of e-waste that is sent for the different types of treatment (recycling, reuse, energy recovery or disposal);

$WG$ weight of e-waste generated.

Equation 4-8. Treatment rate (recycling, reuse, energy recovery or disposal) based on WEEE generated approach.

### 4.5.8. Recycling rate per target element

WEEE officially reported as collected by the official schemes are usually efficiently recycled in terms of overall weight. However, high recycling rates are a consequence of base metals recycling (e.g. ferrous metals, aluminum, copper), which form most of the weight of e-waste (Vidal-Legaz et al., 2018). Therefore, complementary to the recycling rate calculated based on the overall weight (Equation 4-8), we propose to calculate the recycling rate per target element (Equation 4-9). This indicator intends to highlight the importance of taking a closer view of the sorting and recycling efficiency of some target elements present in small quantities, among which the CRMs. With this indicator, together with the collection rate per target element (Equation 4-3), it is possible to identify the influence of collection and recycling losses per target element.

$$RR_e = \frac{WR_e}{WG_e}$$

Where:

$$WR_e = \sum_{i=1}^{n1} WC_i \times \sum_{j=1}^{n2} CH_{eij} \times (1 - SG_j) \times \sum_{k=1}^{n3} ST_{ik} \times PR_{ek}$$

| | |
|---|---|
| $RR_e$ | recycling rate of the target element in the WEEE category; |
| $WR_e$ | weight of target element recycled; |
| $WG_e$ | weight of target element generated (calculated based on Equation 4-3); |
| $n1$ | number of different UNU-Keys related to the WEEE category; |
| $WC$ | weight of e-waste collected by the official schemes; |
| $n2$ | number of components and materials present in the different products; |
| $CH_e$ | element content per component and materials in the products; |
| $SG$ | scavenging rate per component; |
| $n3$ | number of treatment paths for all the products; |
| $ST$ | share of the treatment scenario per product; |
| $PR_e$ | efficiency of the recycling of the target element per treatment scenario; |
| $WG$ | weight of e-waste generated. |

Equation 4-9. Recycling rate per target element.

The target element recycling efficiency ($PR_e$) is calculated based on the pre-processing efficiency rate (see Table 4-11 for an example on sorting efficiency rate of screens) and the efficiency of the final recycling operation (see Table 4-12 for efficiency rate of a few elements). As indicated in Equation 4-9, it depends on the treatment path of each product (UNU-key). In some cases, the device is treated directly in an end-processing and it is assumed no losses with pre-processing (e.g. tablets that are treated in a copper smelter).

$RR_e$ quantifies only the elements recycled into secondary raw materials with same or similar properties (closed or semi-closed loop recycling approach). Thus, it does not capture alloying elements in secondary metals recycled in an open-loop approach (often with lesser quality and reduced functionality).

## 4.6 Case study: focus in screens flows in France

In order to validate the indicators proposed in the previous section, in this section we apply them to a case study in France. For most indicators, the results are focused on category II (screens), but for some of the indicators, we present results for a wider range of categories.

The screens category comprises different equipment, including both cathode-ray tubes (CRT) and flat panel displays (FPD) as presented in Table 4-5. It is selected as a case study in this thesis since the switch from CRT to FPD clearly represents the influence of a change in technology on the volumes and types of material available for collection and recycling (Cucchiella et al., 2015). In certain analyses we present aggregated data on CRT (including both TVs and monitors) and FPD (TVs, monitors, laptops and tablets). For some indicators, results for different years are discussed, and in others, a focus in a specific year (e.g. 2018) is presented to enable a more detailed discussion.

Table 4-5. UNU-keys in WEEE category II.

| UNU-keys | Description |
|---|---|
| 0303-01 | Tablets |
| 0303-02 | Laptops |
| 0308 | Cathode Ray Tube Monitors |
| 0309 | Flat Panel Display Monitors (LCD, LED) |
| 0407 | Cathode Ray Tube TVs |
| 0408 | Flat Panel Display TVs (LCD, LED, Plasma) |

### 4.6.1. Screens placed on the market and waste generated

Regarding the monitoring of EEE placed on the market, in addition to the assessment based on the overall weight and number of pieces per (W)EEE categories, we suggest a monitoring per type of equipment (UNU-Keys). Currently, the French national register does not track this type of data, therefore we used data from the "Waste over time" tool developed by Statistics Netherlands which, as previously mentioned, is the source of POM data of the WEEE Calculation Tool (Van Straalen et al., 2016). In the mentioned source, the UNU-key 0303 presents aggregated results for the tablets (030301) and laptops (030302). We considered a share of 7% and 93% of the total weight, respectively, based on data of surveys performed in the EU during the ProSUM project (Huisman et al., 2017).

The comparison between the amount of equipment in pieces and the weight placed on the market allows us to track changes in WEEE categories due to the evolution of technologies and changes in consumers' behavior (Figure 4-8). As can be noticed, the overall weight is decreasing over the years, while the total quantity of pieces/ items is rising. In 2018, the total weight of EEE POM was two times lower than in 2000, but the number of items was multiplied by a factor of 3.8. LCD TVs and monitors have replaced CRT TVs and monitors with an associated reduction in weight. Until the beginning of the 2000s, TVs represented the biggest share of POM both in terms of weight and pieces, but since 2007 tablets and laptops account for more than 50% of screens POM per pieces. However, in terms of weight, FPD monitors account for the biggest share of screens purchased. These changes in EEE POM impact the type of WEEE to be treated in the following years.

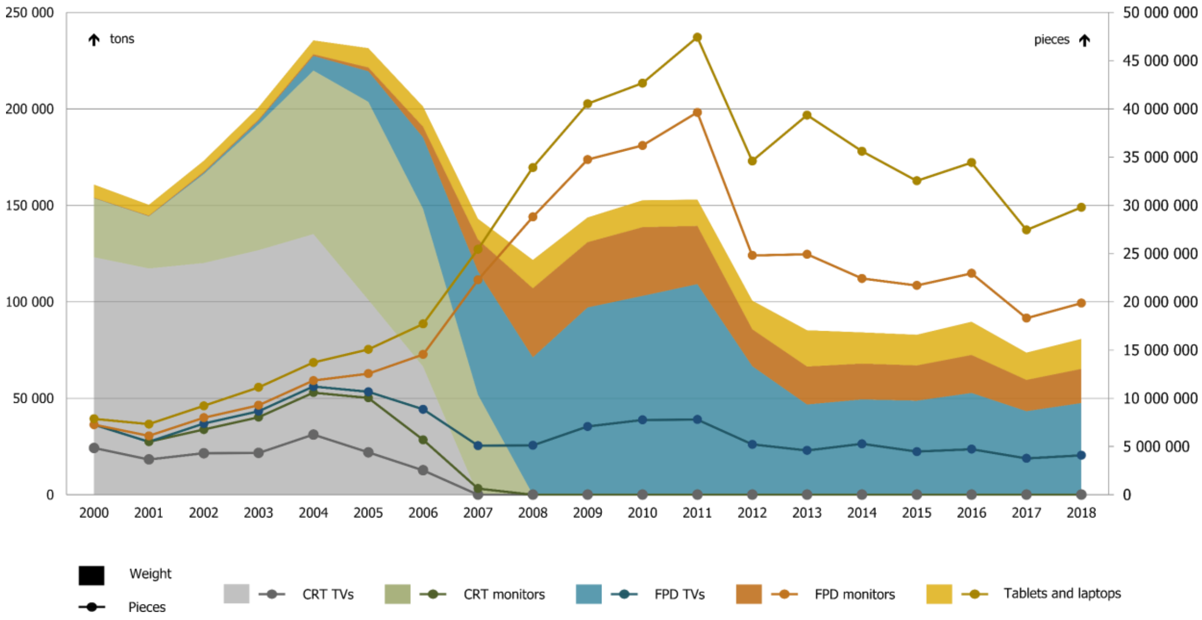



Figure 4-8. Screens placed on the market per UNU-Keys in France.

Source: data from Van Straalen et al. 2016, and graph inspired in Huisman et al. 2017.

#### 4.6.1.1. Follow-up of screens placed on the market

In 2018, 28.8 kg/inhabitant or 14 pieces/inhabitant of EEE were placed on the market, an increase of 10% and 24% in comparison to 2015 (see Figure 4-9 calculated according to Equation 4-1).

Even if the electronic components are getting smaller, resulting in a lower average weight per device, in number of pieces more EEE are purchased per household, thus the overall weight of EEE POM is also increasing. The screens category represented 4% and 3% in terms of weight and pieces, respectively, of the overall amount of EEE placed on the market.

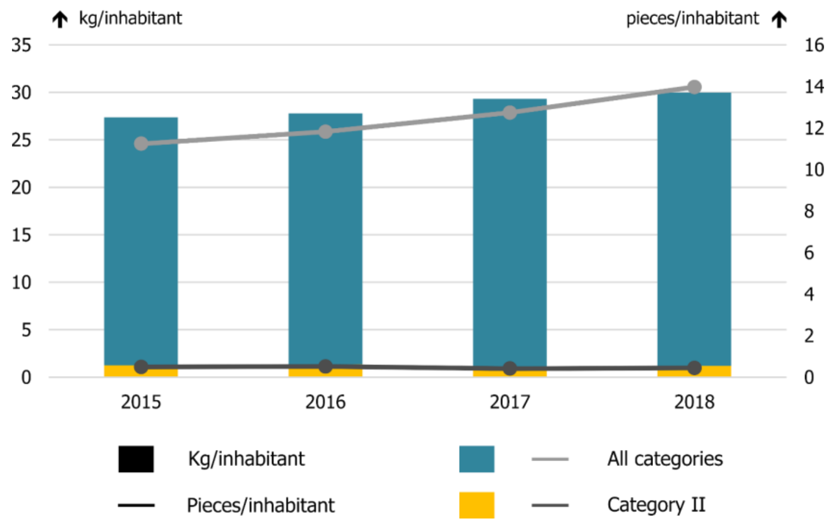



Figure 4-9. Follow up of EEE placed on the market in France (kg and pieces/inhabitant).

#### 4.6.1.2. Waste generated – focus in screens in France

The waste generated approach is the basis of some of the indicators proposed in this work and is calculated with the WEEE Calculation. The waste generated is not an indicator, but it is an important parameter for part of the indicators.

The composition of e-waste generated depends on the products placed on the market, and on their life spans (including use and stock). In terms of weight, 170,754 tons of screens were generated in 2018, 14.5% less than in 2012. Table 4-6 presents the share of WEEE generated per UNU-Keys in France for the category II.

Table 4-6. Share of UNU-keys in screens generated in France.

| UNU-keys | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 |
|---|---|---|---|---|---|---|---|
| Tablets | 0.4% | 0.4% | 0.4% | 0.5% | 0.5% | 0.6% | 0.6% |
| Laptops | 5.2% | 5.6% | 5.9% | 6.4% | 6.9% | 7.5% | 8.2% |
| CRT Monitors | 24.4% | 23.8% | 23.1% | 22.4% | 21.6% | 20.8% | 19.8% |
| FPD Monitors | 7.0% | 8.7% | 10.2% | 11.4% | 12.3% | 12.9% | 13.3% |
| CRT TVs | 51.1% | 47.0% | 43.0% | 39.3% | 35.6% | 32.0% | 28.4% |
| FPD TVs | 11.9% | 14.6% | 17.2% | 20.0% | 23.0% | 26.2% | 29.6% |

As can be noticed, CRT screens still represent around 50% of waste generated in category II. The quantities are decreasing annually and will go down to zero in a few years. This is interesting information for the compliance schemes, because nowadays they only monitor the share of CRT and FPD collected by the official schemes.

### 4.6.2. Collection by the official schemes

#### 4.6.2.1. Collection rate based on the WEEE generated approach

Figure 4-10 presents the collection rate for screens category ($CR_f$) in France from 2012 to 2018, based on POM approach (current approach) and WEEE generated (Equation 4-2). The official schemes in France comply largely with the targets based on the POM approach. This high performance is explained by the fact that the official schemes collect mostly CRT screens and that since 2008 only FPD screens, which are much lighter and more valuable for reuse and export, are placed on the market.

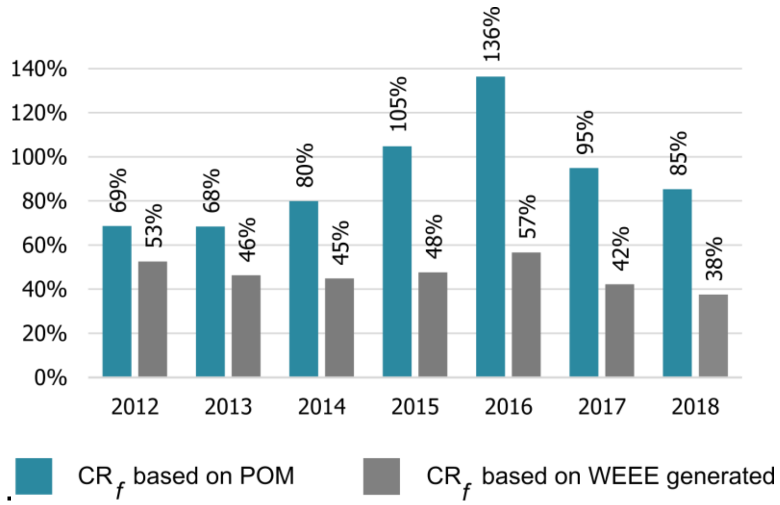



Figure 4-10. Comparison of different approaches for calculating WEEE category II collection rate.

As presented in **Chapter 2**, in France, household e-waste is collected per waste stream. Thus, to estimate the amount of e-waste collected per category, the compliance schemes perform characterization campaigns every year. This assessment allows the conversion of data on the amount collected and treated per waste stream, in the amount collected per category. Nevertheless, there is no detailed assessment in terms of breakdown of the flow per type/category of products and, or their clustering per UNU-key (required to calculate some of the indicators suggested in this thesis).

Due to lack of official data on the share of WEEE collected per UNU-Keys, we combined data from ProSUM and Ecologic. For the EU 28+2, the ProSUM project determined an average share of UNU-keys collected per Member State (Huisman et al., 2017). According to data provided by Ecologic, until 2016 screen flows was mostly comprised of CRT screens (90%). The share of UNU-keys considered in this study is presented in Table 4-7.

Figure 4-11 presents the weight of screens collected per UNU-Keys (t), as well as the collection rate based on the waste generated approach (according to Equation 4-2) per type of screen (FPD and CRT). CRT screens have a higher collection rate in comparison to FPD screens, but, except in 2016, lower than the target that will be in force by 2019. In terms of weight, the amount of FPD collected in 2018 (15,306t) is three times higher than the collection in 2012 (5,138t), and should increase in the following years.

Table 4-7. Share of UNU-keys in screens collected in France by the official schemes.

| UNU-keys | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 |
|---|---|---|---|---|---|---|---|
| Tablets | ≤ 0,1% | ≤ 0,1% | ≤ 0,1% | 0,1% | 0,1% | 0,1% | 0,2% |
| Laptops | 0,5% | 0,5% | 0,6% | 0,9% | 1,2% | 1,5% | 2,3% |
| CRT Monitors | 13,3% | 13,3% | 13,1% | 12,7% | 12,3% | 11,9% | 10,7% |
| FPD Monitors | 2,1% | 2,1% | 2,5% | 3,8% | 5,0% | 6,2% | 9,5% |
| CRT TVs | 81,7% | 81,7% | 80,9% | 78,1% | 75,7% | 73,1% | 66,3% |
| FPD TVs | 2,4% | 2,4% | 2,9% | 4,4% | 5,7% | 7,2% | 11,0% |

Source: based on data from Ecologic and ProSUM.

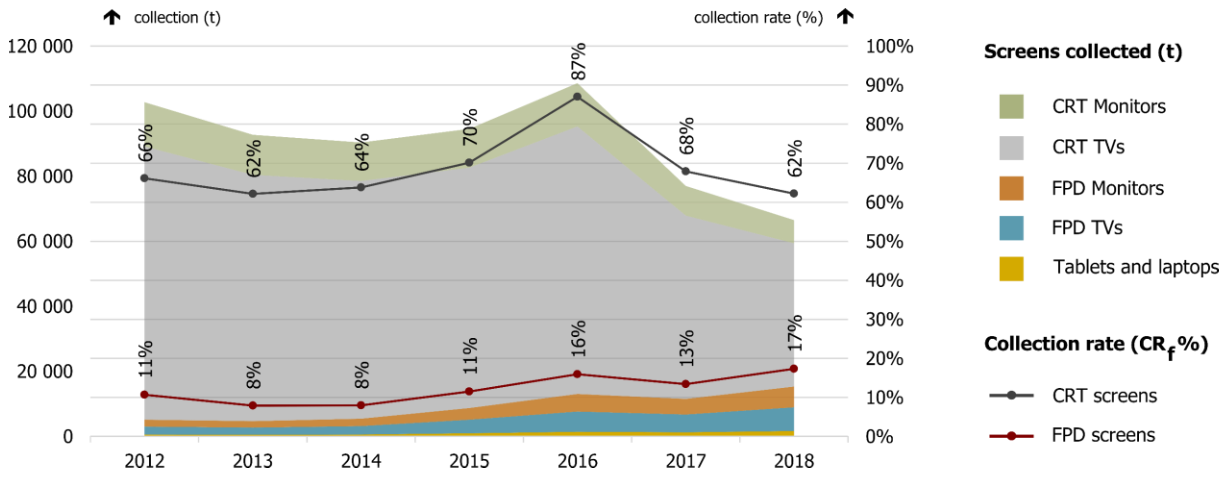



Figure 4-11. Collection performance per type of WEEE screens.

#### 4.6.2.2. Collection rate per target element

As described in Equation 4-3, to calculate the collection rate per target element we require: the data to calculate collection rate based on the overall weight; the scavenging rate of components and materials in screens (see Table 4-1); and the element content per component and materials in the UNU-Keys are required (Table 4-8).

The collection rate per element ($CR_e$) in 2017 is presented in Figure 4-12, applying Equation 4-3 and including the batteries (nowadays treated in a different EPR scheme in France).

All targeted elements have a lower performance than the collection rate calculated based on the overall weight of screens ($CR_f$) calculated with Equation 4-2). The elements with a higher collection rate ($CR_e$) are those significantly present in CRT screens (Cu and Mg, both with 37%). Even if CRT devices contain more than 50% of the Mg and Cu generated, as previously shown in Figure 4-11, CRT's do not have a 100% collection rate. Consequently, in the case of copper, only 53% of the copper generated in CRT devices is collected by the official schemes, the remaining travels with complementary product flows and components scavenging. This, combined with the low collection of FPD devices by the official schemes (only 13% of the copper generated by FPD is collected), explains the limited resulting collection rate of the element.

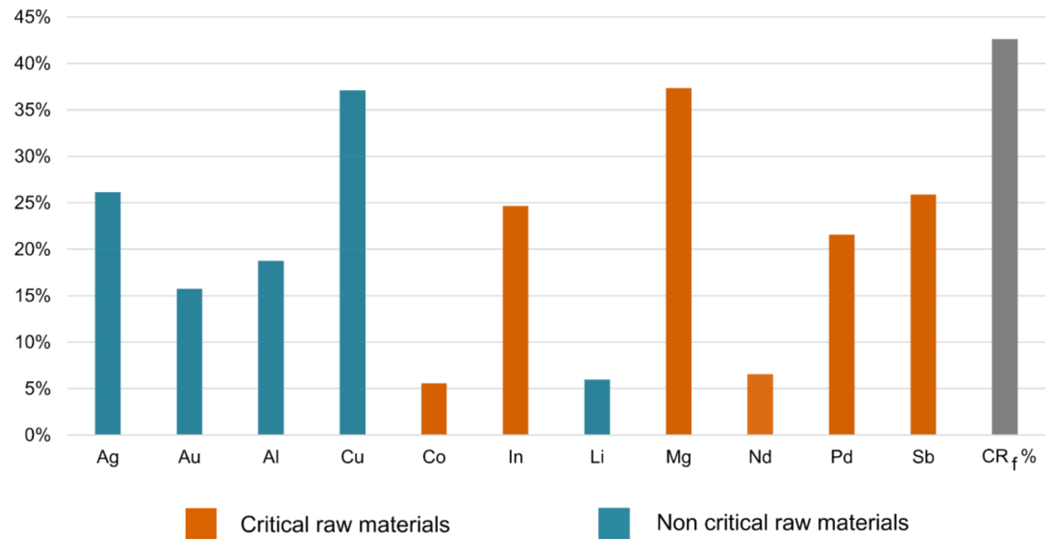



Figure 4-12. Collection rate per target element for WEEE category II in France (2017).

From a weight-based perspective, aluminum and copper account for around 80% of the elements targeted in the study. The critical raw materials represent less than 20% of WEEE generated and collected by the official schemes. Silver, gold, neodymium and palladium represent only about 0,1% of the overall weight of WEEE generated and collected by the official schemes (about 22t and 4t in 2017, respectively). Nevertheless, precious metal recycling is the main economic driving force for WEEE recycling (>95%) (Charles et al., 2017; Ueberschaar et al., 2017). The concentration of copper, gold, silver and palladium in the e-waste stream is significantly higher compared to primary ore grades in conventional mining operations (Kumar et al., 2017).

Figure 4-13 presents the palladium content in WEEE generated per UNU-keys in France in 2017, and the subsequent share in WEEE collected by the official schemes and complementary flows. Besides being a precious metal, palladium is also in the list of critical raw materials for the EU.

Table 4-8. Screens composition (mg/kg UNU-Keys).

| UNU keys | Materials/ components | Ag | Al | Au | Co | Cu | Hg | In | Li | Mg | Nd | Pd | Sb |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 030301 Tablets | Cables | 0,00 | 309,79 | 0,00 | 0,00 | 3 029,04 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | Drives | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 54,02 | 0,00 | 0,00 |
| | PCB | 96,38 | 2 513,11 | 26,41 | 46,07 | 35 424,50 | 0,00 | 0,00 | 0,00 | 154,88 | 3,91 | 0,58 | 0,00 |
| | Cu/Fe coils, motors | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | Display LCD | 174,13 | 5 724,62 | 44,66 | 0,00 | 2,97 | 0,23 | 34,40 | 0,00 | 533,15 | 0,00 | 7,88 | 0,00 |
| | Display CRT | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | Display TFT | 0,00 | 0,00 | | | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | Al alloys | 0,00 | 175 108,07 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 4 477,45 | 0,00 | 0,00 | 0,00 |
| | Cu alloys | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | Mg alloys | 0,00 | 8 281,73 | 0,00 | 0,00 | 3,01 | 0,00 | 0,00 | 0,00 | 91 149,22 | 0,00 | 0,00 | 0,00 |
| | Background lighting CFL | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | LED | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| 030302 Laptops | Cables | 0,18 | 11,07 | 0,05 | 0,00 | 4 156,58 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,01 | 0,00 |
| | Drives | 0,00 | 0,00 | 0,00 | 15,34 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 505,61 | 0,00 | 0,00 |
| | PCB | 120,26 | 3 098,41 | 53,80 | 1,67 | 27 472,38 | 0,00 | 6,03 | 0,52 | 63,69 | 9,24 | 14,49 | 317,22 |
| | Cu/Fe coils, motors | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | Display LCD | 151,61 | 4 984,26 | 38,89 | 0,00 | 2,58 | 0,20 | 29,95 | 0,00 | 464,19 | 0,00 | 6,86 | 0,00 |
| | Display CRT | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | Display TFT | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | Al alloys | 0,00 | 132 182,77 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 3 379,87 | 0,00 | 0,00 | 0,00 |
| | Cu alloys | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | Mg alloys | 0,00 | 3 727,78 | 0,00 | 0,00 | 1,36 | 0,00 | 0,00 | 0,00 | 41 028,15 | 0,00 | 0,00 | 0,00 |
| | Background lighting CFL | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,39 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |

| | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | LED | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| 0308 CRT Monitors | Cables | 0,00 | 1 273,68 | 0,00 | 0,00 | 6 148,96 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | Drives | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | PCB | 16,05 | 4 283,60 | 1,18 | 0,00 | 16 865,60 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,30 | 204,91 |
| | Cu/Fe coils, motors | 0,10 | 23,66 | 0,35 | 276,94 | 20 412,01 | 0,00 | 0,00 | 0,00 | 0,00 | 0,88 | 0,12 | 0,10 |
| | Display LCD | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | Display CRT | 6,33 | 5 998,18 | 0,58 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 7 136,03 | 0,00 | 0,00 | 26 655,21 |
| | Display TFT | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | Al alloys | 0,00 | 15 696,20 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 401,35 | 0,00 | 0,00 | 0,00 |
| | Cu alloys | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | Mg alloys | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | Background lighting CFL | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | LED | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| 0309 FPD Monitors | Cables | 0,00 | 1 259,55 | 0,00 | 0,00 | 6 195,63 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | Drives | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | PCB | 17,59 | 3 693,17 | 5,73 | 5,99 | 26 227,59 | 0,00 | 2,03 | 0,40 | 0,00 | 0,36 | 0,96 | 30,69 |
| | Cu/Fe coils, motors | 0,09 | 23,35 | 0,00 | 1,02 | 0,83 | 0,00 | 0,00 | 0,00 | 0,00 | 0,07 | 0,00 | 0,00 |
| | Display LCD | 84,31 | 2 771,64 | 21,62 | 0,00 | 1,44 | 0,11 | 7,33 | 0,00 | 258,13 | 0,00 | 3,81 | 0,00 |
| | Display CRT | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | Display TFT | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | Al alloys | 0,00 | 58 625,06 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | Cu alloys | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | Mg alloys | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 1 499,02 | 0,00 | 0,00 | 0,00 |
| | Background lighting CFL | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,35 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | LED | 0,05 | 12,78 | 0,03 | 0,002 | 26,85 | 0,0005 | 0,01 | 0,00 | 0,00 | 0,00 | 0,003 | 0,01 |

| | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Cables | 0,00 | 0,00 | 0,00 | 0,00 | 10 254,08 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 162,70 |
| | Drives | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | PCB | 23,18 | 3 149,90 | 1,11 | 1,68 | 6 710,41 | 0,00 | 7,25 | 0,26 | 45,70 | 0,64 | 1,18 | 142,74 |
| | Cu/Fe coils, motors | 0,04 | 8,70 | 0,00 | 9,66 | 14 031,59 | 0,00 | 0,00 | 0,00 | 0,00 | 0,32 | 0,00 | 250,50 |
| | Display LCD | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| 0407 CRT TVs | Display CRT | 7,33 | 6 950,16 | 0,67 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 8 268,61 | 0,00 | 0,00 | 30,89 |
| | Display TFT | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | Al alloys | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | Cu alloys | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | Mg alloys | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | Background lighting | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | LED | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | Cables | 0,00 | 0,00 | 0,00 | 0,00 | 1 767,85 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | Drives | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | PCB | 29,24 | 10 320,32 | 2,97 | 0,75 | 14 945,65 | 0,00 | 2,73 | 0,41 | 0,00 | 0,64 | 0,69 | 28,52 |
| | Cu/Fe coils, motors | 0,09 | 33,76 | 0,00 | 5,32 | 7,98 | 0,00 | 0,00 | 0,00 | 0,00 | 0,11 | 0,00 | 0,00 |
| | Display LCD | 8,23 | 270,58 | 2,11 | 0,00 | 0,14 | 0,01 | 1,63 | 0,00 | 25,20 | 0,00 | 0,37 | 0,00 |
| 0408 FPD TVs | Display CRT | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | Display TFT | 4,22 | 10 029,65 | 0,00 | 0,00 | 3,69 | 0,00 | 26,74 | 0,00 | 471,09 | 0,00 | 0,00 | 0,00 |
| | Al alloys | 0,00 | 34 105,95 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 872,08 | 0,00 | 0,00 | 0,00 |
| | Cu alloys | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | Mg alloys | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | Background lighting | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 | 0,00 |
| | LED | 1,45 | 380,47 | 0,84 | 0,07 | 799,05 | 0,01 | 0,42 | 0,00 | 0,00 | 0,00 | 0,10 | 0,26 |

Source: ProSUM Project (Huisman et al., 2017)

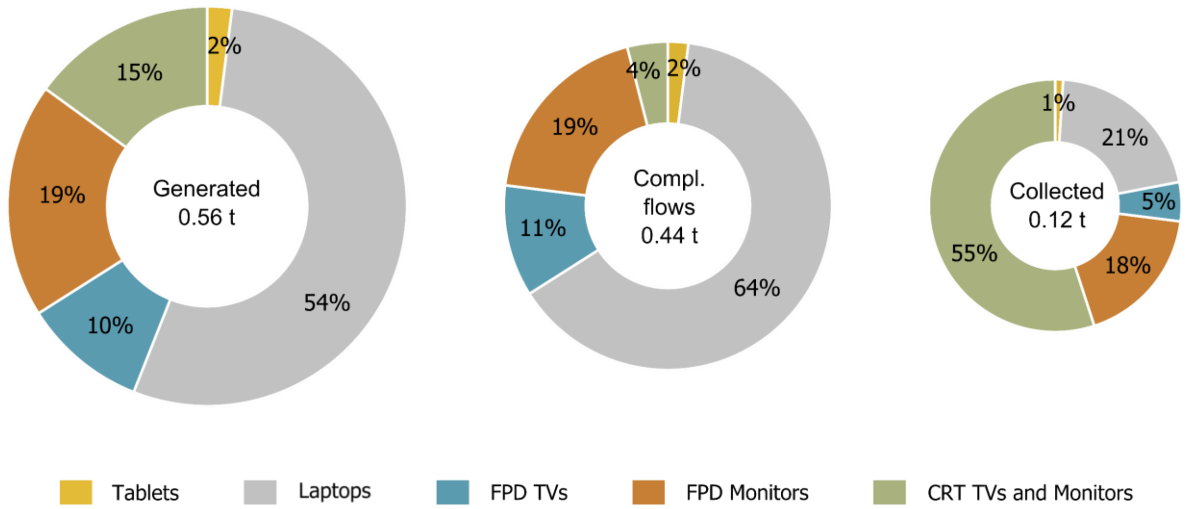



Figure 4-13. Total weight and share of palladium per UNU-keys in screens in France (2017).

Palladium and other precious and/or critical materials (e.g. Au, Ag, In and Nd) are largely present in FPD screens (>75% of screens generated content). As it can be noticed in Figure 4-13, more than 50% of the palladium collected nowadays is from PCBs of CRT devices. Due to the low collection of FPD, most of palladium, as well as other precious and/or critical materials, are not collected by the official schemes and consequently, are not yet available in reported recycling.

Scavenging of higher-valued components (e.g. PCB, drives) from the official collection channels considerably reduces the value of the reported material content (Huisman et al., 2017). In 2017, scavenging accounted for up to 6% of the amount diverted from official schemes. The impact of component scavenging per target element is presented in Figure 4-14. Precious metals like gold and palladium, are mostly concentrated in PCB in FPD screens (above 50%). Based on EERA data (see Table 4-1), it is considered that 5% of PCB collected is scavenged.

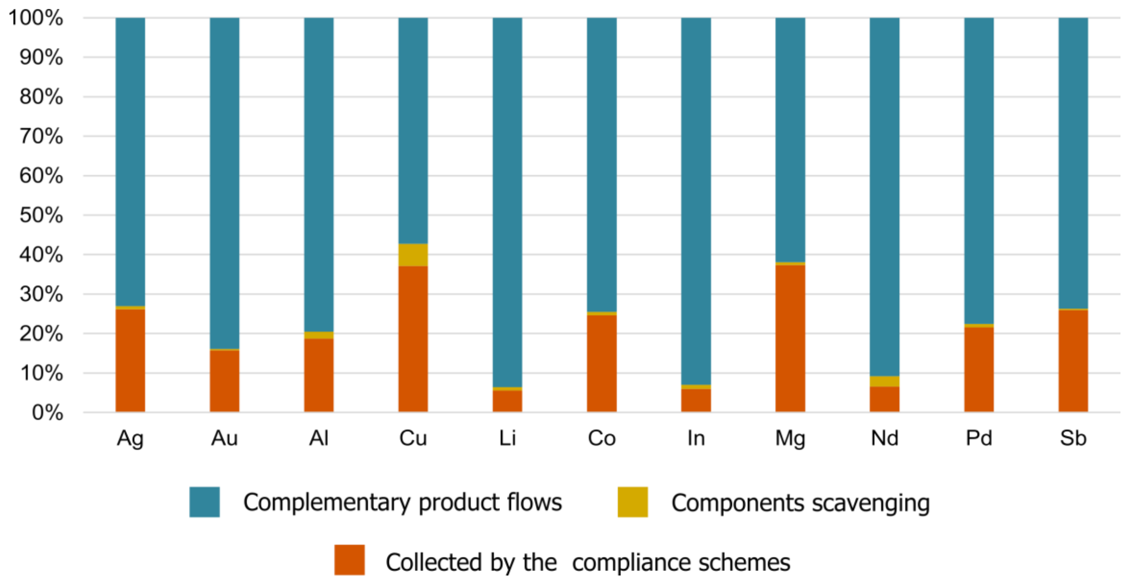



Figure 4-14. WEEE generated per target element in WEEE category II in France (2017).

As previously mentioned, most of the FPD are diverted into complementary flows, consequently, the impact of component scavenging seems negligible when compared with the total generation, but it is significant to the volume collected by the official schemes. Copper is more affected by component scavenging, since it is present in PCB, but also in other components of different types of screens that have similar or higher scavenging, like coils (copper wire) and cables (Figure 4-15).

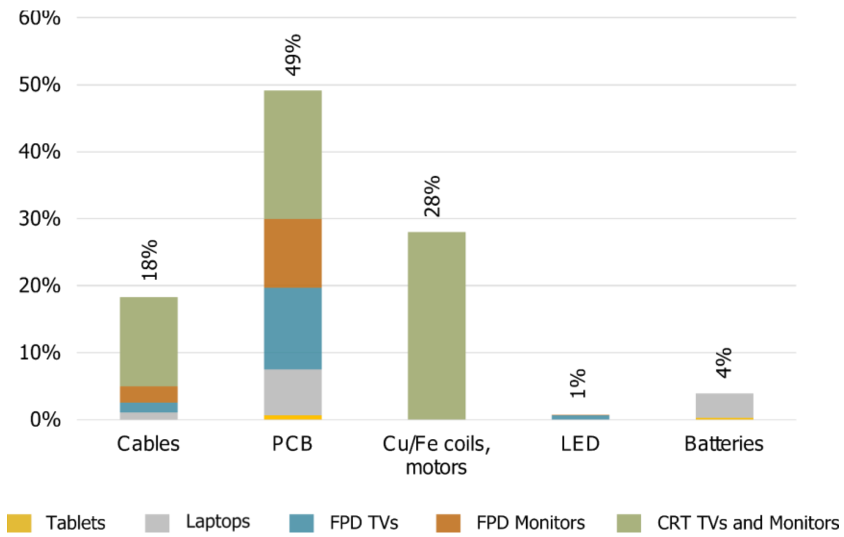



Figure 4-15. Distribution of Cu per component per UNU-keys in WEEE category II generated (2017).

#### 4.6.2.3. French departments complying with collection targets

Table 4-9 present the ratio of departments complying with the collection target (for all categories) fixed by the WEEE Directive (according to Equation 4-4). As can be noticed, the change in the collection target from 4 kg/inh, to 45% (POM approach) led to a drop in the number of departments complying with the target.

Table 4-9. French departments complying with the collection target.

| Details | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 |
|---|---|---|---|---|---|---|---|
| Rate | 93% | 94% | 94% | 95% | 82% | 79% | 84% |
| Collection target | 4 kg/inh | 4 kg/inh | 4 kg/inh | 4 kg/inh | 45% | 45% | 45% |
| Higher (kg/inh) | 17.4 | 15.9 | 15.9 | 14.8 | 16.7 | 17.4 | 21.0 |
| Lower (kg/inh) | 0.3 | 0.4 | 0.4 | 1.1 | 5.0 | 6.0 | 5.2 |

Moreover, there is high variability among the departments. The department with the highest collection rate is Tarn-et-Garonne, in the southwest of France, and the lowest is Mayotte (an overseas department in the Indian Ocean). Taking a closer look at Metropolitan France, Paris is the only department with collection lower than the first WEEE Directive target – and it is densely populated and economically important. Figure 4-16 presents the population and the collection performance of the France metropolitan departments in 2018.

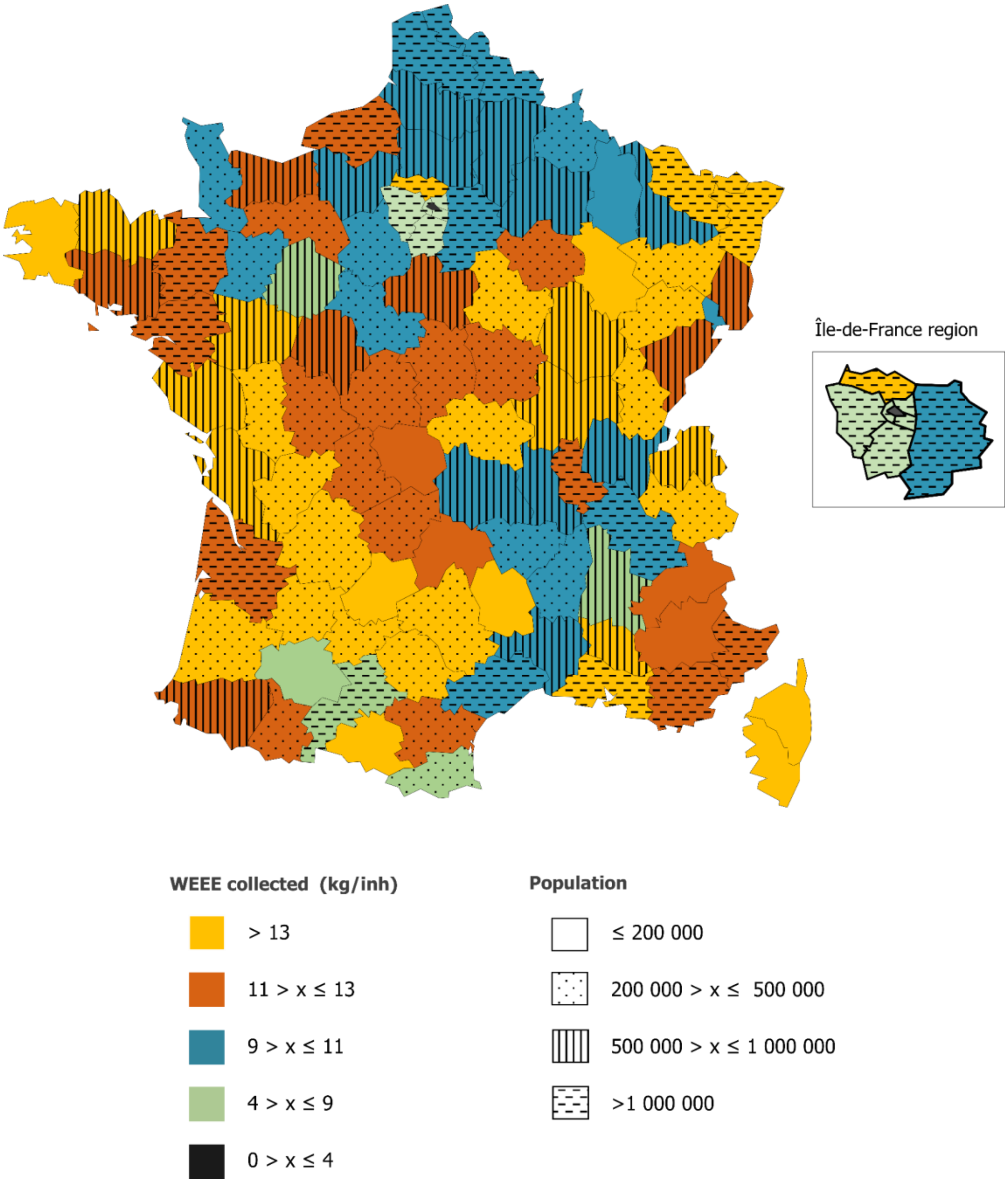



Figure 4-16. Collection per inhabitant and population in French departments (2018).

Population density seems to have a correlation with the quantity collected, but it does not seem to be the only factor that influences it. Among the departments with the lowest collection performance (lower than 9 kg/inh), 65% have a population of over one million inhabitants. In this group, there are four departments which account for 11% of the population of France and are part of the Île-de-France region. However, most of the departments with a population of over one million inhabitants have a collection rate above 11 kg/inh (30%), including one department of the Île-de-France region (Vald'Oise). We believe that other analyses comparing WEEE collection and socio-economic factors (e.g. number of collection points, purchasing power, educational level) may help to identify strategies to improve collection at a local level. This could support compliance with future targets at the country level.

### 4.6.3. Treatment scenarios

In order to calculate the recycling rate per target element by the official schemes (Equation 4-9), various treatment paths of the current practices are defined based on literature and expert judgment (researchers, compliances schemes and recyclers). The treatment scenarios considered in the study are summarized in Table 4-10 and detailed in the following paragraphs.

#### 4.6.3.1. Televisions and monitors

Prior to treatment, the screens are sorted into CRT and FPD due to differences in technology as well as the cleaning requirements. CRT screens include both televisions and monitors and there are no significant differences in their treatment. The treatment steps are dismantling of casings, removal of the electron gun, pressurization of the CRT and, in accordance with the EU WEEE Directive, removal of the CRT and PCBs. The remaining electronics undergo shredding and separation into fractions that are sent to further recovery processes (Huisman et al., 2008; Monier et al., 2013a).

FPD screens comprise a wider range of equipment types and include different types of technologies: LCD, light emitting diode (LED) and plasma. Nowadays in France, the FPD waste stream is mostly comprised of mercury backlights containing LCD’s (Froelich and Sulpice, 2016). Mercury free LED screens are the natural evolution of LCD screens and, since they were put on the market later on, it is expected that their volume in screen waste streams will increase in the following years (Cucchiella et al., 2015).

The first step of FPD treatment is manual sorting according to the different technologies since LCD require specific clean-up to remove the mercury backlights. Then, treatment is followed by dismantling (manual or mechanical) and removal of high-value components and/or components required by the EU WEEE Directive (e.g. PCBs and backlighting lamps that contain mercury). Remaining parts of the screens are forwarded to a shredder then further separation of ferrous, aluminum, copper, plastics and smaller PCB fractions (Monier et al., 2013a). Recently, automatic LCD recycling processes started to be implemented in Europe. In these processes the screens are shredded at negative air pressure for mercury extraction, followed by sorting of fractions (aluminum, PCB, PMMA, mixed plastics and glass) (Horta Arduin et al., 2019).

Table 4-10. Treatment paths by the official schemes considered in the case study in France.

| UNU-keys | Path A | | Path B | | Path C | |
|---|---|---|---|---|---|---|
| | Description | Share (%) | Description | Share (%) | Description | Share (%) |
| 030301 Tablets | Shredding of the whole device via cross-flow shredder and fractions sorting | 85 | Deep-level manual dismantling of the subassemblies | 5 | Direct treatment in copper smelter after removal of the battery | 10 |
| 030302 Laptops | After removal of the battery and display panel, the entire device is treated in a medium shredder followed by fractions sorting | 50 | Manual dismantling of battery display panel and high value components. Remaining parts are forwarded to a medium shredder followed by fractions sorting | 50 | - | - |
| 0308 0407 CRT TVs and monitors | Manual dismantling of the casings, removal of components (e.g. CRT and PCBs). Remaining parts are forwarded to a medium shredder followed by fractions sorting | 100 | - | - | - | - |
| 0309 0408 FPD TVs and monitors | Manual dismantling of LCD screen and backlighting systems, as well as of high value components. Remaining parts are forwarded to a medium shredder, followed by fractions sorting | 60 | Mechanical dismantling of LCD screen, backlighting systems and PCBs greater than 10 $cm^2$. Remaining parts are forwarded to a medium shredder, followed by fractions sorting | 30 | Shredding of the whole device at negative air pressure followed by fractions sorting | 10 |

Source: Cucchiella et al., 2015; Horta Arduin et al., 2019; Huisman et al., 2008; Monier et al., 2013; Tecchio et al., 2018

### 4.6.3.2. Tablets and laptops

The number of tablets and laptops in the reported collection flow reaching recycling facilities is still limited (see Table 4-7), and much lower than the amount estimated in the WEEE generated calculations (see Table 4-6). This difference is due to scavenging of products and reuse outside the official schemes. The treatment paths considered in this study are based on interviews with recyclers in Europe and identified as representative scenarios for the EU (Tecchio et al., 2018).

According to the WEEE Directive, batteries of tablet and laptops should be removed prior to e-waste treatment, but it is possible that some tablets are directly shredded and mixed with other WEEE. Laptops can follow two main processing routes after battery and display panel removal: a first one based on the shredding and sorting of fractions; and a second one including a medium-depth manual dismantling of components prior to shredding and mechanical sorting. The manual dismantling of tablets and laptops is more effective in terms of material recovery. However, it entails in higher labor costs.

#### 4.6.3.3. Pre-processing and recycling

As can be noticed from the treatment description, dismantling is typically followed by a size reduction step (Işıldar et al., 2018). After the waste is shredded, it is separated into different fractions by various combinations of sorting techniques, such as magnetic separators, optical sorting technologies, inductive sorting technologies, eddy current separators, magnetic ballistic separators, etc. (Bigum et al., 2012; De Meester et al., 2019).

Due to technological limitations, output fractions of sorting processes are not pure. For this reason, a transfer coefficient matrix is used to determine the real quantity of elements (e) in the sorting fractions (f*) (see sorting and shredding rate in Equation 4-9). The transfer coefficient matrix used is presented in Table 4-11.

Table 4-11. Transfer coefficient matrix.

| Fractions (f*) | Al | Cu | PCB | Plastics | Fe | Other |
|---|---|---|---|---|---|---|
| Aluminium | **88.1%** | 0.5% | 3.5% | 7.3% | 0.0% | 0.6% |
| Copper | 0.0% | **85.0%** | 0.0% | 10.0% | 0.0% | 5.0% |
| PCB | 0.2% | 0.1% | **93.1%** | 6.6% | 0.0% | 0.0% |
| Plastics | 12.1% | 2.8% | 3.4% | **77.2%** | 1.2% | 3.3% |
| Steel | 0.0% | 1.4% | 13.8% | 16.8% | **65.9%** | 2.2% |
| Other fractions | 0.0% | 6.3% | 2.9% | 90.9% | 0.0% | 0.0% |

Source: Horta Arduin et al., 2019 based on data provided by MTB Recycling[14].

The transfer coefficient can change significantly between different companies according to the technologies used, as well as the dismantling and shredding processes prior to mechanical sorting. Lastly, the efficiency of final operations that recover secondary raw materials from the sorted components and fractions (gate-to-gate approach) is also considered.

[14] MTB Recycling is a recycling operator and manufacturer of recycling machines in France.

This performance is commonly called "recycling rate" in the literature, but to distinguish it from the recycling rate indicators discussed in this study, it is defined here as the "efficiency of the final recycling operation" (see Equation 4-9). The efficiencies considered in this study are presented in Table 4-12.

Table 4-12. Efficiency of the final recycling operation per target material.

| Elements | Efficiency rate |
|---|---|
| Aluminum | 98% |
| Antimony (PCB) | 80% |
| Cobalt (PCB and batteries) | 90% |
| Copper | 70% |
| Copper (PCB) | 95% |
| Copper (batteries) | 90% |
| Silver (PCB) | 95% |
| Gold (PCB) | 97% |
| Indium (PCB) | 10% |
| Palladium (PCB) | 95% |

Source: Ardente and Mathieux, 2012; Chancerel and Marwede, 2016; Tecchio et al., 2018

### 4.6.4. Screens treatment indicators

#### 4.6.4.1. Clean-up efficiency

Among the different components and substances that must be removed and treated separately, we suggest a clean-up indicator focused on the presence of brominated flame retardants (BFR) in WEEE collected by the official schemes (Equation 4-5).

The overall weight of plastics with BFR that are extracted from e-waste treated are given in the annual report released by ADEME (e.g. in 2017 18,115 t of plastics that potentially contain BFR were extracted from household WEEE according to Deprouw et al., 2018). Additionally, in the annual characterization campaign performed by the compliance schemes (undisclosed data), the e-waste composition is estimated per collection waste stream, including an estimation on the quantity of plastics that may contain BFR.

We used data regarding the composition of screens based on the annual characterization campaign performed by the compliance schemes (Fangeat, 2018) and Ecologic data on the share of FPD and CRT collected (Table 4-7). Based on Equation 4-5 , we estimate that 21% of the plastics collected from screens in 2017 may contain flame retardant. Thus, most of the plastic fraction (79%) could be potentially recycled, if there is the technology available to recover the different polymers, and if there is a market that uses these secondary raw materials to ensure the profitability of the WEEE chain.

As presented in Figure 4-17, this result is also influenced by the higher presence of CRT screens in the officially collected waste. It can be noticed that the presence of BFR is higher in FPD, but in 2017 they represented only 15% of the WEEE collected (see Table 4-7). When the FPD collected increases, consequently, the share of plastics with BFR will rise.

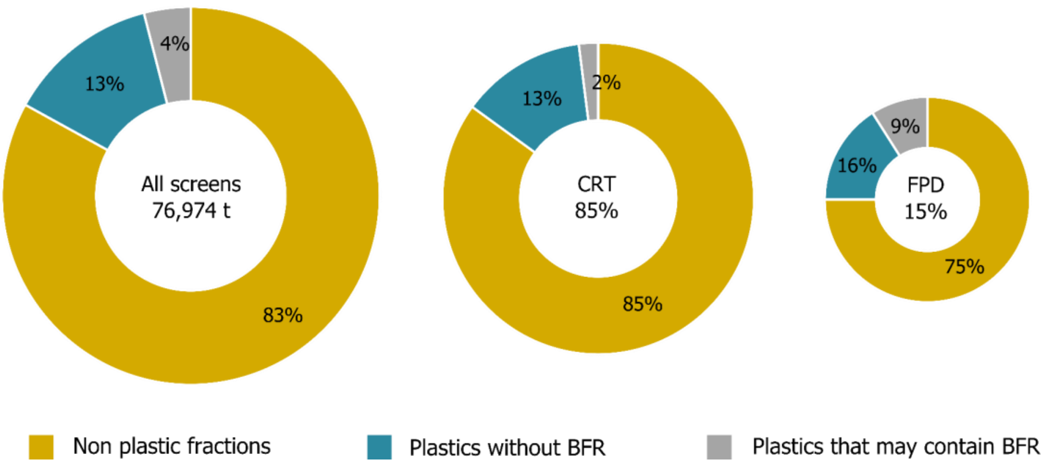



Figure 4-17. Composition of screens collected in France (2017).

#### 4.6.4.2. Reuse by the official chain

Figure 4-18 shows the rates of WEEE "prepared for reuse" in France from 2012 to 2017 by category, including both household and professional e-waste. It is calculated as the amount of e-waste reused divided by the total waste treated by the official schemes (Equation 4-6).

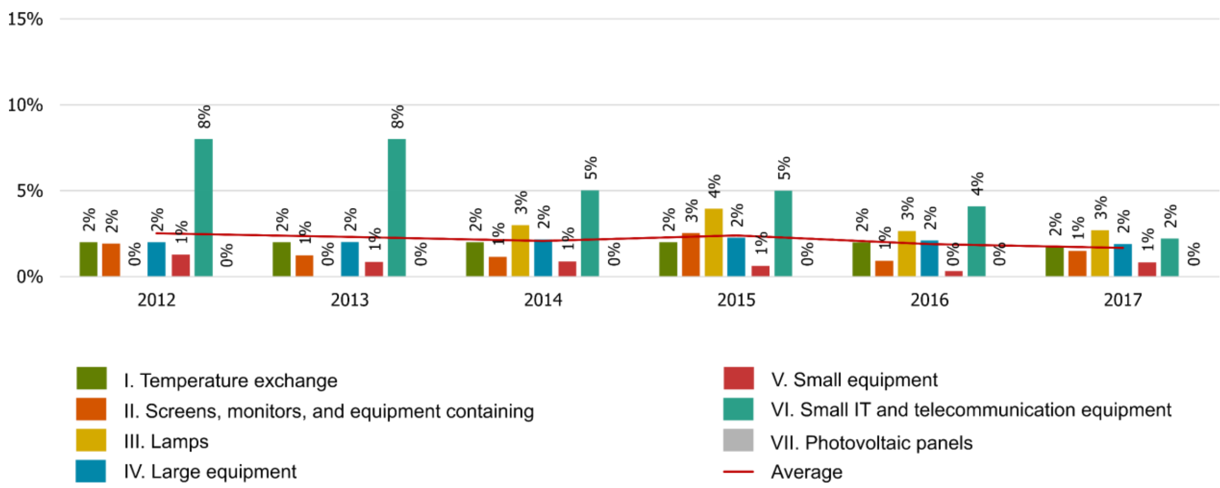



Figure 4-18. Reuse rate per (W)EEE categories from 2012 to 2017 in France.

In 2017, it can be seen that, for most of the (W)EEE categories, reuse represented less than 2% of the total weight treated by the compliance schemes. WEEE category II is among the categories with lowest reuse in the past 2 years (around 1%). Moreover, for certain categories such as IT and telecommunication equipment, reuse has sharply decreased.

These results imply that France, like other countries in the EU (see Figure 4-6), has potential to improve preparation for reuse. In fact, more equipment is reused nowadays outside the official schemes. However, they are not included in the scope of the indicator.

### 4.6.4.3. Recycling by the official chain

Figure 4-19 shows the recycling rate of WEEE in France from 2012 to 2017 per category, including both household and professional e-waste.

The system boundary of the indicator includes from the e-waste collection by the official schemes to the end-of-waste status for the fractions (Equation 4-7). It can be noticed that the recycling rate per WEEE categories, as well as the average of the categories have been stable in past years (ranging between 72% to 94%).

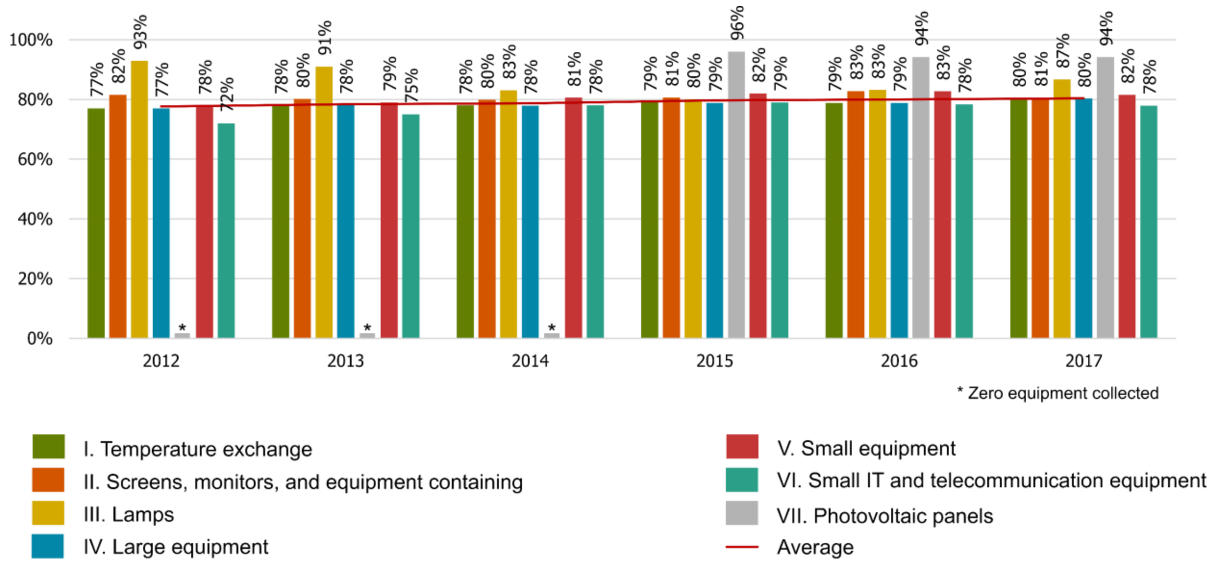



Figure 4-19. Recycling rate per (W)EEE category from 2012 to 2017 in France.

### 4.6.4.4. Treatment rate based on the waste generated approach

In this study, we also suggest calculating the treatment rate per WEEE category (recycling, reuse, energy recovery or disposal) as the ratio of materials sent to treatment facilities, divided by the waste generated (Equation 4-8). This approach complies with the goal of the study of understanding and quantifying the flows of WEEE screens better.

Figure 4-20 presents the repartition of different treatments and the percentage of complementary flows per WEEE category. When comparing this result with the previous indicator (recycling by the official schemes), the size of the complementary flows is obvious, and it focuses attention on what this may represent in terms of economic and environmental impacts.

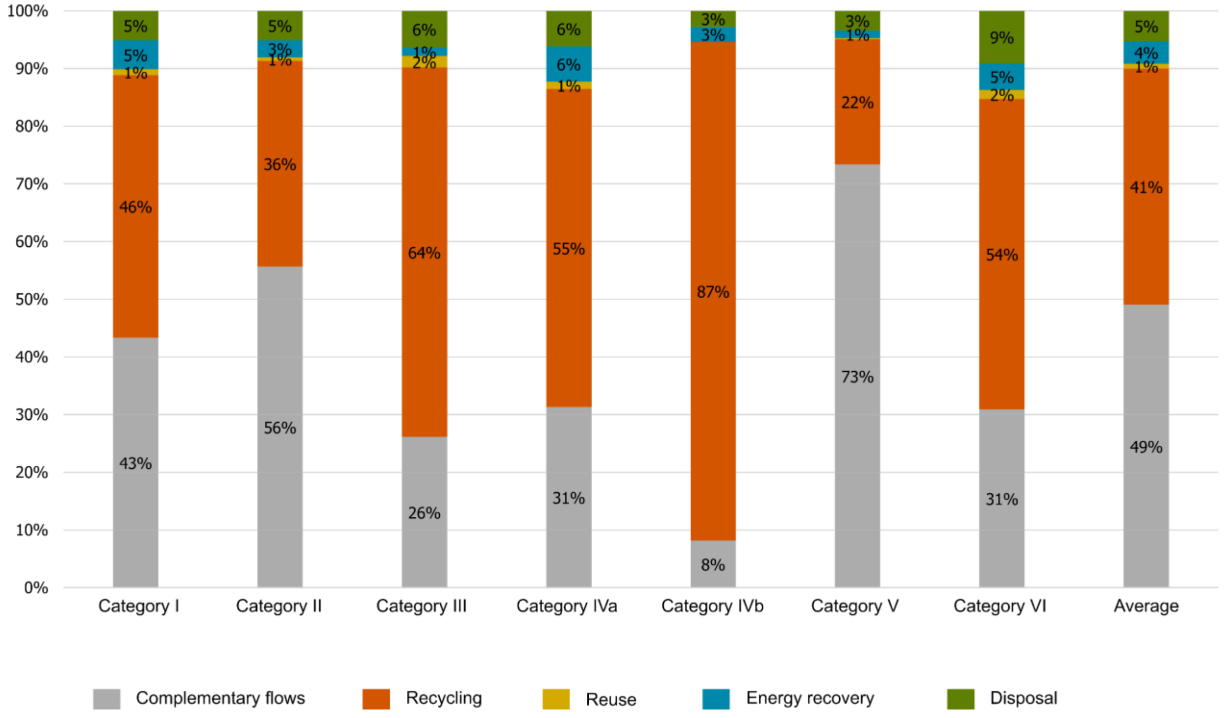



Figure 4-20. WEEE generated by type of treatment in France (2017).

We can also identify the categories more susceptible to complementary flows (category VI followed by the category II, focus of the case study), and that recycling is the main treatment in France, as previously pointed out. Photovoltaic panels (PV panels), classified as category IVb in the EU and category VII in France, is the category where official collection is most efficient, but is still important to stress that only a small quantity is generated and collected in comparison to other categories (it represented 2% of WEEE generated and less than 1% of e-waste collected by the official schemes). The percentage of PV recycling is surprisingly high, that is probably explained by the fact that the system boundary of the data reported by the official schemes ends in the end-of-waste status. According to the last report released by ADEME (Deprouw et al., 2018), 95% of the weight of the PV panels is recovered (including metals, plastics and glass), however it is not clear what is the percentage of materials recycled nor the quality of the secondary raw materials. For example, the recovery of critical materials, such as indium, in PV is still under research and is not implemented at an industrial scale.

#### 4.6.4.5. Recycling rate per target element

Figure 4-21 presents, from 2012 to 2017, the recycling rate (in %) and the weight of the target elements recycled calculated in accordance with Equation 4-9. The results consider only the elements recycled by the official schemes. In 2017, 16,649 tons of target elements were generated by the screens category in France, 4,508 tons were collected by the official schemes and 2,697 tons were recycled.

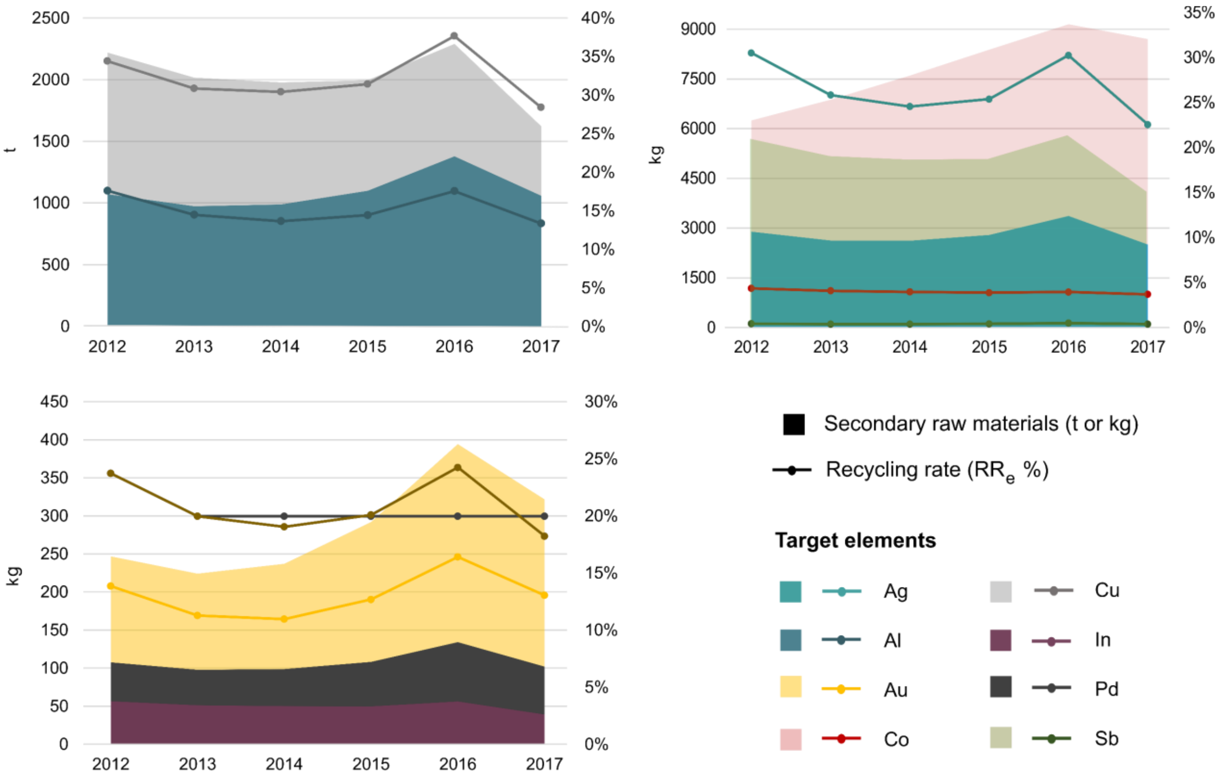



Figure 4-21. Recycling rate of target elements ($RR_e$) in WEEE category II (2012-2017).

It is assumed, based on the literature review, that no industrial-scale process is currently available to recover neodymium, magnesium and lithium from batteries – hence their recycling rate is equal to zero and is not presented in the graph. In terms of weight, aluminum and copper accounted for 99% of target elements recycled. Copper is the target element with the highest recycling rate (28% in 2017, corresponding to 1,623 tons for WEEE screens in France).

Due to the absence of data, we assumed the same UNU-Keys composition over time, as well as the same treatment performance. Thus, when comparing the differences in recycling performance from 2012 to 2017, besides the impact of the overall weight collected by the official schemes, the influence of the type of screens collected over time can be identified.

Even if the total weight of screens collected in 2017 decreased in comparison to previous years (Figure 4-11), the amount of gold recycled increased, driven by the increase of FPD collected. The same effect can be observed for elements like aluminum and palladium, even despite a decrease in the overall weight of screens collected. Nonetheless, the elements recycling rate has decreased due to the growth of FPD generated in contrast to their modest collection.

## 4.7 Discussion on case study results

Follow-up of EEE placed on the market (POM) is an interesting indicator to monitor EEE consumption over time. Moreover, together with life span data, it allows the amount of WEEE generated to be calculated. The indicators currently used for WEEE chain performance assessment do not include waste generated within the system boundary. Therefore, the complementary flows are neither quantified nor officially reported.

In contrast to the collection rate results based on POM approach, the results based on WEEE generated approach indicate there is a need to improve collection performance. In the case of screens, we observed a significant difference in the collection performance per type of screen (CRT and FPD). In 2017, FPD represented 47% of WEEE generated. However, 85% of screens collected by the official schemes are composed of cathode ray tube. Consequently, the collection rate of elements mostly present in FPDs, and more specifically in tablets and laptops, is three to four times lower than elements present in CRT devices.

As presented in section 4.5.3, all targeted elements have lower performance than the collection rate calculated based on the overall weight of screens, among other things due to the low collection of FPD and scavenging. Changes in the type of screens collected will impact collection rate results, as well as the type of materials recovered due to differences in screens compositions, which is why it is important to have indicators to monitor the quality of e-waste collected by the official schemes.

Currently, 84% of departments comply with the WEEE Directive collection target. This result will potentially drop when the new target is imposed from 2019. Collection performance is quite diverse among the French departments, and an understanding of the influence of socio-economic factors seems necessary to improve the collection performance at local, and country level.

E-waste clean-up is essential to avoid uncontrolled dispersion of regulated substances in secondary raw materials. Among the controlled substances, we propose an indicator to quantify the share of plastics containing brominated flame retardants. Monitoring plastics with BFR provides, at the same time, information regarding depollution and data on the potential of recycling. In the case of screens, with the increase of FPD collection, the amount of plastics with BFR will rise. Nowadays, potentially only the plastic fraction without BFR is sent to recycling facilities for further treatment. However, the fraction containing BFR has the potential to be a secondary source of antimony (critical raw material for the EU). Approximately 52% of antimony produced is used in flame retardants, of which 70% is used for EEE plastics (Dupont et al., 2016).

The separate results of recycling and reuse rates (calculated according to the WEEE Directive system boundary) stress the need to develop reuse. On average, less than 2% of e-waste collected by the official schemes in sent for reuse. Further studies should be developed in order to establish compulsory, separate targets for reuse and recycling.

Based on the results of the case study, we identified four main obstacles to improving treatment rates: (1) low collection by the official schemes, especially of FPD screens; (2) difficulties in manual or mechanical pre-processing; (3) absence of recycling processes on an industrial scale; (4) low economic incentives compared to recycling costs. Except for the low collection by the official schemes that affects the performance rates of all elements targeted, the challenges are different depending on individual elements and their content in different components and equipment.

Part of the equipment and components diverted from the official schemes is potentially recycled and reused in unofficial flows. In addition to not contributing to official targets, some of them are recycled/reused outside EU, and this results in economic and material losses for the EU. Assuming that all components scavenged from screens in France in 2017 were sent to recycling, it is estimated that around 450 t of target elements were recycled (mostly Cu and Al, but also of elements like Au and Pd that have high intrinsic value).

A lack of efficient pre-processing mechanical technologies also limits recycling. Design plays an important role to improve the recycling yields with easier dismantling and sorting of components (Ardente et al., 2014). Magnets in drives, for example, are often embedded and glued in place within the products making their extraction and recycling difficult (Lixandru et al., 2017).

Moreover, for some elements, the costs (economic and/or environmental) of recycling as a pure element may be higher than raw material production. For example, magnesium recycling requests a high purity fraction, which is not achieved via mechanical separation because magnesium has similar physical properties to aluminum (Tecchio et al., 2018). Nowadays, most of the end-of-life magnesium is recycled as part of the aluminum value stream in alloys (Mathieux et al., 2017). Seeing the limits of recycling all elements present in WEEE, a metric to consider open-loop recycling should be analyzed in a future study.

Low recycling rates of CRMs are, among others, related to the fact that processes to recover some CRMs are only found in pilot plants (Peeters et al., 2018; Zimmermann and Gößling-Reisemann, 2013). Several potential recycling processes for the neodymium available in nickel-plated neodymium magnets (NdFeB) and indium in TFT panels have been described in the literature, but none of them has been developed commercially due to low productivity and high costs (Padhan et al., 2017; Ylä-Mella and Pongrácz, 2016). The increase of FPD share in screens collected may contribute to reaching the critical mass necessary to make the recovery process of certain metals such as indium economically viable (Ardente et al., 2014). Hence the need for an indicator to monitor the changes in WEEE POM in order to organize e-waste chains for the new types of products and materials to be recovered.

High recycling costs and low economic and regulatory incentives tend to discourage recycling of certain materials. Besides neodymium and indium previously described, lithium present in batteries and Al-Li alloys is also generally not considered for recycling because virgin material is relatively low-priced (Meshram et al., 2019).

Overall, improving the recycling of CRMs from WEEE could decrease the EU demand for raw materials and eventually even reduce the criticality of some of them (Blengini et al., 2017). The assessment of economic viability and the potential environmental benefits of the envisaged recycling strategies for CRMs could stimulate investment in industrial scale processes. Nevertheless, information about the content of CRM in (W)EEE is crucial for viability studies (Peeters et al., 2018). The recent approval of ecodesign requirements for servers and data storage products (European Commission, 2018) making the indication of cobalt content in batteries and neodymium in the hard-drives compulsory, can be seen as an important step towards data availability and transparency to support tracking CRM in (W)EEE.

The indicators proposed in this study could support compliance schemes and enable policy makers to have more information on the grade and quality of the materials present in WEEE generated, collected and treated by the official schemes.

## 4.8 Conclusions

This chapter presents nine technical indicators to improve monitoring of WEEE collection and treatment performance, as well as the methods and data required. Table 4-13 summarizes the indicators, as well as the sections in which the indicators are validated with the screens case-study.

Table 4-13. Summary of the indicators proposed in Chapter 4.

| | Technical indicators | Equations | Validation with case study |
|---|---|---|---|
| 1. | EEE placed on the market per inhabitant | Equation 4-1 | Section 4.6.1.1 |
| 2. | Collection rate based on the WEEE generated approach | Equation 4-2 | Section 4.6.2.1 |
| 3. | Collection rate per target element | Equation 4-3 | Section 4.6.2.2 |
| 4. | Rate of departments complying with the WEEE collection target | Equation 4-4 | Section 4.6.2.3 |
| 5. | Rate of plastics that may contain brominated flame retardants in WEEE collected | Equation 4-5 | Section 4.6.4.1 |
| 6. | Reuse rate by the official schemes per WEEE category | Equation 4-6 | Section 4.6.4.2 |
| 7. | Recycling rate by the official schemes per WEEE category | Equation 4-7 | Section 4.6.4.3 |
| 8. | Treatment rate based on WEEE generated approach | Equation 4-8 | Section 4.6.4.4 |
| 9. | Recycling rate per target element | Equation 4-9 | Section 4.6.4.5 |

This set of indicators aims to reply to the second research question of this thesis presented in **Chapter 1** regarding the identification of the most suitable technical indicators to support quantifying the collection and treatment performance beyond the overall mass-based approach.

The feasibility for compliance schemes and policy-makers to run the indicators suggested in this work on a regular basis depends on the availability of data to calculate them. Indicators 1, 4, 5, 6 and 7 require data that is currently monitored by the compliance schemes and ADEME. Thus, they could be immediately adopted.

On the other hand, to put into practice indicators 2 and 8, once the system boundary of the indicators includes waste generated, it is necessary to adopt the calculation of WEEE generated according to WEEE Calculation Tool. The parameters are duly detailed in the WEEE Calculation Tool, and it would be possible to develop an interface between this tool and the current databases used by ADEME and compliance schemes to calculate the waste generated based on updated data on the EEE placed on the market in France.

Lastly, indicators 3 and 9 require more precise data. The system boundary of this indicators, besides including waste generated, requires information on WEEE composition (ProSUM Project data is used in the case study), as well as in components scavenging (we considered data of a study performed by EERA). Additionally, indicator 9 requires data on the treatment performance that is not monitored by current compliances schemes. As previously discussed, the current focus of the WEEE chain is on ensuring the treatment of WEEE until the end-of-waste status is fulfilled. Consequently, information on the final recovery of secondary raw materials is not monitored.

The results of the case study confirm that monitoring the performance per target element (indicators 3 and 9) in comparison to overall weight indicators allows an overview of the volume and the grade of potential secondary resources. It also allows identification of the main challenges to improving materials recovery towards a circular economy. Furthermore, these indicators can support future legislation developments to set incentives to recover specific elements, in particular, those contained in low or trace amount, among which we find critical raw materials. Thus, in the following paragraphs, we suggest some strategies to obtain the missing data.

As previously mentioned, in France, the compliance schemes perform annually characterization campaigns to quantify the amount of WEEE collected per category, as well as the average composition per waste stream. An assessment of the share of equipment (e.g. UNU-keys) in WEEE collected, as well as of the components scavenged could be included in the annual characterization. That would allow the changes in the type of equipment collected by the official schemes to be followed and the quantifying of the materials potentially available for recycling.

Regarding (W)EEE composition, a collaborative work between producers and recyclers to find a cost-effective and efficient way of producing and sharing product and component data would support data gathering, as well as recycling infrastructure development (Downes et al., 2017). However, this is not an issue that can be solved in the short term. A possible solution in the short/medium term is to use ProSUM data. According to information available in Urban Mine Platform, specific data per appliance type/UNU-key is available upon request (paid services). Nevertheless, this is not a solution for the long term because electronic composition changes rapidly with advances in technology.

Lastly, to obtain more detailed data related to e-waste treatment, it is necessary to establish compulsory mechanisms to enforce waste treatment providers to report data to the compliance schemes (e.g. through requirements in the national specifications included in the Ministerial Orders for household and professional WEEE). To achieve materials circularity, the WEEE chain needs to be more transparent, and this includes improving WEEE flows monitoring.

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## Chapter 5 | Environmental impacts and benefits of WEEE chain

*"No dia que a universidade me deu um diploma e uma ciência que estava longe de carregar no cérebro confesso que me senti ao mesmo tempo enganado e orgulhoso.*[15] *"*
Machado de Assis, Memórias Postumas de Brás Cubas (1881).

### 5.1 Indicators and methods to assess the environmental performance

As stressed in **Chapter 2**, according to the specifications for household and professional Waste Electrical and Electronic Equipment (WEEE) provided by the Environmental Code, the approved compliance schemes must provide indicators related to the environmental impacts of WEEE schemes. However, the authorities specify neither the method nor the scope of the indicators. Thus, in this chapter, the potential of Life Cycle Assessment (LCA) methodology and its results for defining indicators for the WEEE chain is explored.

LCA is widely recognized as the most advanced method for calculating the potential environmental performance of products on a quantitative and comparable basis (EC-JRC, 2010a). The method choice is grounded on the vast use of LCA in recent years to assess impacts of (W)EEE, and because it is recognized both by industry and academic institutions as a well-established method to evaluate the potential environmental impacts of product systems.

The principle of LCA is outlined by ISO standards ISO 14040 (2006a) and ISO 14044 (2006b), however, there is still room for different methodological choices (Arushanyan et al., 2014). Before applying the LCA to a case study, a literature review of this method and its application to WEEE assessment are presented, in sections 5.2 and 5.3 respectively. Due to the importance of data quality, and the fact that LCA practitioners often use secondary data (for at least part of the inventory), an overview of the databases that provide data on the end-of-life (EOL) of WEEE is presented in section 5.4.

In order to quantify the impacts of recycling in France, in section 5.5, an LCA study for WEEE screens in France is presented. Based on the conclusions of the case study for WEEE screens, as well as the results of the screens studies identified in the literature, the potential and difficulties of using LCA to monitor the impacts of the WEEE chain are discussed in section 5.6.

Section 5.7 presents the environmental indicators that we suggest for the WEEE chain performance assessment, including the data required, followed by the conclusion of this chapter (section 5.8).

[15] On the day that the university gave me a degree and a science that I was far from carrying in my brain I confess that I felt both deceived and proud.

## 5.2 Life Cycle Assessment

LCA is defined as the compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle (ISO, 2006a). The environmental impacts are the consequences of human intervention on the environment, either physical, chemical or biological, such as resource extraction, emissions and land use (Guinée et al., 2002).

The ISO framework separates an LCA into four phases, as presented in Figure 5-1. It is an iterative method illustrated by the both direction arrows in this figure. Thus, the development of one of the phases may give new approaches to another, and even require a redefinition of the previous phases. The phases are briefly described below based on different references (EC-JRC, 2010a, 2010b, 2010c; ISO, 2012, 2006a, 2006b). A detailed technical guidance to the ISO standards is presented in the ILCD Handbook published by the European Commission - Joint Research Centre. The overall objective of this handbook is to provide a common basis for consistent and quality-assured life cycle data and robust studies (EC-JRC, 2010a).

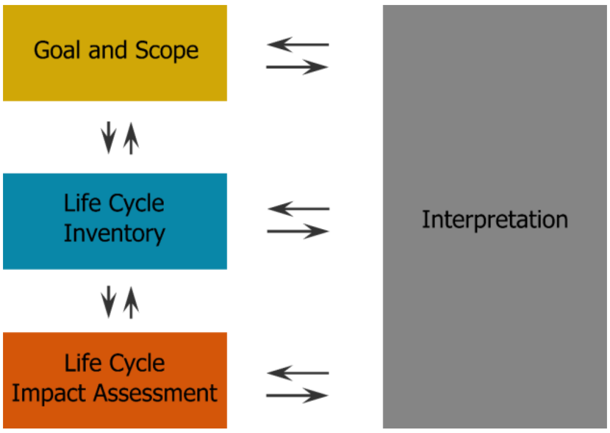



Figure 5-1. LCA phases according to ISO (2006a).

### Goal and Scope

Definition of the product system to be assessed, and other important aspects such as the functional unit of the study (quantified performance of a product system for use as a reference unit to which all flows and impacts are going to be referred) and the system boundaries (geographical, temporal and physical interface between a product or system and the environment or other product systems).

### Life Cycle Inventory (LCI)

It consists of analyzing each process in relation to the functional unit, including related inputs and outputs. This phase involves data collection, compilation and calculation procedures to quantify the exchanges within the studied product system and between the product system and the environment. The source of data may include direct measurements, and secondary data from scientific literature and LCI databases. This phase requires the greatest efforts and resources of an LCA. Further details on this phase are presented in section 5.4

### Life Cycle Impact Assessment (LCIA)

The inputs and outputs collected and reported in the previous phase are translated into impact indicators. During this phase, LCA practitioners use validated LCIA methodologies to evaluate the magnitude and significance of the potential environmental impacts for the system under study. Different LCIA methodologies (e.g. ReCiPe, CML, TRACI, ILCD, among others) have been developed with diverse approaches to dealing with modeling the effect of the emissions on the environment.

For this phase, the ISO standards define three mandatory stages and two optional stages (Rodriguez-Garcia and Weil, 2016):

- Mandatory: selection of impact categories, classification of the different emissions based on the impact category they affect, and quantification of the impact associated with each flow by multiplying the amount of the flow by a characterization factor (characterization);
- Optional: comparison of the potential impacts of the system with those caused by a reference scenario (normalization) and aggregation of all impact categories into a single score using subjective criteria (weighting).

LCIA methods include different impact categories such as eutrophication, global warming or human toxicity, and characterization factors for different substances. Consequently, the choice of the LCIA method already implies the three mandatory stages.

### Interpretation

The findings, either of the inventory analysis or the impact assessment, or both, are evaluated in relation to the defined goal and scope in order to reach conclusions and recommendations. This includes also sensitivity analysis (aiming to identify how sensitive the results are to changes in parameters) and uncertainty analysis with Monte Carlo analysis (an approach examining how the results change when they are simulated with random parameter constellations by setting a minimum and maximum standard deviation) (Ikhlayel, 2017).

Currently, LCA is used in many applications in a variety of sectors. It can be used to support decision making at different decision-contexts (EC-JRC, 2010a). Several elements may affect the LCA results and, consequently, the decision support, including: data quality, modeling and methodological choices, uncertainty analysis etc. (Sala et al., 2016).

## 5.3 LCA of WEEE

In the last years, several LCA studies in the context of (W)EEE have been published (Clarke et al., 2019). According to Rodriguez-Garcia and Weil (2016), even though the first publications appeared in the late 1990s, the field was quite immature until 2005 and there were few peer-reviewed papers published. All studies identified in the literature have an attributional approach. As described by UNEP/SETAC (2011), this means that all inputs and outputs are attributed to the functional unit of a product system by linking and/or partitioning the unit processes of the system according to a normative rule.

Some studies evaluated the whole life cycle of EEE: raw material extraction, manufacturing, transport, use and end-of-life (e.g. Achachlouei and Moberg, 2015; Choi et al., 2006). Others are focused on WEEE treatment (e.g. Biganzoli et al., 2015; Hong et al., 2015). There are also works that present a literature review of LCA studies for (W)EEE (e.g. Andrae and Andersen, 2010; Ismail and Hanafiah, 2019; Rodriguez-Garcia and Weil, 2016). Arushanyan et al. (2014) published a review specific of LCAs for information and communication technology (ICT) products and services.

The comparison of different scenarios has been widely applied in LCA studies for e-waste treatment. It allows the best environmental performance to be identified, based on quantitative indicators. The scenarios include different treatment scenarios like recycling (comparing different technologies), reuse (including scenarios with diverse product durability), incineration and landfilling. For example, Ardente and Mathieux, (2014) compared different durability scenarios for washing machines, Tasaki et al. (2013) compared treatment scenarios for TVs, air conditioners and refrigerators and Pérez-Belis et al. (2017) for vacuum cleaners. In some cases, the scenarios intend to capture the best practices, however, the reality is that the end-of-life may not proceed exactly as intended in best practice (Baxter et al., 2016).

The quotes for environmentally weighted recyclability (QWERTY) concept, developed by Huisman (2003), focuses on the determination of environmentally-weighted recycling scores rather than weight-based recycling scores. The materials' environmental values include the secondary material itself with the EOL treatment and is based on LCA results. The QWERTY concept was applied to several WEEE case studies such as sound machines, mobile phones and DVD players.

Some studies have combined LCA with other methods like MFA, Multi-Criteria Decision Analysis (MCDM), Life Cycle Cost (LCC) and Risk Assessment (RA). For example, combined MFA and LCA or carbon footprint analysis have been performed by different authors (e.g. Biganzoli et al., 2015; Clarke et al., 2019; Hischier et al., 2005; Turner et al., 2016; Wäger et al., 2011). Souza et al. (2016) combined LCA and MCDM to assess sustainability and prioritize system alternatives for potential implementation of e-waste management in Rio de Janeiro, Brazil. Dowdell et al. (2000) presented an integrated LCA and LCC for assessing the impacts of treating eight products in the United Kingdom (UK). Wang (2014) introduced some elements of risk assessment to LCA methodology to evaluate the human toxicity impacts on a global scale and on the occupational environment with different scenarios varying parameters such as space and ventilation.

The following paragraphs present a summary of the methodological approaches of some of the studies identified in the literature. Section 5.3.1 presents a closer look at the LCA studies focusing on screen devices.

### Goal and Scope

Considering that WEEE is very heterogeneous, *most of the studies focus on one or more types of WEEE, or on one of its components or materials*. For example, Menikpura et al. (2014) assessed the benefits of e-waste recycling of washing machines, refrigerators, air conditioners, and televisions. Bigum et al. (2012) modeled the recovery of some metals, like aluminum, copper, gold, and palladium, from high-grade e-waste. Wäger and Hischier (2015) performed an in-depth life cycle assessment of the recycling of mixed plastics-rich residues from WEEE.

Because the studies published in the literature assess different types of (W)EEE products, components, materials, and/or processes, *almost every study has a different functional unit*. For example, Baxter et al. (2016) assessed and compared the potential life cycle impacts of treating three types of WEEE. The functional unit defined is "treatment of one typically sized device of refrigerators (51 kg), LCD screens (20 kg) or mobile telephones (140 g)". Song et al. (2013b) assessed the impacts of an e-waste treatment enterprise in China. The functional unit of this study is: "recycling 1 t of e-waste in the company".

Regarding the system boundaries, among those studies dealing specifically with WEEE, *we identified three groups of studies: waste management, recycling, and reuse*. The first type of study includes boundaries from the collection of the e-waste to the different types of treatment according to specified treatment scenarios (e.g. Noon et al., 2011a; Wäger et al., 2011). The second type focuses on the impact of the different steps to recycle the e-waste (e.g. Ikhlayel, 2017; Soo and Doolan, 2014; Wäger and Hischier, 2015) or one component like printed circuit board scrap (e.g. Rubin et al., 2014). The third regroups a few studies that assessed the impacts of reuse (e.g. Lu et al., 2014; Zink et al., 2014).

### Life Cycle Inventory

Many studies used *a combination of primary and secondary data*. For example, Duan et al. (2009) used Ecoinvent databases, combined with literature and field investigations. Studies in which the authors present the data inventory in detail are rare, see Achachlouei et al. (2015), Biganzoli et al. (2015), Clarke et al. (2019) and Turner et al. (2016).

### Life Cycle Impact Assessment (LCIA)

According to Rodriguez-Garcia and Weil (2016), the most commonly used methodologies in the studies they identified from the 1990s to 2015 are CML 2001, Eco-Indicator ('95 or '99), and Impact 2002+. *We noticed that in the most recent studies, ReCiPe and ILCD methods are also frequently used* (e.g. Amato et al., 2017; Bigum et al., 2017; Hischier and Wäger, 2015; Wäger and Hischier, 2015).

Concerning the impact categories, *global warming is the most widely assessed*. Other categories related to toxicity (human and ecotoxicity), acidification, eutrophication, resource depletion, and energy (cumulative energy demand or emergy) are also addressed in some of the studies identified in the literature. *In addition, most of the studies did not include normalization and weighting.*

**Interpretation**

In the studies that assessed the whole life cycle of the EEE, the end of life had a lower impact in comparison to other phases (Rodriguez-Garcia and Weil, 2016). Some authors highlighted the potential of WEEE recycling to lead to an environmental benefit if the avoided impacts are considered in the assessment (Duan et al., 2009).

### 5.3.1. Focus on LCA studies of screens

In the literature review for ICT products performed by Arushanyan et al. (2014), they identified that the products assessed most in single LCA studies were computers and laptops, monitors, TVs, and mobile phones. In the literature review of LCA studies performed by Ismail and Hanafiah (2019), among the 47 studies they identified that analyzed one or more types of WEEE devices, 13 studies assessed one or more equipment type from the WEEE category II.

The earliest LCA study to include monitor disposal was completed for the European Commission and published in 1998 (Amato et al., 2017). In the 2000s, some LCA studies of screen devices were published. Seeing the importance of WEEE composition for the LCA modeling, some of the studies published the bill of materials considered in the assessment (e.g. Achachlouei and Moberg, 2015; Eugster et al., 2007).

Socolof et al. (2005) studied the environmental impacts of CRT and LCD monitors. The end-of-life is included in the system boundary of the study (recycling, landfilling, and incineration). However, neither the EOL scenarios nor their results are discussed in detail. The authors only stressed that the EOL had a low contribution in comparison to manufacturing and use phases.

Lu et al. (2006) compared the environmental impacts of various scenarios of notebook treatment in Taiwan, including different ratios of recycling, reuse, incineration and landfilling. The authors identified that the control unit is the main contributor to environmental impact in the recycling process.

In the Netherlands, Barba-Gutierrez et al. (2008) assessed the environmental impacts of TV sets, among other devices. The studied is focused in the assessment of transport distances and types of vehicles. Two scenarios are considered: landfill versus recycling. The LCIA method used was Eco-Indicator 99.

Dodbiba et al. (2008) published an LCA study comparing two treatment options of plastic wastes from discarded TV sets (energy recovery and mechanical recycling), and different resins were also assessed. The authors concluded that recycling entails fewer impacts and suggested reducing the number of plastic types being used in the manufacturing process of TVs (preferably excluding PVC due to its impact in production).

Duan et al. (2009) assessed the complete life cycle of the PC system: desktop computer including the screen (CRT and LCD technologies), the keyboard and the mouse. The Ecoinvent database is used as a source of LCI data. Concerning the EOL, three scenarios are considered: worst-case (landfilling), recycling worst-case and recycling best-case (including avoided primary production). The environmental benefits from the 'recycling best case" scenario are almost as high as the environmental impact in the worst case.

Feng and Ma (2009) assessed the environmental impacts of a color TV set in China. It included the production of manufacturing materials, their transport, color TV set manufacturing, transport, use, discarding and partial plastic waste energy utilization. Except for the category "bulk waste", the EOL had the lowest impact for all impact categories considered in the study (EDIP LCIA method). Nevertheless, only the seven major materials used to manufacture the color TV set are considered in the discard phase. The plastic is considered to be 100% recovered and incinerated.

Noon et al. (2011) compared the impacts of LCD and CRT monitor treatment in two parts: first, a comparison of the impacts of disposing of the monitors in landfills in 2007; and second, the change in total monitor disposal between 2008 and 2010, including monitor recycling. Besides the global warming potential, the authors analyzed some inventory data (e.g. total energy consumption, total fossil fuel consumption, mercury and lead content). For the first part, the disposal of LCD had a lower impact than CRT, except for the mercury content. Regarding the second part of the study, the results are separated into two scenarios: one with credit for the avoidance of primary material produced - in response to the generation of recyclables - and another one where the credit is not considered.

Some studies focused on the impacts on e-waste recycling of recovering certain elements. For instance, Dodbiba et al. (2012) used LCA to compare two different types of liberation methods to recover indium from LCD: (1) grinding and incineration of LCD modules, followed by leaching; (2) electrical disintegration of LCD modules, followed by leaching. In turn, Compagno et al. (2014) compared the impacts of current CRT recycling practice, to a recycling strategy to recover the metallic lead.

Niu et al. (2012) assessed the environmental impacts of scrap CRT display treatment in three scenarios (incineration, manual dismantling and mechanical dismantling). LCI is based on literature data and a commercial database (Ecoinvent). The authors concluded that CRT incineration is the main contributory factor in the LCA of CRT treatment. Rocchetti and Beolchini (2014) also assessed scenarios for CRT treatment, but they defined different scenarios: landfilling and three recycling scenarios. Moreover, they included the avoided impacts in the LCA results.

Song et al. (2013a) employed LCA to investigate environmental performances of personal computers (PCs) in Macau, including desktop units, CRT screens and LCD screens. The system boundaries included the collection of e-waste, manual dismantling, recycling of some materials, incineration, and disposal. Similar to other studies, the authors used primary data and the Ecoinvent database and included the avoided impact in the assessment. The highest benefits of recycling are for human health, followed by resources and ecosystem quality (Ecoindicator 99 LCIA method).

Achachlouei et al. (2015) assessed and compared the life cycle impact of reading magazines on paper and tablets. The EOL treatment of the tablet device is modeled based on estimates of waste treatment obtained from a recycling company, and generic data for some processes (Ecoinvent database). The EOL phase had a low contribution to the overall impact. According to the authors, the limits of data used for EOL modeling contributed to the study's uncertainties.

Hischier and Wäger (2015) evaluated the impacts of consumer behavior with a shift from desktop computers to mobile devices such as laptops and tablets. The functional unit "1 hour of active use" is considered for the comparison. The authors used data from the Ecoinvent database, and the ReCiPe method for the impact assessment. The only impact category with significant impacts for the EOL phase is freshwater ecotoxicity potential, with a share of 20% of the total impact for all devices.

In Belgium, Van Eygen et al. (2016) studied the environmental performance of laptop and desktop computer recycling. The analysis was performed in different steps, including LCA. The focus of the LCA lies on raw material consumption, and the impact on the natural resources is quantified by the Cumulative Exergy Extraction from the Natural Environment (CEENE). The end-processing step has by far the biggest CEENE impact, compared to the impact of the collection and primary treatment steps (current scope of the compliance schemes).

Amato et al. (2017) compared four scenarios of LCD management: disposal in landfilling sites (scenario 1), incineration (scenario 2), traditional recycling treatment (scenario 3) and an innovative process to recover of indium (scenario 4). The authors presented an average composition of LCD and mentioned the use of primary and secondary data in the LCI. The environmental assessment is focused on the categories of global warming and acidification (based on ILCD method). Scenario 1 and 2 presented the highest impacts for climate change and acidification. Based on the LCA results, the scenario with lowest impact is the monitor recycling without indium recovery. Yet, the result is influenced by limitations in LCIA methods and the absence of impacts connected with the reagents used in the panel treatment.

Ikhlayel (2017) evaluated the environmental impacts of treating laptops, mobile phones, CRT TVs, LCD TVs, washing machines, and refrigerators. Five EOL scenarios are considered and three e-waste management processes are considered: sanitary landfills, recycling of metals and materials, and the incineration of plastic and PCBs residues after recycling of precious metals (PMs). The scenario with highest recycling performance (the materials considered for recycling are copper, iron, aluminum, palladium, gold, and silver) is the one with lower environmental impacts. The authors noticed that the impacts are sensitive to variations in the recycling rate, followed by the amounts of PMs, but this does not influence the overall evaluation.

In China, Song et al. (2018) estimated the energy use and greenhouse gas (GHG) emissions arising from CRT TV treatment. Besides the current treatment scenario, the authors included two other scenarios with higher recycling rates based on the China's WEEE Directive encouragement system. The authors concluded that the overall energy use and GHG emissions of the CRT TV treatment systems are dependent on two key processes: collection with transport, and funnel glass recycling.

Dewulf et al. (2019) assessed and compared the potential environmental impacts of five treatment scenarios for plastics from housings of 100 kg of LCDs. Four scenarios included recycling of some polymers (HIPS and ABS), and one considering 100% incineration with energy recovery. The potential environmental benefits were included in the assessment, and the Ecoinvent database was used for LCI data. The maximum recovery scenario resulted in half the impact compared to the incineration scenario. If a significantly amount of material can be recovered, the potential environmental benefit of post-consumption recycling is clear.

### 5.3.2. Main difficulties and limitations

Although LCA is the most widely used methodology to analyze the potential environmental impacts of products, the method has its limitations. As some of the disadvantages of this method Huisman (2003) has listed the subjectivity of methodological choices, the time consumed performing the LCA study, and the amount and level of data required.

As highlighted by Andrae and Andersen (2010), there is a lack of transparency in LCA studies which makes understanding the results difficult. *The lack of transparency ranges from the methodological choices, to the source of data.* Rodriguez-Garcia and Weil (2016) identified a lack of definition of the impact assessment methodology in the LCA studies, as well as in those classified as carbon footprint studies. As stressed by Arushanyan et al. (2014), the types and sources of data for the LCI are not always specified, which produces difficulties in judging the reliability of the results and the conclusions. Moreover, *it is not possible to fully compare the LCA results from one study to another due to different methodological choices*.

## 5.4 LCI: data quality and databases for e-waste treatment

Depending on the scope of the LCA, the amount of data to be gathered in the LCI phase may be significant. The reliability and reproducibility of data are essential and must be transparent in order to ensure credibility and enable comprehension of the results. Lack of data, including data gaps and lack of representative data, as well as data inaccuracy, contribute to result uncertainties. Uncertainty means, basically, lack of certainty (Ciroth et al., 2013). It is the quantitative degree of the lack of confidence that a parameter value is truly representative.

When there are limits of time and/or budget for data gathering, it is recommended searching for higher quality data for the foreground system (processes which are under the control of the decision-maker for which an LCA is carried out) and adopting secondary data for background processes.

Aiming to support LCA studies with inventory data, different databases have been developed in recent years. Since databases are an important data source for LCA, how they describe uncertainty in their data is the key to subsequently evaluating the overall uncertainty of the study (Muller et al., 2014). Some companies have their own database – for example, Huisman et al. (2003) used the Philips internal LCA database in QWERTY assessments. Recently in France, ESR (one of WEEE compliances schemes) released a database for LCI of e-waste treatment (more details are presented in section 5.4.2).

The Ecoinvent database is one of the main LCI databases with over 14,000 datasets in many areas such as energy supply, agriculture, transport, and waste treatment. A significant part of the (W)EEE studies presented in the previous section used this database for at least part of LCI data. Among the advantages of using the Ecoinvent database is the access to reports and detailed data about the modeling assumptions (available in ecoQuery), and also that the uncertainties of each input and output of the datasets is disclosed. In section 5.4.1, the uncertainty approach used in the Ecoinvent database, and the e-waste treatment datasets are presented in more detail.

Lloyd and Ries (2007) categorized the uncertainty in LCA studies into three types: parameters, scenario, and model. Parameter uncertainty involves the precision of the inventory values and parameters, as well as technological characteristics. Scenario uncertainty relates to methodological choices regarding the functional unit, time horizons, geographical scales, allocation procedures, and treatment scenarios. Model uncertainty includes LCIA approaches (mathematical models and characterizations factors).

Parameter and model are stochastic uncertainties and, therefore, can be represented by statistical parameters, such as the variance and the standard deviation. In turn, the scenario uncertainty comprises a limited number of possible choices and can be evaluated separately through sensitivity analysis (Belizario Silva et al., 2017; EC-JRC, 2010a).

### 5.4.1. Ecoinvent database

The Ecoinvent database quantifies uncertainty in a semi-quantitative approach. For each dataset, two kinds of uncertainty are considered in the final uncertainty calculation for all inputs and outputs (Ecoinvent, n.d.; Muller et al., 2014; Weidema et al., 2013):

- Basic uncertainty: is the intrinsic variability and stochastic error of the parameters. This value reflects the fact that even a "perfect" data is uncertain and there is fluctuation over time, errors in measurements, etc. The basic uncertainty can be either measured or estimated from a table provided by Ecoinvent with basic uncertainty factors. It is assumed that different types of exchanges differ in their uncertainty. Thus, the basic uncertainty factors are differentiated by exchange type (resources, pollutants emissions to different compartments, etc.) and by class of process (combustion, process or agricultural).

- Additional uncertainty: uncertainty due to the use of imperfect data (e.g. data resulting from estimates, lacking verification, spatially and/or technologically different conditions, etc.). The procedure is based on a Pedigree Matrix, composed of five data quality indicators: reliability (sampling methods and verification procedures); completeness (statistical representativeness of the data and time periods for data collection); temporal, geographic and further technological correlation (for data used outside its proper context). The individual indicators are scored from 1 to 5, where 5 is low quality.

The LCI data for EOL treatment of e-waste is based on a report released by Hischier et al. (2007). The inventory data for the various datasets are based on the authors' experience and supplemented by information from various literature sources. The datasets were developed for Ecoinvent version 2 and were not individually updated for version 3.

The e-waste treatment datasets consider two treatment levels: the first level includes the dismantling (manual or mechanical) of the devices, and the second level includes the refining or disposal of the fractions and components of the first stage. As presented in **Chapter 4**, mechanical dismantling (shredding) is followed by sorting techniques (e.g. magnet, eddy current). In order to calculate the number of fractions (e.g. ferrous, copper and aluminum) sent to the second level of treatment, they used data on the composition of the devices, and a transfer coefficient that indicates the share of metals in each fraction (see Table 5-1). All burdens of this step are allocated to the residues of shredding and sorting.

Figure 5-2 exemplifies how the EOL of e-waste devices is modeled within the database with the dataset "treatment of 1 kg of liquid crystal display". For the mechanical treatment scenario, the authors assumed that part of the LCD was shredded, and that part of the fractions was sent to further treatment for metals recovery (the three outputs identified in blue). The outputs highlighted in green are fractions and components for which a disposal scenario was considered. The components that followed further treatment before material recovery and/or disposal are identified in orange. Regarding the printed wiring board (PWB) treatment, it includes weighing, shredding and sampling of PWB. The recovery of the secondary raw materials is not included in the system boundaries. The inputs and outputs related to the WEEE facility (e.g. land occupation, energy consumption) are grouped in the dataset highlighted in grey.

The modular approach allows the user to change the fractions/components treatment from those considered in the scenario defined in the database. Moreover, even if the database is limited to some devices, the modular datasets together with the transfer coefficient matrix (Table 5-1) enable the user establishing various scenarios for the EOL treatment of different electric and electronic device.

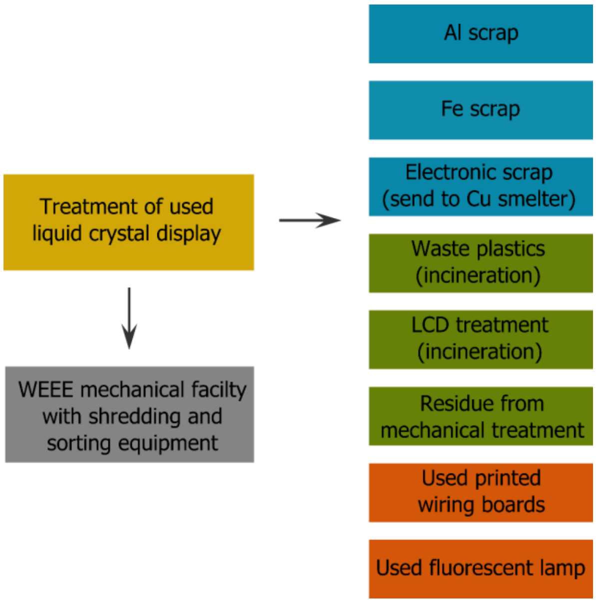



Figure 5-2. Ecoinvent modulation for treatment of used LCD displays.

Table 5-1. Transfer coefficient matrix considered by Ecoinvent database.

| Fractions (f*) | Fe | Al | Cu | Residue |
|---|---|---|---|---|
| Aluminum | 0.50% | **82.58%** | 4.92% | 12.00% |
| Copper | 0.94% | 5.00% | **78.21%** | 15.85% |
| Ferrous | **95.0%** | 1.00% | 1.00% | 3.00% |
| Plastics | 1.21% | 0.50% | 10.00% | 88.89% |
| Ag | 0.99% | 0.99% | 84.92% | 13.10% |
| Au | 0.99% | 0.99% | 80.00% | 18.02% |
| Pb | 1.18% | 1.18% | 80.0% | 17.65% |
| Other fractions | 0.69% | 0.67% | 35.29% | 63.35% |

Source: Hischier et al. (2007) based on data from Huisman (2003).

### 5.4.2. ESR database

In 2017, ESR published an LCI database for household WEEE[16]. The study was co-funded by ADEME and developed by Bleu Safran. The LCIs were initially created to satisfy the needs of eco-design (ESR and Bleu Safran, 2018) so that EEE producers could have more consistent data for the EOL that is usually a phase for which they do not have primary data. The LCI database includes all WEEE categories. However, regarding the screens category (current category II), only flat panel displays are addressed since CRT screens are no longer produced, and the users targeted by the database are EEE producers.

[16] Available in: http://weee-lci.eco-systemes.com and http://weee-lci.recylum.com

To achieve this modeling, different sources were used, including WEEE sampling and composition analysis, and data gathering of ESR operators in the different activities related to e-waste collection and treatment. It was a pioneer project in the field of EOL treatment, and since ESR accounts for around 75% market share of compliance schemes (Fangeat, 2019), the results are representative of e-waste treatment technology in France (Osset et al., 2018).

The datasets have been developed at the scale of a coupled material-WEEE category, resulting in inventories of "end-of-life of 1kg of material in the WEEE category". The system boundaries of the inventories include all management phases from the collection of WEEE at collection points to all final destinations of the processed fractions (recycling, energy recovery, landfilling), including transport. Each process is available in two modeling approaches: including the benefits of recycling and energy recovery (system expansion) and not including them (cut-off).

The datasets are available in system processes ILCD format (input and output data are aggregated). Table 5-2 presents examples of the datasets available for flat screens in the ESR database. It is important to highlight that the output of the LCI is 1 kg of material treated by the official schemes that may include recycling and other types of treatment, and not 1 kg of material recycled (e.g. LCI of the end-of-life of 1kg of steel in large household appliances, LCI of the end-of-life of 1kg of glass in lamps, etc.).

Table 5-2. Flat screens: examples of datasets available on ESR database (1 kg of material in WEEE).

| **Materials** | **Unit process** |
|---|---|
| ABS-PC | ABS-PC with BFR, Substitution benefits included or not included<br>ABS-PC without BFR, density <1.3, Substitution benefits not included<br>ABS-PC without BFR, density >1.3, Substitution benefits not included |
| Copper | Copper within PCB, Substitution benefits included or not included<br>Copper within wire, Substitution benefits included or not included |
| Gold | Gold within PCB, Substitution benefits included or not included |

The recovery or recycling paths considered by the ESR study for the materials present in FPD screens are presented in Table 5-2. The table provides details of the type of recycling or recovery per material. The information disclosed allows the identification of which materials are considered for recycling in the database scenario, but not the details regarding the recycling efficiency, and the share between recycling and other treatments (e.g. the percentage that was recycled in 1kg of end-of-life gold).

As presented in Table 5-3, part of the copper is considered to be sent to the copper refinery (production of secondary raw materials), but part is mixed with Al or Fe fractions and sent to aluminum refineries and steelworks, as well as a percentage sent to a non-hazardous waste storage facility (NHWSF). No information on uncertainties is available in the datasets.

Table 5-3. Flat screen modeling in ESR database: details on recovery and substitution point.

| Materials/ Components | Final destination | Details |
|---|---|---|
| Steel | Steelworks | Secondary steel at steelworks output |
| | MWIP | Steel recovered from clinkers and then secondary steel at steelworks output |
| Aluminum | Aluminum refinery | Secondary aluminum at aluminum refinery output |
| | Copper refinery | Dross recovered in construction sector at copper refinery output |
| | NHWSF | - |
| | MWIP | Aluminum recovered from clinkers and then secondary aluminum at steelworks output |
| Copper | Copper refinery | Secondary copper at copper refinery output |
| | Aluminum refinery | Substitution of alloying elements in aluminum refiner |
| | Steelworks | Inclusion in secondary steel at steelworks output |
| | NHWSF | - |
| Copper within wire | Copper smelter | Secondary copper at wire treatment operator output |
| | Copper refinery | Secondary copper at copper refinery output |
| Brass (Cu/Zn/Pb/Sn) | Copper refinery | Secondary Cu/Pb/Sn at copper refinery output |
| | Steelworks | Inclusion in secondary steel at steelworks output |
| | NHWSF | - |
| | MWIP | - |
| Zinc | Copper refinery | Oxide Zn dust toward construction sector at copper refinery output |
| | Steelworks | Inclusion in secondary steel at steelworks output |
| | Aluminum refinery | Substitution of alloying elements in aluminum refinery |
| | Recovery in construction sector | Included in materials recoverable in construction sector at crushing output |
| | NHWSF | - |
| | MWIP | - |
| Precious metals and standard metals in electronic boards | Copper/precious metals refinery | Secondary copper at copper refinery output<br>Secondary gold, silver or platinoids at refinery output (after refining of electrolytic sludge) Secondary lead at refinery output (after refining of tin/lead slags)<br>Standard metals in dross recoverable in construction sector at refinery output |
| PCB support | Copper/precious metals refinery | Epoxy resins (25%): role of reducing agent (same as coke) in refinery<br>Epoxy resins (25%): energy intake in copper refinery<br>Epoxy resins (50%): combustion without energy recovery<br>Fiberglass in dross oriented toward construction sector at refinery output |
| | MWIP | Epoxy resins: combustion with energy recovery at incinerator output<br>Fiberglass: no material or energy recovery |
| Rigid plastic without BFR, density <1.3 | High quality recycling | Secondary ABS or PS at regenerator output Secondary PC or ABS/PC at regenerator output Secondary PMMA at regenerator output |
| | Low quality or mixed plastic recycling | Secondary ABS or PS at regenerator output Secondary PC or ABS/PC at regenerator output Secondary PMMA at regenerator output |

| | | |
|---|---|---|
| | Copper refinery | Role of reducing agent (same as coke) in refinery<br>Combustion with and without energy recovery |
| | MWIP | Combustion with energy recovery at incinerator output |
| | HWIP | Combustion without energy recovery (proxy) |
| Rigid plastic without BFR, density >1.3 | High quality recycling | Secondary ABS or PS at regenerator output Secondary PC or ABS/PC at regenerator output<br>All plastics to regenerator rejection (then NHWSF) |
| | Low quality or mixed plastic recycling | Secondary ABS or PS in mixed plastics at regenerator output<br>Secondary PC in mixed plastics at regenerator output<br>Secondary ABS/PC in mixed plastics at regenerator output<br>Secondary PET in mixed plastics at regenerator output |
| | Copper refinery | Role of reducing agent (same as coke) in refinery<br>Combustion with and without energy recovery |
| Rigid plastic with BFR | HWIP | Combustion without energy recovery (proxy) |
| | NHWSF | - |
| | High quality recycling | All plastics to regenerator rejection (then NHWSF) |
| | Copper refinery | Role of reducing agent (same as coke) in refinery<br>Combustion with and without energy recovery |
| Plastics within wire (PE, PVC) | Copper refinery | Role of reducing agent (same as coke) in refinery<br>Combustion with and without energy recovery |
| Glass | HWIP | Refractory function at the arrival in incinerator furnace |
| | NHWSF | Combustion with energy recovery at incinerator output |
| | MWIP | Combustion with energy intake in cement works |

MWIP: Municipal waste incineration plant / NHWSF: Non-hazardous waste storage facility / HWIP: Hazardous waste incineration plant

Source: Adapted from ESR and Bleu Safran (2018)

So far, ESR has decided not to make public the individual reports with details on the data used and further assumptions of the modeling. As stressed by the critical review panel report addressed to ESR (Osset et al., 2018), not disclosing this information limits the transparency of the overall study. Public access to these reports would be highly welcome to enable better understanding and would stimulate the use of the database.

Depending on the goal of the LCA study, it is not always possible to use the ESR database. The system process format does not allow the inventory to be adapted – for example, to increase energy consumption due to a different sorting process. Moreover, the lack of modeling assumptions does not permit a full understanding of the treatment scenario. For example, if the goal of the study is to compare different treatment paths, the database cannot be used.

## 5.5 Case study: LCA of screens recycling in France

### 5.5.1. Goal and scope

The goal of this LCA is to assess the impacts and benefits of e-waste recycling by the official schemes in France in a given year. Similar to the other case studies presented in this thesis, we focused on the screens category.

The functional unit of the study is treatment of category II devices in a reference year by the compliance schemes in France. The total quantity treated varies depending on the e-waste collected by the official schemes in the year, as well as on the processing capacity of the compliance schemes (see **Chapter 4** for more details).

The system boundaries of the study comprise the WEEE recycling chain: from dismantling and depollution to the recovery of secondary raw materials in e-waste or the disposal of non-recyclable fractions. Reuse is not considered in the assessment because it is still a limited treatment in France (in 2018, it represented only 2% of the e-waste treated by the official schemes). The system boundary does not include collection and regrouping due to lack of data on the inputs and outputs of these steps. Lastly, transport was not included, but a study on the impact of transport in LCA of e-waste treatment is presented in section 5.5.4 with a focus on treatment of tablets.

### 5.5.2. LCI of screens

Due to the absence of CRT data, the ESR database cannot be used to assess the impacts of screens currently treated by the official schemes. As described in **Chapter 4**, CRT devices represent 85% of the screens collected and treated nowadays. Furthermore, the lack of transparency on the treatment scenario, and the impossibility of adapting the datasets limit the use of ESR database.

The foreground processes are based on literature data on the composition of EEE, as presented in Table 5-4 to Table 5-7, and treatment scenarios. The Ecoinvent v3 database is used in this study for the background processes of the treatment paths. Part of the LCI data is based on information presented in **Chapter 4.** The device composition is relevant to quantify the amount of materials and components sent to the different treatment paths described in Table 4-9. Nevertheless, neither the components or materials production nor their use is included in the study boundaries. The uncertainty of the input and output data of the foreground processes is assessed based on the basic uncertainty and the Pedigree matrix defined by Ecoinvent. The combination of different treatment paths per device described in Table 4-9 is defined in this chapter as the baseline scenario. The baseline scenario considers the shredding and sorting efficiency described in Table 4-10 (**Chapter 4**). Aiming to assess the impact on the LCA results of varying the treatment path share, as well as of shredding and sorting performance, a sensitivity analysis based on the variation of these two parameters is included in the LCIA.

Although plastics recycling is not yet well established in France, a treatment path including it is considered for all the devices. For the tablets and laptops, the path including plastics recycling is similar to path B described for each device in Table 4-9. For the CRT and FPD TVs and monitors, they are similar to path A. Among the different types of polymers present in screen components, based on discussions with experts, it was considered that only those in polycarbonate (PC), polystyrene (PS) and acrylonitrile butadiene styrene (ABS) - and without flame retardants - are potentially recycled nowadays.

Table 5-4. Tablets: bill of materials and materials/components treatment.

| Materials/Components | Average composition (%) | Treatment after dismantling and/or shredding and sorting |
|---|---|---|
| Aluminum alloys | 7.8% | Recycling |
| Steel | 0.7% | Recycling |
| Magnesium alloys | 2.8% | Recycling mixed with aluminum alloys |
| Plastics (unspecified) | 0.8% | Incineration |
| Acrylonitrile butadiene styrene (ABS) | 0.2% | Paths A, B and C: incineration<br>Path D: recycling |
| Polycarbonate (PC) | 2.5% | Paths A, B and C: incineration<br>Path D: recycling |
| PC with glass fiber | 1.7% | Paths A, B and C: incineration<br>Path D: recycling |
| ABS + PC | 4.7% | Paths A, B and C: incineration<br>Path D: recycling |
| Display panel | 42.9% | Incineration |
| PCB | 8.3% | Recycling |
| Speaker | 0.6% | Recycling (basic metals) |
| Battery | 23.6% | Path A: incineration<br>Paths B, C and D: recycling |
| Other components | 3.4% | Incineration with remaining treatment losses |

Source for average compositions: Tecchio et al., 2018.

Table 5-5. Laptops: bill of materials and materials/components treatment

| Materials/Components | Average composition (%) | Treatment after dismantling and/or shredding and sorting |
|---|---|---|
| Aluminum alloys | 9.8% | Recycling |
| Magnesium alloys | 9.2% | Recycling |
| Steel | 4.0% | Recycling |
| Plastics with flame retardant | 12.3% | Incineration |
| Poly-methyl methacrylate (PMMA) | 5.4% | Incineration |
| Unspecified plastics | 5.3% | Paths A, B and C: incineration<br>Path D: ABS recycling (considered as 3% of unspecified plastics) |
| Display panel | 8.5% | Incineration |
| Battery | 13.6% | Recycling |
| PCB | 13.7% | Recycling |
| Drives | 16.0% | Recycling |
| Fan | 0.5% | Recycling (basic metals) |
| Speakers | 0.3% | Recycling (basic metals) |
| Lamps | 0.4% | Treatment and disposal |
| Cables | 0.9% | Recycling (copper) |

Source for average compositions: Tecchio et al., 2018.

Table 5-6. FPD monitors and TVs: bill of materials and materials/components treatment

| Materials/Components | Average composition (%) | Treatment after dismantling and/or shredding and sorting |
|---|---|---|
| Aluminum alloys | 5.3% | Recycling |
| Steel | 26.8% | Recycling |
| ABS | 21.0% | Paths A and B: incineration<br>Path C: recycling |
| PC | 0.2% | Paths A and B: incineration<br>Paths C: recycling |
| Unspecified plastics | 25.1% | Incineration |
| PCB | 9.3% | Recycling |
| PCB smaller than 10cm$^2$ | 0.1% | Recycling |
| LCD screens | 6.6% | Incineration |
| Lamps | 0.1% | Treatment and disposal |
| Fan | 0.3% | Recycling (basic metals) |
| Speakers | 2.7% | Recycling (basic metals) |
| Cables | 2.0% | Recycling (copper) |
| Capacitors | 0.1% | Treatment and disposal |
| Screws | 0.4% | Recycling (basic metals) |

Source for average compositions: Talens Peiró et al., 2016.

Table 5-7. CRT monitors and TVs: bill of materials and materials/components treatment

| Materials/Components | Average composition (%) | Treatment after dismantling and/or shredding and sorting |
|---|---|---|
| Steel | 18.8% | Recycling |
| Aluminum | 2.0% | Recycling |
| Copper | 0.3% | Recycling |
| Other metals | 0.9% | Recycling (basic metals) |
| Plastics | 16.4% | Path A: Incineration<br>Path B: Recycling |
| Cable | 2.3% | Recycling (copper) |
| PWB | 4.7% | Recycling |

Source for average compositions: Hischier et al., 2007.

Regarding the recovery of secondary raw materials and disposal of non-recyclable fractions (incineration or landfill), the modeling is limited by the datasets available in the Ecoinvent database. For example, for PCB recycling, Ecoinvent datasets are modeled considering the recycling of gold, silver, nickel, palladium, and copper. Thus, it was not possible to include the recycling of other metals (e.g. antimony). In the literature review, some studies included recycling of other components, such as display panels. In these studies, the authors used primary data to carry out the LCI. In France, display panels are currently not recycled on an industrial scale, but are disposed. The only treatment available in the Ecoinvent database is incineration.

Seeing that monitors and televisions have similar compositions and undergo similar treatment, in the absence of a specific bill of materials for these products we considered an average composition for CRT and FPD monitors and devices.

### 5.5.3. Impact assessment

The environmental assessment is focused on the categories of global warming (IPCC) and human toxicity (USEtox). The choice of impact categories is because climate change is a very topical issue, and toxicity is one of the drivers for waste management. Both categories were widely included in the LCA studies for WEEE identified in the literature (see section 5.2).

Firstly, we present the impacts of treating the different types of screens according to their treatment paths. Then, we assess the variation of the treatment paths in different waste treatment scenarios to identify the influence of the parameters on the results. The treatment scenario uncertainty is assessed by running a Monte Carlo analysis with 1000 simulations of random parameters.

#### 5.5.3.1. Tablets

Table 5-8 presents the comparison of treating 1t of tablets with the three treatment paths described in **Chapter 4**, and a fourth path that includes plastic recycling. Considering an average weight of 528 g per tablet, 1 t of tablets corresponds to 1,894 devices. The paths with lower global warming impacts are those with manual dismantling (B and D). The sorting and shredding efficiency have a limited impact on the results (less than 1.7% lower than the current efficiency). The disposal of non-recyclable components and fractions (e.g. display panels and residues of shredding and sorting) account for the main impacts – even in the scenario with the highest recycling (around 50%).

Regarding the toxicity, path A entails lower impacts. However, we are convinced that the impacts of path A are underestimated, because there is no dataset in Ecoinvent for battery disposal (landfill or incineration). Thus, we considered the quantity of batteries not recycled as a generic residue of shredding and sorting. When comparing the other three scenarios where the battery is manual dismantled and recycled, path B results in a lower impact, followed by path D, which includes plastic recycling. The increase in shredding and sorting performance results in an increase of up to 3% in path A.

Table 5-8. Treatment of 1 t of tablets: impact assessment per treatment path.

| Treatment paths | Impact categories | Current performance | >2% | >5% | Coefficient of Variation (CV) |
|---|---|---|---|---|---|
| A - Shredding of the whole device | Global warming (kg $CO_2$-eq) | 1,355.73 | 1,353.86 | 1,344.97 | 12.8% |
| | Human toxicity (CTUh) | 1.19E-08 | 1.20E-08 | 1.22E-08 | 17.4% |
| B - Deep-level manual dismantling | Global warming (kg $CO_2$-eq) | 902.70 | 897.11 | 888.21 | 9.2% |
| | Human toxicity (CTUh) | 1.98E- -08 | 2.00E-08 | 2.02E-08 | 12.6% |
| C - Direct treatment in copper smelter | Global warming (kg $CO_2$-eq) | 959.86 | 954.27 | 945.38 | 9.1% |
| | Human toxicity (CTUh) | 2.18E-0 | 2.20E-0 | 2.19E-08 | 13.2% |
| D - Deep-level manual dismantling with 25% plastics recycling | Global warming (kg $CO_2$-eq) | 838.44 | 832.85 | 823.95 | 8,5% |
| | Human toxicity (CTUh) | 2.00E-08 | 2.02E-08 | 2.02E-08 | 12.5% |

Table 5-9 presents the different scenarios considered in the sensitivity analysis for tablet treatments. Scenario 1 considers an increase in manual dismantling once, in line with the WEEE Directive, the battery and PWB is removed before the treatment. Scenario 2 considers that 5% of screens are treated in facilities with plastic recovery (path D), and a similar share as the baseline scenario among the remaining paths. Lastly, in scenario 3, 25% is sent to treatment path D, and the share among the remaining paths is proportional to scenario 1.

Table 5-9. Tablets: scenarios considered in the LCIA.

| Treatment paths | Scenarios | | | |
|---|---|---|---|---|
| | Baseline | 1 | 2 | 3 |
| A - Shredding of the whole device | 85% | 60% | 81% | 45% |
| B - Deep-level manual dismantling | 5% | 25% | 5% | 19% |
| C - Direct treatment in copper smelter | 10% | 15% | 10% | 11% |
| D - Deep-level manual dismantling with 25% plastics recycling | 0% | 0% | 5% | 25% |

Based on these scenarios, Figure 5-3 presents the results for the global warming potential. As can be seen, the combination of deep dismantling of the devices and plastic recycling reduces the environmental impacts (Figure 5-3.a).

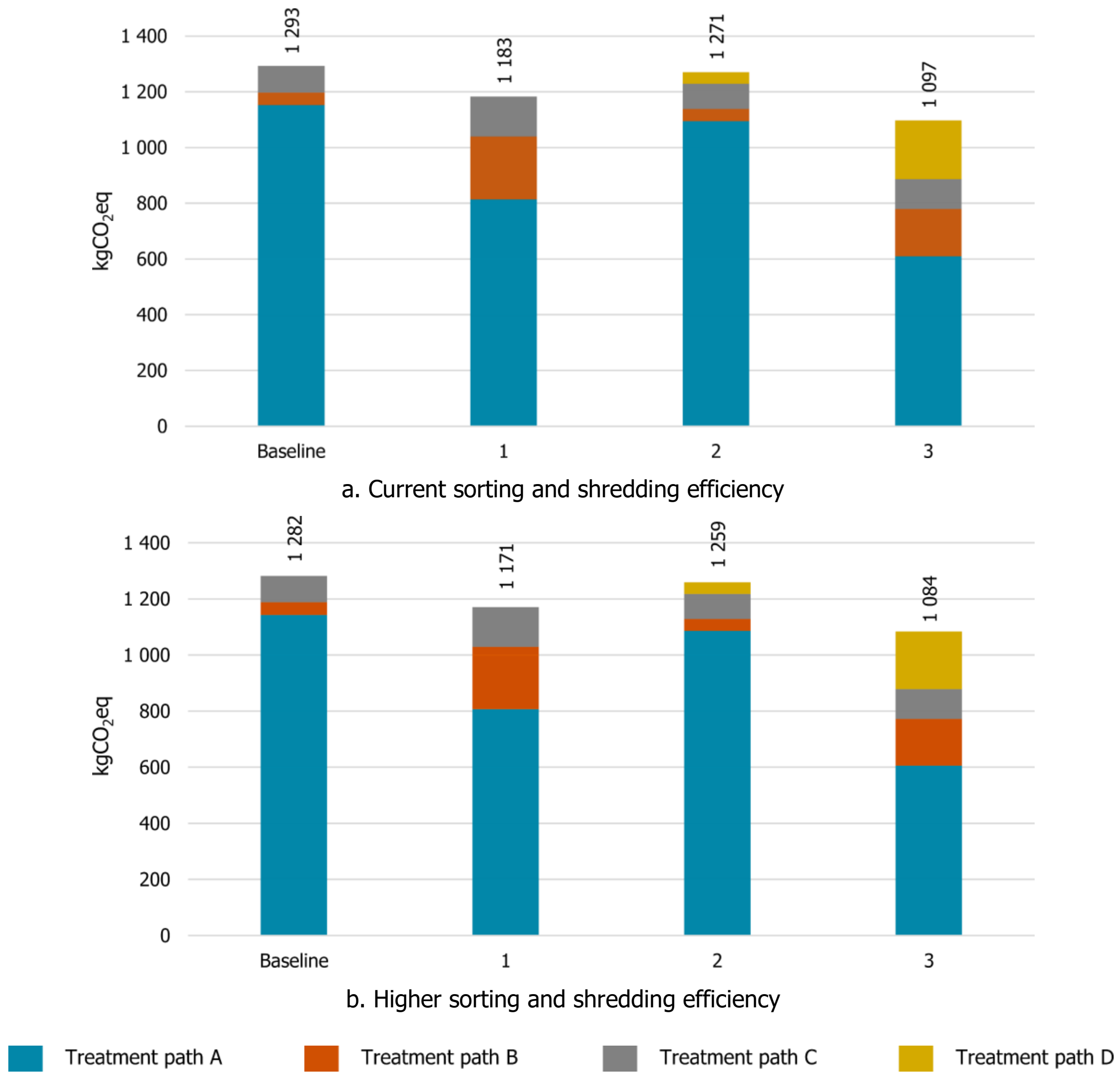



Figure 5-3. Sensitivity analysis of tablets treatment scenarios: global warming potential.

The rise in plastic recycling (scenario 3) can reduce the potential global warming impact by 17% in comparison to the baseline scenario. An increase in sorting and shredding efficiency (Figure 5-3.b) can contribute to reducing the overall impacts, but from the global warming outlook, it results in a slight decrease (1.2%).

For the toxicity, as mentioned previously, the results of path A are potentially underestimated, which influences the scenarios assessment, especially for the baseline and scenario 2. Scenario 3 with higher sorting and shredding efficiency (Figure 5-4.b) assumes that 25% of tablets were sent to facilities with plastic recycling, which results in 8.6% more impact than scenario 2 with current efficiencies and no plastic recycling (Figure 5-4.a).

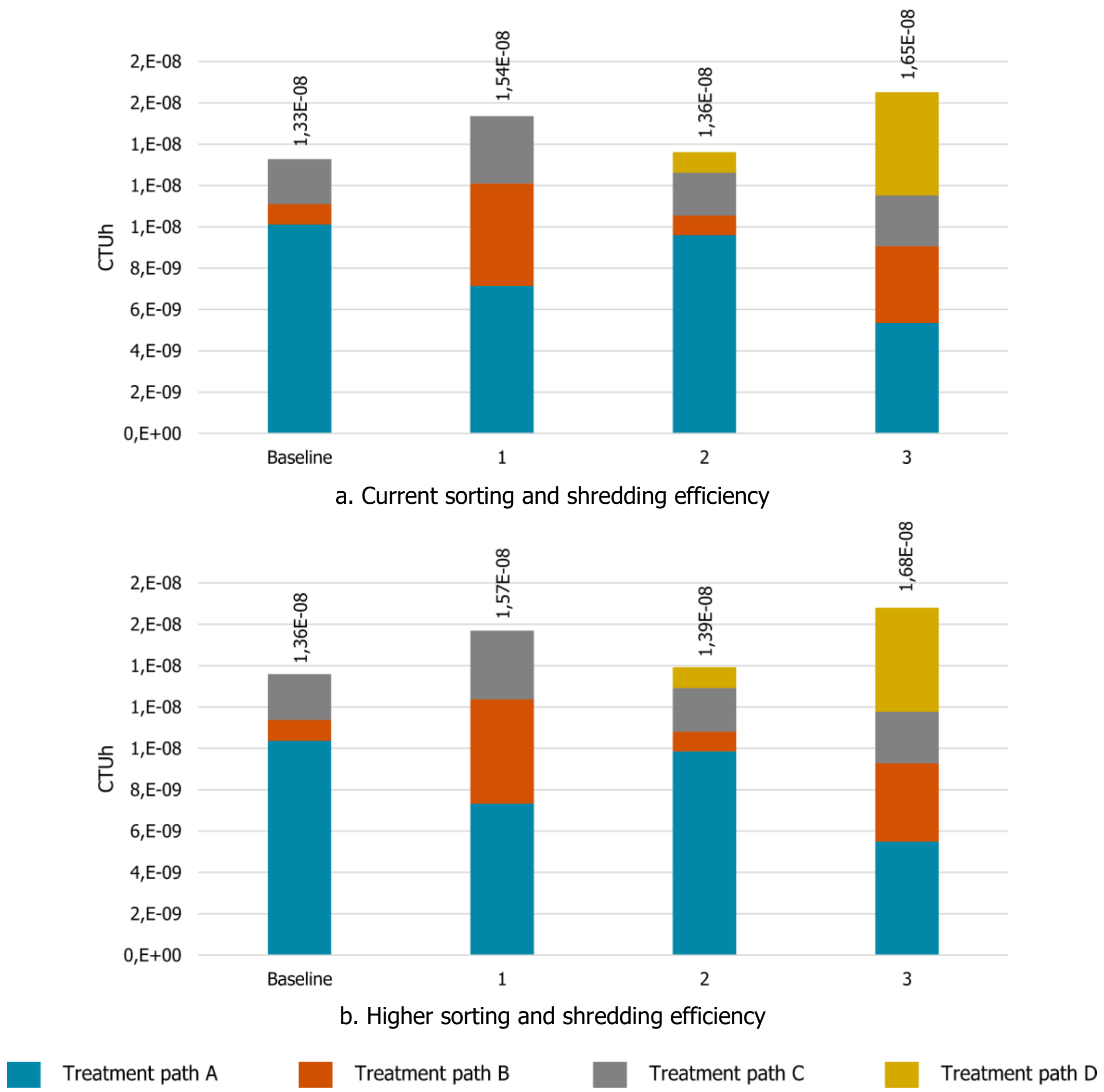



Figure 5-4. Sensitivity analysis of tablets treatment scenarios: human toxicity potential.

#### 5.5.3.2. Laptops

Table 5-10 presents the results per treatment path for 1t of laptops (approximately 500 laptops considering an average weight of 1.9 kg per device). Similar to tablet treatment, the paths with lower global warming impacts are those with manual dismantling (B and C). Nevertheless, the differences between the three scenarios are not relevant (less than 1%).

Regarding human toxicity, results for paths A and B do not vary because the same materials are recycled and/or disposed in both paths. The difference between these paths is on dismantling and component pretreatment. The datasets related to these activities account for less than 1% of the total impacts.

Table 5-10. Treatment of 1 t of laptops: impact assessment per treatment path.

| Treatment paths | Impact categories | Current performance | >2% | >5% | Coefficient of Variation (CV) |
|---|---|---|---|---|---|
| A - Battery removal followed by shredding and sorting | Global warming (kg $CO_2$-eq) | 1,358.7 | 1,351.1 | 1,340.0 | 10.8% |
| | Human toxicity (CTUh) | 3.04E-08 | 3.09E-08 | 3.14E-08 | 16.5% |
| B - Manual dismantling for depollution and removal of valuable components followed by shredding and sorting | Global warming (kg $CO_2$-eq) | 1,348.3 | 1,340.8 | 1,329.6 | 11.4% |
| | Human toxicity (CTUh) | 3.04E-08 | 3.09E-08 | 3.14E-08 | 16.5% |
| C - Idem scenario B with 9% plastics recycling | Global warming (kg $CO_2$-eq) | 1,343.1 | 1,335.6 | 1,324.4 | 11.4% |
| | Human toxicity (CTUh) | 3.03E-08 | 3.08E-08 | 3.13E-08 | 16.5% |

Path C considers plastic recycling (9% of the plastic content), which gave slightly different results in comparison to the other paths for both global warming and human toxicity. To define path C, data on the type of polymers present in laptops are considered (based on Wagner et al., 2019), as well as the potential presence of flame retardants in FPD devices (data annually assessed by the French compliance schemes).

Table 5-11 presents the different scenarios considered in the sensitivity analysis. Scenario 1 proposes an increase in manual dismantling. Scenario 2 and 3 propose plastic recycling (5% and 25%), and the share among the remaining paths is proportional to the previous scenarios.

Table 5-11. Laptops: scenarios considered in the LCIA.

| Treatment paths | Scenarios | | | |
|---|---|---|---|---|
| | Baseline | 1 | 2 | 3 |
| A - Battery removal followed by shredding and sorting | 50% | 30% | 48% | 23% |
| B - Manual dismantling for depollution and removal of valuable components followed by shredding and sorting | 50% | 70% | 48% | 53% |
| C - Idem scenario B with 9% plastics recycling | 0% | 0% | 5% | 25% |

Figure 5-5 presents the results for global warming potential. Considering that plastics treatment is the process contributing the most impact for laptops end-of-life, as the recycling of plastics in these devices is not very high, the variations in the scenarios resulted in little change in the results. The scenario with the highest shredding and sorting efficiency and recycling resulted in a potential impact 1.7% lower than the baseline scenario with current efficiency.

Seeing that an even smaller difference is identified between the paths for the human toxicity category, the variations of the scenarios do not impact the results. This is potentially due to the lack of primary LCI data to model the different recycling paths.

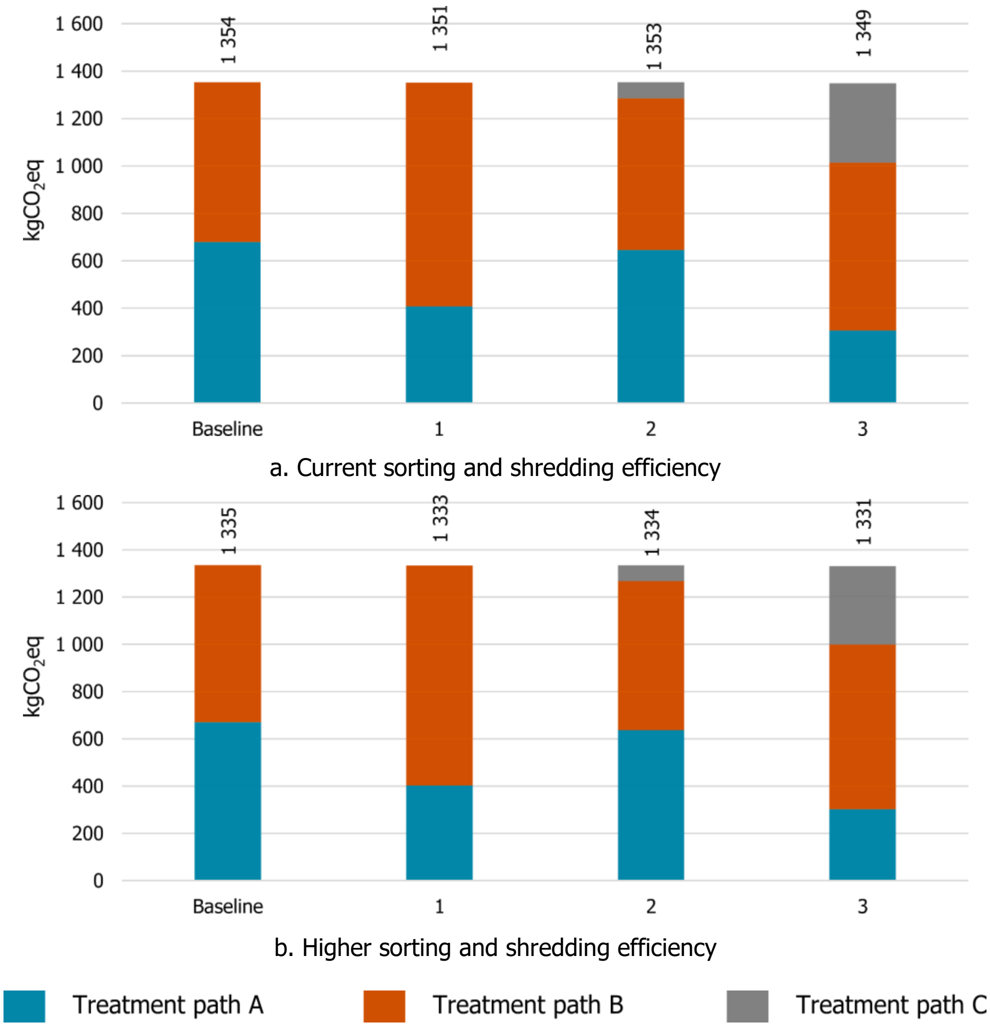



Figure 5-5. Sensitivity analysis of laptops treatment scenarios: global warming potential.

#### 5.5.3.3. FPD Monitors and TVs

Table 5-12 presents the comparison of treating 1t of FPD monitors with the three treatment paths described in **Chapter 4** and a fourth path that includes plastic recycling. Considering an average weight of 7.2 kg per device, 1t corresponds to around 140 devices.

For the global warming potential, manual dismantling generates smaller impacts than the two paths that include mechanical treatment. These results may be influenced by the lack of primary data on the energy consumption of both mechanical processes. The disposal and incineration of non-recyclable components and fractions (e.g. residues of sorting processes) accounts for around 90% of the impacts. The fourth path (including plastic recycling) generates 20% less impact than the other three paths.

Table 5-12. Treatment of 1 t of FPD monitors and TVs: impact assessment per treatment path.

| Treatment paths | Impact categories | Current performance | >2% | >5% | Coefficient of Variation (CV) |
|---|---|---|---|---|---|
| A - Manual dismantling followed by shredding and sorting | Global warming (kg $CO_2$-eq) | 1,707.5 | 1,705.0 | 1,701.6 | 16.2% |
| | Human toxicity (CTUh) | 1.54E-08 | 1.56E-08 | 1.59E-08 | 17.7% |
| B - Mechanical dismantling followed by shredding and sorting by fractions sorting | Global warming (kg $CO_2$-eq) | 1,710.7 | 1,708.3 | 1,704.9 | 15.7% |
| | Human toxicity (CTUh) | 1.54E-08 | 1.56E-08 | 1.60E-08 | 18.5% |
| C - Shredding of the whole device at negative air pressure followed by fractions sorting | Global warming (kg $CO_2$-eq) | 1,722.4 | 1,720.0 | 1,716.5 | 16.1% |
| | Human toxicity (CTUh) | 1.55E-08 | 1.57E-08 | 1.60E-08 | 17.2% |
| D - Idem B with 46% of plastics recycling | Global warming (kg $CO_2$-eq) | 1,354.5 | 1,352.1 | 1,348.6 | 12.0% |
| | Human toxicity (CTUh) | 1.84E-08 | 1.86E-08 | 1.89E-08 | 15.5% |

Concerning human toxicity, there is no significant difference between the paths A and B. Path C has an impact slightly higher due to differences in the preprocessing of the FPD devices. Contrary to global warming results, path D leads to higher impacts in terms of human toxicity due to plastic recycling.

Similarly to other screen devices, for all the treatment paths the increase of sorting and shredding performance results in a limited reduction of global warming impacts (less than 0.5%) For the toxicity, it generates higher impacts because more than 50% of the impacts are related to material recycling.

Table 5-13 presents the scenarios considered in the sensitivity analysis. Scenario 1 considers a slight increase in both mechanical treatments. Scenarios 2 and 3 include a progressive increase in plastic recycling (path D), with a proportional reduction on the other paths.

Table 5-13. FPD monitors and TVs: scenarios considered in the LCIA.

| Treatment paths | Scenarios | | | |
|---|---|---|---|---|
| | Baseline | 1 | 2 | 3 |
| A - Manual dismantling followed by shredding and sorting | 60% | 50% | 57% | 38% |
| B - Mechanical dismantling followed by shredding and sorting by fractions sorting | 30% | 35% | 29% | 26% |
| C - Shredding of the whole device at negative air pressure followed by fractions sorting | 10% | 15% | 10% | 11% |
| D - Idem B with 46% of plastics recycling | 0% | 0% | 5% | 25% |

As presented in Figure 5-6.a, the difference between the baseline scenario and scenario 1 is lower than 1%. It can be noticed that when plastic recycling is included in the scenarios, it progressively reduces the total impact.

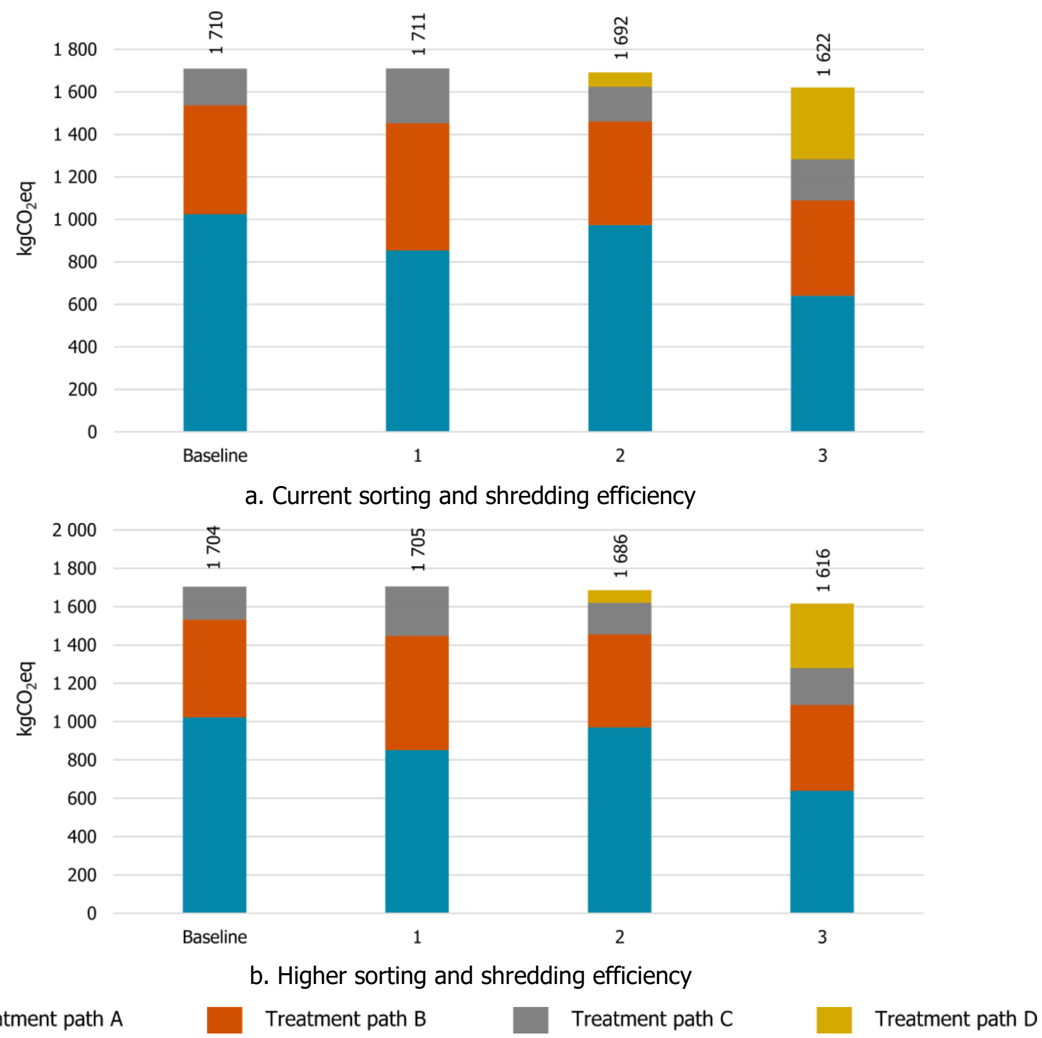



Figure 5-6. Sensitivity analysis of FPD monitor and TV treatment scenarios: global warming potential.

Scenario 3 results in 5.4% less potential global warming impacts than the baseline scenario. When combining both plastic recycling and more efficient shredding and sorting (<5%), the total impact dropped by up to 5.8% (Figure 5-6.b). Plastic recycling is limited by the presence of flame retardants, as well as by technology availability and the secondary market for the different polymers.

For the toxicity (Figure 5-7), as previously discussed, the impacts grow with the rise in sorting and recycling, mainly due to the impact of recycling processes. The difference between the baseline scenario and the scenario with the highest sorting and recycling performances is 7.6%.

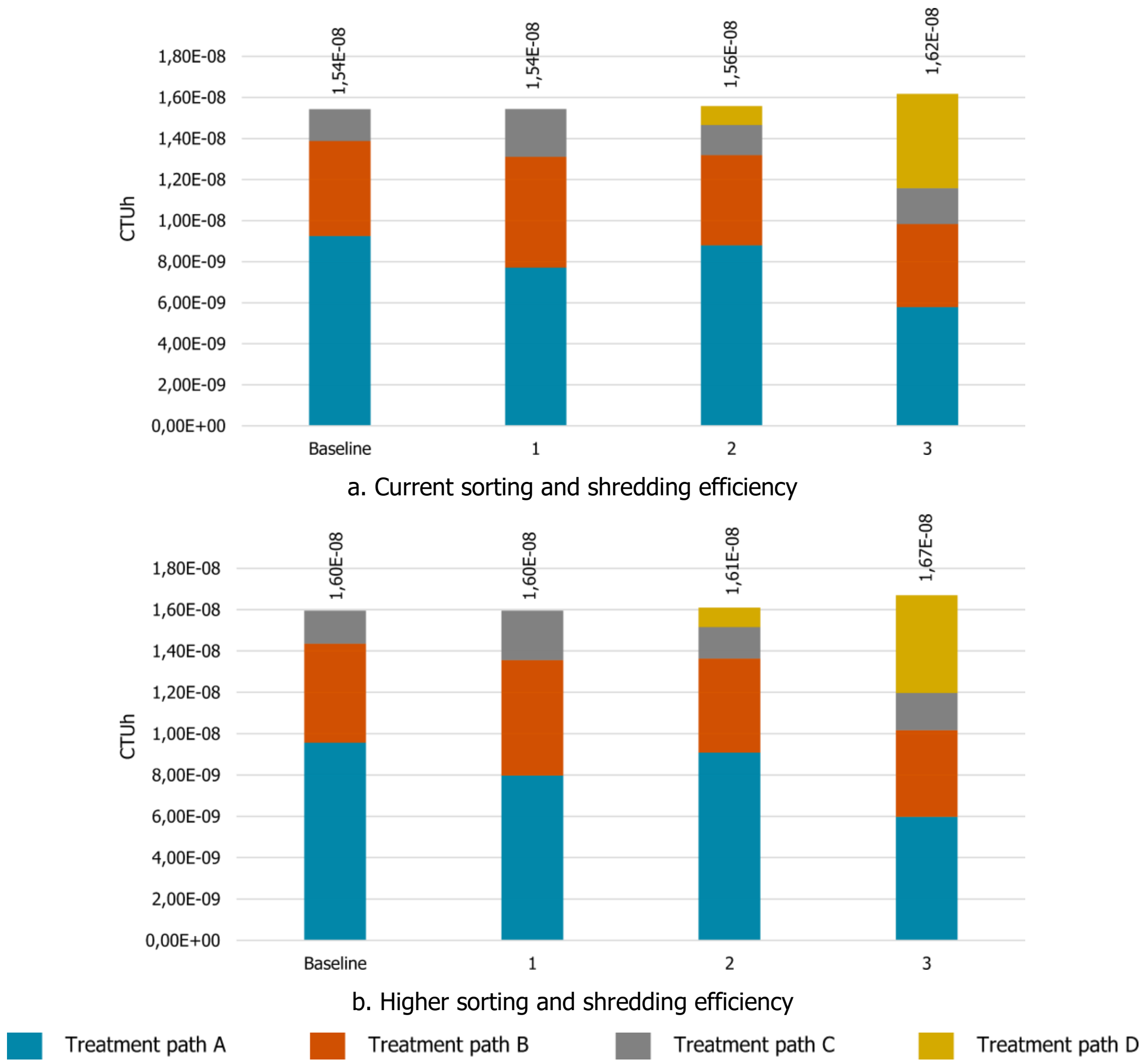



Figure 5-7. Sensitivity analysis of laptops treatment scenarios: human toxicity potential.

### 5.5.3.4. CRT Monitors and TVs

Table 5-14 shows the results per treatment path for treating 1 t of CRT monitors and TVs (approximately 20 devices considering an average weight of 19.9 kg per CRT). Concerning global warming potential, the processes with the highest contribution are those related to the disposal of non-recyclable fractions.

The increase in sorting efficiency results in a reduction of the global warming impacts of up to 2.7% for path B (the most significant influence observed among the screens assessed in this study).

Regarding human toxicity, the treatment of CRT contributes significantly to the impact category (more than 80%). Unlike global warming potential, the increase in shredding and sorting processing raises the potential impacts of toxicity. This slight increase (around 0.5%) is due to the contribution of secondary raw materials production (e.g. steel recycling) to the total impact (around 15%). The path B produces higher impacts than path A due to plastics recycling.

Table 5-14. Treatment of 1t of CRT monitors and TVs: impact assessment per treatment path.

| Treatment paths | Impact categories | Current performance | >2% | >5% | Coefficient of Variation (CV) |
|---|---|---|---|---|---|
| A - Manual dismantling, depollution, and remaining parts are forwarded to a shredder followed by fractions sorting | Global warming (kg $CO_2$-eq) | 823.1 | 816.1 | 805.7 | 16.4% |
| | Human toxicity (CTUh) | 2,14E-08 | 2,15E-08 | 2,15E-08 | 19.9% |
| B - Idem A with 40% of plastics recycling | Global warming (kg $CO_2$-eq) | 651.7 | 644.8 | 634.3 | 12.9% |
| | Human toxicity (CTUh) | 2,19E-08 | 2,20E-08 | 2,22E-08 | 19.0% |

Table 5-15 summarizes the different scenarios assessed in the sensitivity analysis, and Figure 5-8 and Figure 5-9 present the results. The scenario considering that 30% of CRT screens are treated in facilities that recycle the plastics without flame retardants reduces potential global warming impacts by 7%. Scenarios 2 and 3, with higher plastic recycling, resulted in a reduction of 11% and 17%, respectively. CRTs are composed of 16% plastics, of which 84% do not contain flame retardants. Moreover, part of the polymers present in CRT are currently recycled by the WEEE chain.

Table 5-15. CRT monitors and TVs: scenarios considered in the LCIA.

| Treatment paths | Scenarios | | | |
|---|---|---|---|---|
| | Baseline | 1 | 2 | 3 |
| A - Without plastic recycling | 100% | 70% | 50% | 30% |
| B - Considering 40% of plastics recycling | 0% | 30% | 50% | 70% |

When increasing both shredding and sorting performance and plastic recycling (Figure 5-8.b), there is a larger decrease in the impacts. Scenario 3 results in 17% less global warming potential impacts than the baseline scenario without plastic recycling and with the current sorting and shredding performance. CRT treatment is more affected by the shredding and sorting performance due to its composition (20% of metals, of which 18.8% is steel).

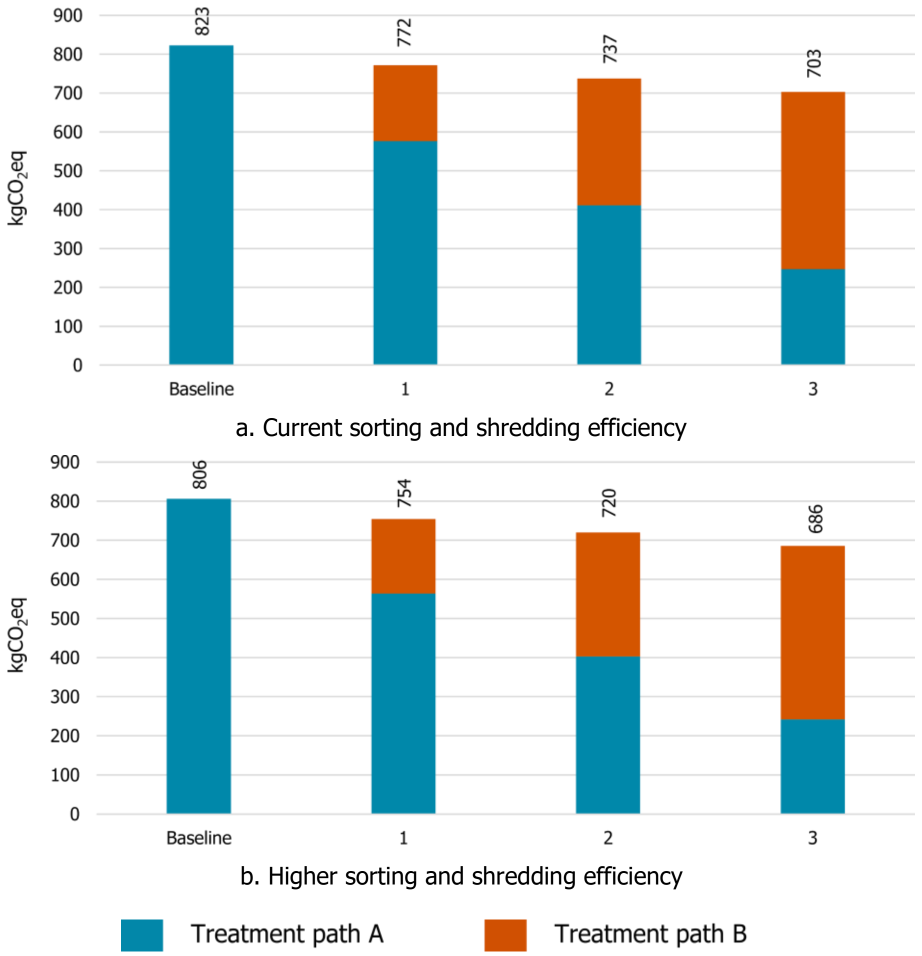
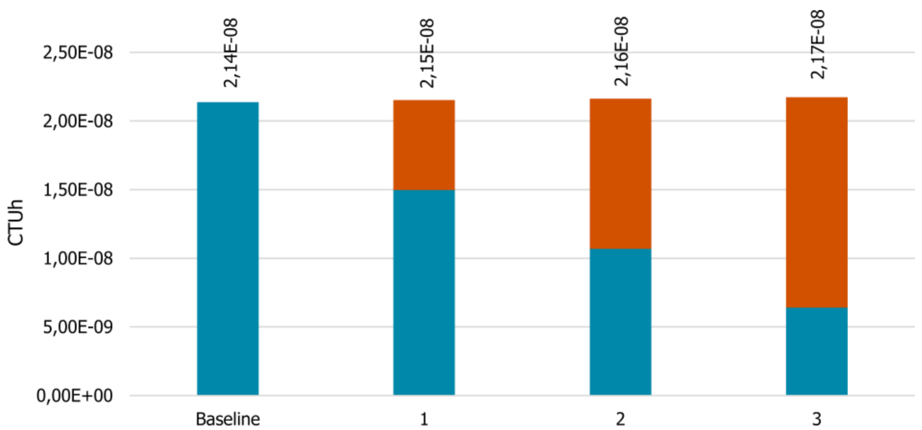



Figure 5-8. Sensitivity analysis of CRT monitors and TVs treatment scenarios: global warming potential.

For human toxicity (Figure 5-9), both the plastics recycling and the increase in shredding and sorting performance entail higher impacts. Thus, there is a rise of 1.6% when comparing scenario 3 to the baseline with current sorting performances, and 2.8% with the higher treatment performance.




a. Current sorting and shredding efficiency

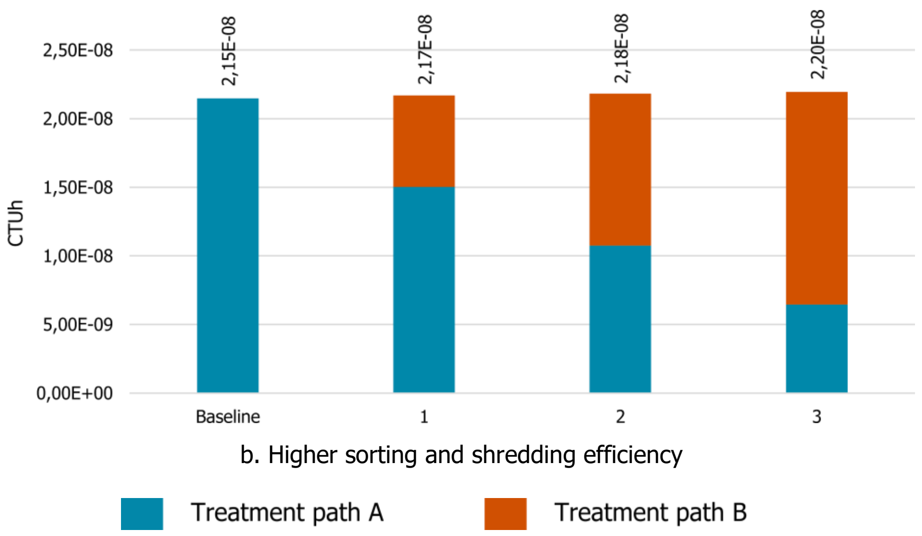



Figure 5-9. Sensitivity analysis of CRT monitors and TVs treatment scenarios: human toxicity potential.

#### 5.5.3.5. Screens treatment in France

Figure 5-10 presents the total global warming impacts of treating the screens collected in France from 2012 to 2018 based on the baseline scenarios with current sorting and shredding efficiency, as previously presented. In 2018, the treatment of 66,546 t of screens, resulted in 67,748 t $CO_2$ equivalent. For the global warming potential, 62.3% of the impacts are related to CRT devices, 34.4% to FPD monitors and TVs, 3.3% to laptops and tablets. The treatment of non-recyclable fractions accounts for 75% to 91% of the impacts.

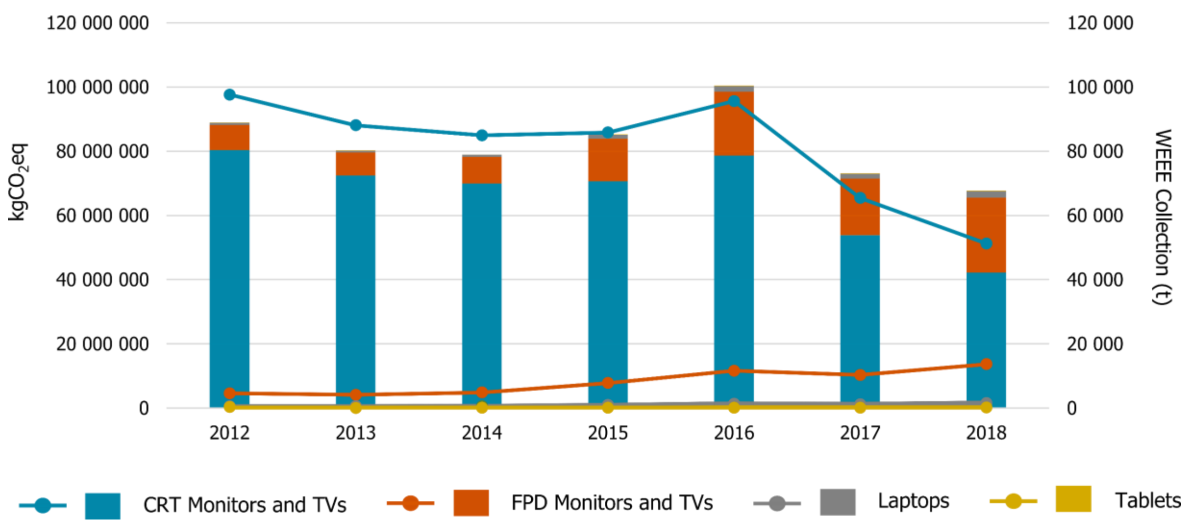



Figure 5-10. Total potential global warming impacts from treating screens collected in France from 2012 to 2018.

The total global warming impacts are decreasing over the years, with the reduction in the number of CRT monitors and TVs collected. When assessing the 2018 treatment with the highest sorting and recycling efficiency scenarios for all devices (detailed in Table 5-9, Table 5-11, Table 5-13 and Table

5-15), the results are 14.1% lower than the baseline scenario presented in Figure 5-10. The impacts of the final processes to recover the secondary raw materials (e.g. gold from PWB and steel) are not significant. Moreover, the increase in material recycling (especially plastics) can reduce the overall impact since less material would be disposed.

Figure 5-11 presents the overall impacts of human toxicity for end-of-life screens collected in France from 2012 to 2018 based on the baseline scenarios previously presented. In 2018, 80.9% of the impacts are related to CRT devices, 15.5% to FPD monitors and TVs, 3.6% to laptops and tablets.

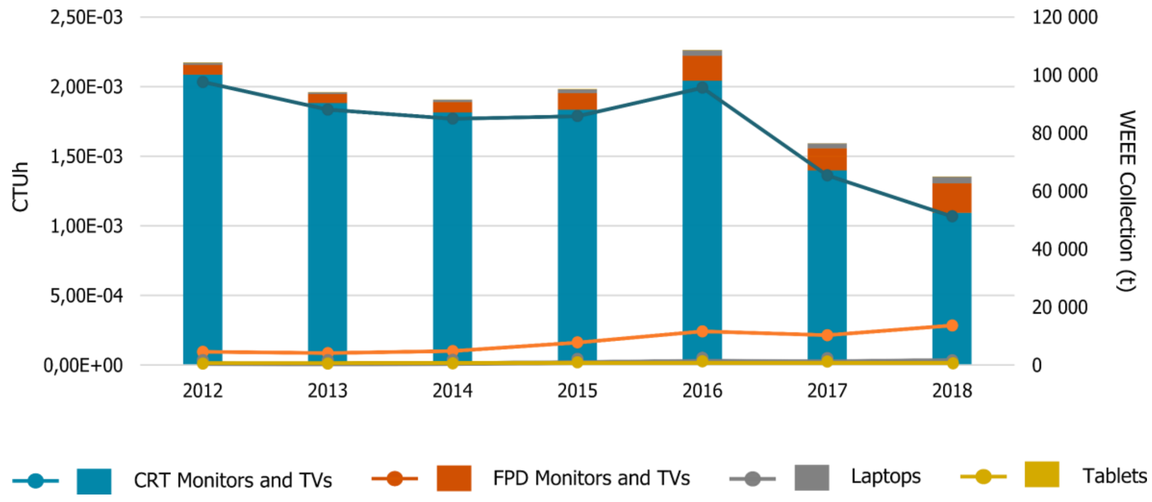



Figure 5-11. Total toxicity impacts for treating screens collected in France from 2012 to 2018.

The human toxicity impact will decrease in the following years with the progressive reduction of CRT collected by the compliance schemes. The human toxicity impact per CRT device is from 4 to 60 times higher than the impact of FPD monitors/TVs and a tablet, respectively. Thus, even if the increase in sorting and shredding efficiency and plastics recycling in screens results in higher toxicity impacts, they will still be lower than the current impact of CRT treatment.

In order to assess the benefits of recycling, we compared the impacts of the EOL with the production of the same materials deemed to be recycled. We used datasets available in the Ecoinvent database, and the assessment is focused on global warming potential (IPCC 2013).

Table 5-16 presents the results for the global warming potential of producing the raw materials available in 1t of screen devices (third column of the table). As can be noticed, the impact of producing secondary raw materials from WEEE treatment (second column of the table) is from 2 to 9 times lower than the production of the same amount of raw materials.

Table 5-16. Assessment of recycling benefits in screens treatment in 2018.

| Screens devices | Impacts ($kg\ CO_2$-eq) | | Avoided impacts ($kg\ CO_2$-eq) | |
|---|---|---|---|---|
| | EOL treatment | Raw materials | Approach 1 | Approach 2 |
| Tablets | 150,475 | 625,639 | - 475,034 | - 358,567 |
| Laptops | 2,093,734 | 19,558,273 | - 17,464,539 | - 16,171,878 |
| FPD Monitors and TVs | 23,327,636 | 90,201,822 | - 66,874,186 | - 53,487,804 |
| CRT Monitors and TVs | 42,175,455 | 96,394,048 | - 54,218,593 | - 40,685,539 |

In order to present the benefits of recycling when comparing the impacts of one ton of raw material and one ton of recycled material, it is common practice to present the results of impacts that were avoided due to secondary raw material and/or energy substitution. These results are based on the environmental value of the primary raw material that has been substituted and was therefore not extracted from ores (metals) or manufactured (plastics). A negative avoided impact means an environmental gain, because the secondary raw materials produce less impact than primary raw material production (Huisman, 2003).

When comparing the impacts of producing primary (raw) and secondary raw materials, it is important to consider that in some cases (such as WEEE recycling) both semi-closed loop recycling and open-loop recycling with downcycling may occur. For example, secondary gold and palladium from PWB are used in another product system, without the materials' inherent properties undergoing any change (semi-closed loop recycling). Conversely, the properties of recycled plastics have usually changed to such an extent that the secondary raw material cannot be used for the same application (open-loop recycling) (Ligthart and Ansems, 2012).

Based on this, the fourth and fifth columns of Table 5-16 present the results of two approaches for calculating the avoided impacts of the WEEE treatment. The first approach considers that recycling avoids the extraction of raw materials (1:1). In its turn, the second approach includes primary and secondary raw materials prices to capture differences in material quality and secondary material application. The second approach shows fewer benefits than the first because of the lower prices of certain secondary materials, especially polymers. In our opinion, the first approach is too simplistic since it considers that quality does not decrease with recycling. However, the second approach is influenced by price volatility. There are other methods in the literature that treat avoided impacts differently. This study does not focus on that aspect, and the avoided impact results are presented to demonstrate possible interpretations of the results.

Regardless of whether the avoided impact is included or not, the impacts of EOL are significantly lower than those caused by extraction of raw materials. The comparison of recycling impacts with raw material production can help communicate data on the benefits of the WEEE chain. For example, in the case study, we concluded that recycling CRT devices generates less than half the impacts as the production of the same quantity of primary raw materials.

### 5.5.4. GIS to support transport inventory modeling

Transport is inherent in recycling chains. In France, the transport sector is the main source of GHG emissions (27.8% of the total GHG) producing 136.4 Mt $CO_2$-eq in 2012 (François et al., 2017). When transport is included in LCA boundaries, there is no consensus about its influence on the results. According to Menikpura et al. (2014), in Japan the logistic chain accounted for a significant amount of greenhouse gas emissions in EOL of e-waste - depending on the type of WEEE treated. Choi et al. (2006) identified significant impacts for WEEE collection in the EOL of a computer in Korea, as well as Grimaud et al. (2016) for cable recycling in France. Conversely, Baxter et al. (2016) concluded that, whilst transport aspects are normally very significant in terms of cost, from the environmental impact standpoint its effects are relatively insignificant. Similar conclusions are addressed in a study of the WEEE chain in Italy (Biganzoli et al., 2015), and for a sound machine (Huisman, 2003).

In this context, our case study aimed to identify the contribution of transport to LCA results of e-waste recycling. A tablet treated in France was used as a case study (baseline scenario described in the previous section). In order to assess, spatialize and quantify the distances between the different stakeholders of the recycling chain, a transport inventory was performed with a Geographic Information System (GIS) and LCA coupling approach.

Data regarding road lengths, type and locations were obtained from ROUTE 120® database available in Institute of Geographic and Forest Information (IGN) website. Primary data regarding the location of the EOL stakeholders was obtained with Ecologic, and one treatment center in France was selected as a case study. Besides considering the real distances between e-waste chain actors, the study assessed the impact of lorry loading rates.

#### Modeling of WEEE chain transport

Before e-waste arrives at the treatment center, it is transported between different stakeholders of the end-of-life chain, as presented in Figure 5-12. French compliance schemes responsibility for waste treatment begins at the different collection points located across the country (e.g. retailers' collection points and municipal waste collection centers). Social and solidarity actors transfer to the recycling channel equipment and or components that could not be repaired (Vadoudi et al., 2015). After collection, e-waste can be transported directly to a treatment center (distances B2 and B3), but more often it is transferred to a regrouping center (distance A1 to A3) where the e-waste from different sources is weighed and stored per waste stream. Finally, the WEEE is transported to a treatment center where it undergoes different operational steps, according to its nature. Often, the same lorry collects waste from different collection points before returning to the regrouping center. However, due to lack of data, it was not possible to take into account the logistics optimization. To minimize the errors of not taking these into account, distances from collection points to regrouping centers are calculated as the most direct path between two points (distances A). Distances between regrouping centers and treatment center, as well as retailer's collection points and social and solidarity actors that transport waste directly to the treatment center are calculated based on the highway network (distances B). The same assumption is applied to the distances between the treatment center, recycling companies and landfill (distances C). Data concerning the percentage of e-waste collected from the different collection points in 2015, as well as the mass of different fractions after shredding and sorting is used to calculate the weight transported per kilometer.

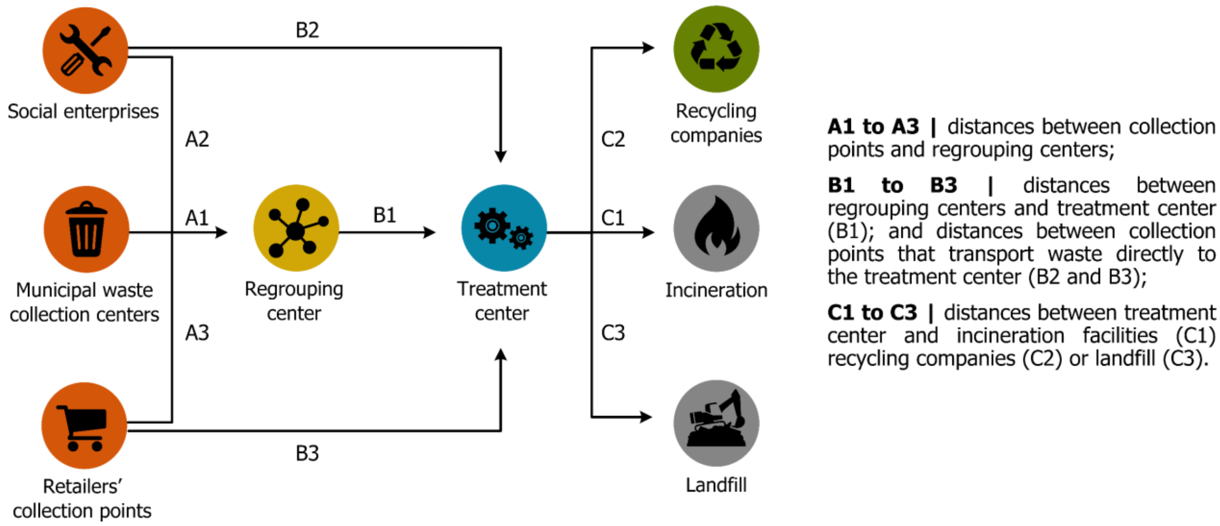



Figure 5-12. Transport in a WEEE chain.

In the Ecoinvent database, lorry processes describe the transport services with average load factors that include an average of empty return trips. Considering the types of lorries used for transport in the WEEE chain, we selected transport lorry 7.5-16 tons (Ecoinvent average load = 3.29t) for distances A and C and transport lorry 16-32 tons (Ecoinvent average load = 5.79t) for distances B. The French lorry fleet in 2015 (Euro emission standard) was taken into account in the modeling (Ministère de la Transition Ecologique et Solidaire, 2017). The locations of e-waste actors considered in the case study are presented in Figure 5-13.

As mentioned previously, the case study presents one of the many treatment centers in France, which treated less than 5% of the total amount of e-waste generated in 2015. More than 600 different stakeholders participated in the reverse logistics. For the treatment center selected in the case study, in 2015, 98% of the e-waste treated was collected in different collection points and regrouped in several regrouping centers. The distances between the different actors were calculated in QGIS software considering the assumptions previously described. The total mass transported per kilometer for each type of lorry is presented in Table 5-17.

Table 5-17. Tablets: transport activities considered in the case study.

| Type of lorry | Euro III | Euro IV | Euro V | Euro VI |
|---|---|---|---|---|
| Lorry 7.5-16 t | 64.45 tkm | 38.32 tkm | 60.67 tkm | 10.45 tkm |
| Lorry 16-32 t | 70.06 tkm | 41.65 tkm | 66.27 tkm | 11.36 tkm |

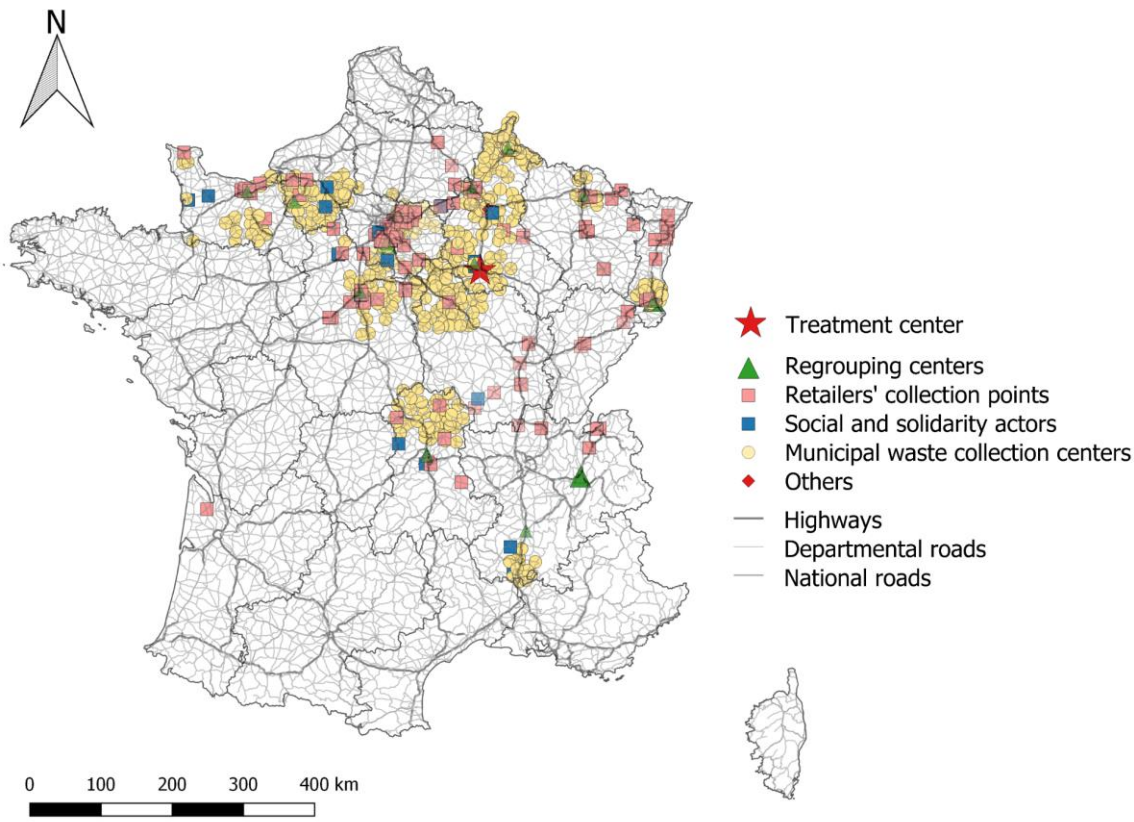



Figure 5-13. Case study: location of WEEE chain actors.

**LCIA of tablet treatment including transport**

Figure 5-14 presents the results for treating 1t of tablets in France, including the transport between EOL chain actors. Transport accounted for 4.2% to 5.0% of global warming potential, depending on the scenario considered (baseline or highest recycling and sorting efficiencies). This impact is low in comparison to the impact of activities associated with WEEE treatment until the end-of-waste status is achieved (depollution, shredding, sorting and disposal of non-recyclable fractions) – from 81.3% to 69.5% - and secondary raw materials recycling - from 14.5% to 25.6%.

The results of this case study are similar to the conclusions of some studies identified in the literature. In Japan, transport accounted for 7 to 27% of the GHG emissions depending on the type of e-waste recycled (washing machine, refrigerator, air conditioning or TV) (Menikpura et al., 2014). In

Norway, transport accounted from 6 to 16% of the total impact of e-waste treatment, also depending on the type of e-waste (refrigerator, TV or mobile phone) (Baxter et al., 2016).

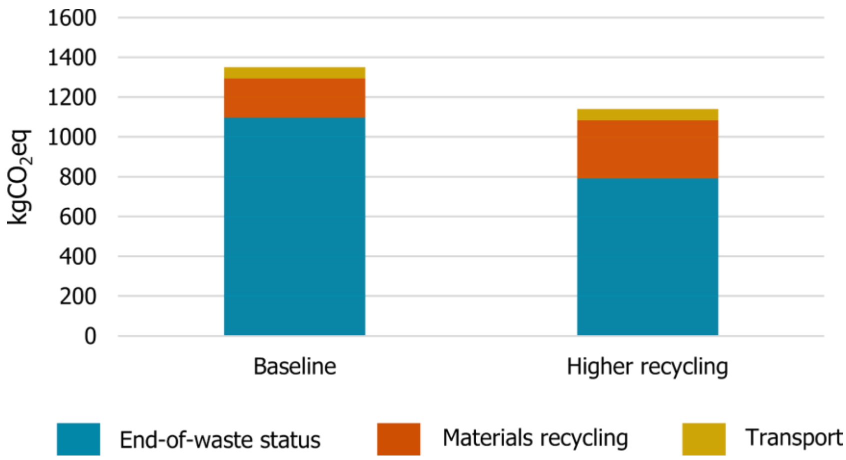



Figure 5-14. Tablets potential impacts including transport.

Based on the results, we suggest that LCA studies of WEEE treatment include the transport when primary data is available. In the absence of data, we recommend not including transport in the system boundaries of the study, rather than adopting average distances that do not represent the complexity of e-waste collection logistics.

GIS is a tool for capturing, manipulating, analyzing, managing and presenting spatial or geographic data from numerous sources. LCA and GIS coupling is a recent field of study and presents potential improvements for both inventory and impact assessment phases. In the case study, GIS was used to calculate the distances to reduce the time needed to obtain individual calculations for the more than 600 distances assessed.

## 5.6 LCA to support decision making in WEEE management

Based on the literature review and the case study results, we confirm that LCA can provide interesting insights into the impacts and benefits of the WEEE chain. However, due to limitations in the current LCI databases and methodological choices, the results should be carefully analyzed.

The main limitation of the case study presented in section 5.5 is the lack of LCI data which results in restricted treatment paths and scenarios which are not completely representative of the reality on the ground. As explained in section 5.4.2, we could not use the ESR database due to the format of the datasets and the lack of data on CRT treatment. In turn, the Ecoinvent database has few treatment paths for WEEE.

In this study, we had little access to primary data on the inputs and outputs of the different treatment steps of e-waste treatment. Nevertheless, to obtain more robust results, it is very important to gather primary data. E-waste treatment operators can monitor LCI data and eventually perform gate-to-gate studies. For example, treatment operators can use LCA data to compare different treatment paths and select treatment options that result in lower environmental impact (Grimaud, 2019; Grimaud et al., 2018).

Although LCA is the most widely used method for assessing the environmental impacts of WEEE treatment, its use by the WEEE chain actors is very challenging. As illustrated in the case study, an indicator providing data on the environmental impact per weight of e-waste recycled requires a large amount of data that is not monitored by the compliance schemes.

Aiming to include an environmental assessment in the WEEE chain monitoring, we recommend adopting different approaches depending on the goal and use of the environmental assessment.

As mentioned earlier, LCA can be successfully used for comparing the impacts of treatment paths. Compliance schemes could include the need for reporting the environmental impacts per ton of e-waste collected/treated in their call for tenders for e-waste treatment providers. This information could be used to select the e-waste providers, together with other technical and economic parameters already considered. Moreover, this data could be used to improve the ESR database.

Nevertheless, before adopting LCA results as indicators to compare the environmental performance of WEEE treatment providers, it is necessary to set general rules and guidelines. Otherwise, each treatment provider could adopt different boundaries and methodological approaches which would entail in difficulties understanding and comparing the results. These guidelines should be similar to the Product Category Rules (PCR) developed to support environmental product declarations (EPDs) for a specific product category. They should be included in the specifications provided by the Environmental Code (see **Chapters 1** and **2** for further information).

Regarding the compliance schemes' reporting on the environmental impacts and benefits of the WEEE chain, in the following section we suggest two indicators based on the LCI data provided by the ESR database. Despite the limitations of the ESR database discussed in section 5.4.2, we believe this is the best information available for use by WEEE chain actors. Nonetheless, the database should be expanded to include other types of WEEE and there should be more transparence on the methods and treatment scenarios.

## 5.7 Environmental indicators for reporting WEEE chain performance

In this section we suggest two environmental indicators to be used by the compliance schemes to report the WEEE chain environmental performance. We suggest applying them to each category and for different target elements and materials. It is important to remember that the assessment is limited by the datasets available in the ESR database.

### 5.7.1. Environmentally weighted-based treatment rate

The goal of this indicator is to assess the treatment rate by the official schemes by including, in addition to the weight of the target elements, their environmental impact including the benefits of substitution (avoided impacts). It uses data on the composition of e-waste generated and collected by the official schemes presented in **Chapter 4**, and information on the avoided impacts by the WEEE treatment available in the ESR database.

The indicator is not based on the fraction recycled but on the WEEE treated because the results of the LCI database are modeled for the treatment of one ton of elements/materials and comprise recycling, energy recovery and landfilling. According to ESR, the substitution ratio (not specified in the database) was determined on the basis of physical considerations, not by conducting a price comparison between the sale prices of new and recycled materials (ESR and Bleu Safran, 2018).

The indicator is the ratio between the sum of elements collected by the official schemes divided by the sum of elements generated multiplied by their avoided impacts (Equation 5-1). It should be calculated per category.

$$ETR_f = \frac{\sum_{l=1}^{n4} \sum_{i=1}^{n1} WC_i \times \sum_{j=1}^{n2} CH_{elij} \times AI_{elij}}{\sum_{l=1}^{n4} \sum_{i=1}^{n1} WG_i \times \sum_{j=1}^{n2} CH_{elij} \times AI_{elij}}$$

| | |
|---|---|
| $ETR_f$ | environmentally weighted-based treatment rate per WEEE category; |
| $n4$ | number of target elements considered in the assessment; |
| $n1$ | number of different UNU-Keys related to the WEEE category; |
| $WC$ | weight of e-waste collected by the official schemes; |
| $n2$ | number of components and materials present in the different products; |
| $CH_e$ | element content per component and materials in the products; |
| $Al_e$ | environmental impact of the target element including the benefits of substitution (avoided impacts); |
| $WG$ | weight of e-waste generated. |

Equation 5-1. Environmentally weighted-based treatment rate.

The impacts avoided by the treatment of all WEEE generated (denominator) represent an optimal scenario in which all WEEE is collected and treated by official systems. Thus, the result of the indicator indicates how close the current performance is to the optimal scenario. The users can decide on the LCIA method and the impact category to carry out the assessment (e.g. global warming potential, human toxicity, etc.). We recommend using the methodologies suggested by the EC (Fazio et al., 2018).

This indicator can help compliance schemes to prioritize the treatment of the elements with the highest environmental interest in order to achieve the optimal scenario. Figure 5-15 presents the results of the treatment of FPD screens in France based on four target elements. When assessing the treatment rate with the weight-based approach (Figure 5-15a), the most prevalent material is copper and aluminum. However, when the avoided impact approach is considered in the calculation (Figure 5-15b), the elements' shares change, especially for gold.

In 2018, the weight-based treatment rate for the four target elements in FPD was 26.7%, and the environmentally weighted-based treatment rate was 22.7%. This difference is due to the *quality* of the elements treated by the official chain. Thus, to improve the environmental performance, instead of increasing the total weight treated, efforts can be made for some elements. For example, if the official schemes had treated 500 kg of gold more and 1 t of aluminum less, it would have resulted in a slight change in the weight-based treatment rate, but in a significant increase of the environmentally weighted-based treatment rate (28.65%).

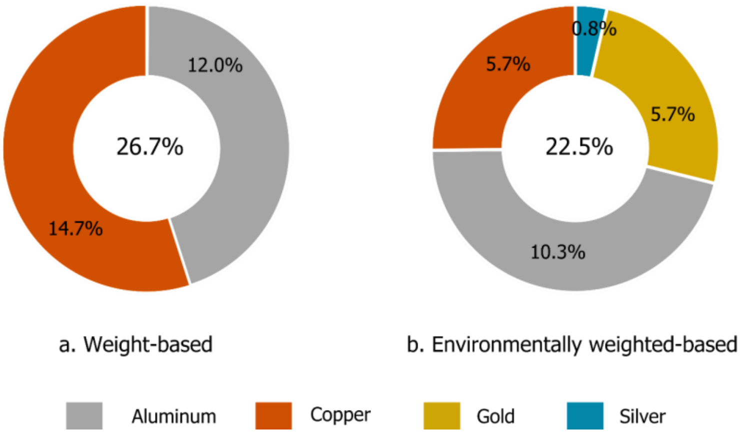



Figure 5-15. Treatment rate of aluminum, copper, gold, silver in FPD screens in France: comparison of two approaches.

### 5.7.2. Environmental impact avoidance opportunity

This indicator quantifies the opportunity to avoid the environmental impacts of the current consumption of target elements by the French economy, by the treatment of WEEE by the official chain (Equation 5-2). The WEEE chain capacity to supply secondary raw materials for the economy is at the core of this approach.

$$WTO_f = \frac{\sum_{l=1}^{n4}\sum_{i=1}^{n1}(WG_i - WC_i) \times \sum_{j=1}^{n2} CH_{elij} \times AI_{elij}}{\sum_{l=1}^{n4} AC_{el} \times IRM_{el}}$$

$WTO_f$ environmental impact avoidance opportunity with WEEE treatment by the official schemes (for each WEEE category);

$n4$ number of target elements considered in the assessment;

$n1$ number of different UNU-Keys related to the WEEE category;

| | |
|---|---|
| $WG$ | weight of e-waste generated; |
| $WC$ | weight of e-waste collected by the official schemes; |
| $n2$ | number of components and materials present in the different products; |
| $CH_e$ | element content per component and materials in the products; |
| $Al_e$ | environmental impact of the target element including the benefits of substitution (avoided impacts); |
| $AC_e$ | apparent consumption of the target element in France; |
| $IRM_e$ | environmental impact of the raw material production. |

Equation 5-2. Environmental impact avoidance opportunity with the increase of WEEE official schemes treatment.

Like the previous indicator, it uses data on the weight of target elements (according to the approach introduced in **Chapter 4**), the avoided impacts by the treatment of target elements (ESR database), and additionally it considers the apparent consumption of the same raw materials in France, and the impact of raw material production. The performance is assessed per WEEE category.

In order to determine apparent consumption in France, we used data on the EU consumption quantified in the criticality study of the EC (Deloitte Sustainability et al., 2017a, 2017b), and an average of the overall percentage of metals consumed in France according to Eurostat statistics (EUROSTAT, 2019). We suggest using more accurate data on consumption in France estimated by the Statistical Studies Department of the General Commission for Sustainable Development (Calatayud and Mohkam, 2018). We did not have access to this data for this estimation, but we believe it is possible that ADEME may be able to access it.

Figure 5-16 presents the results of the indicator for FPD screens treatment in France including four target elements (aluminum, gold, silver and copper).

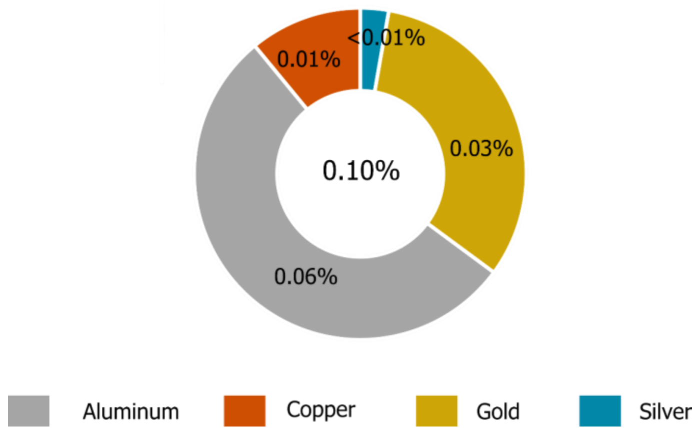



Figure 5-16. Avoided impact opportunity with the increase of WEEE treatment: focus on four target elements.

In 2018, the target elements not captured by the official schemes (dissipated in the complementary flows) could have avoided 0.10% of the impact of the overall demand of these raw materials by the French economy. If more target elements were treated by the official schemes this result would decrease, indicating that the performance is progressing towards the optimal scenario.

It is important to clarify that the result of the indicator is low because it considers the contribution of the EEE sector as a supplier of secondary raw materials to the total demand of the economic sectors. For some elements, such as copper and aluminum, overall demand is much higher than the maximum supply of secondary raw materials available from the EEE sector. However, for other elements such as gold and cobalt, the EEE content and, consequently the potentially available secondary raw material, is closer to the total demand.

Similar to the previous indicator, this indicator enables the identification of the target elements that, by increasing the supply of secondary raw materials, could make the highest contribution to reducing the impacts of raw materials production.

## 5.8 Conclusions

This chapter explores the potential of Life Cycle Assessment (LCA) methodology and its results for defining indicators for the Waste Electrical and Electronic Equipment (WEEE) chain. A literature review on the LCA method (section 5.2) and studies applied to WEEE chain (section 5.3) allowed an understanding of the state of the art of LCA application in the literature, as well as the common points and recurrent difficulties of the studies. The source of secondary data (databases) and result uncertainties are the two main issues identified through the literature review. These points were addressed in section 5.4.

Aiming to quantify the impacts and benefits of the WEEE official schemes in France, we performed an LCA study of screens recycling (section 5.5) from 2012 to 2018. We assessed different treatment scenarios, including variable combinations of treatment paths for each screen device (sections 5.5.3.1 to 5.5.3.4). Then, we estimated the total impacts of the screens recycling per year focused on potential global warming and human toxicity impacts (5.5.3.5). The influence of transport on WEEE reverse logistics and treatment are discussed in section 5.5.4, with a case study on tablets. The transport inventory was performed with a Geographic Information System (GIS) and LCA coupling approach.

We concluded that the total global warming and human toxicity impacts are decreasing over the years, with the reduction in the amount of CRT monitors and TVs collected. Most of the potential global warming impacts are related to treatment of the non-recyclable fractions, thus the increase in materials recycling (especially plastics) can reduce the overall impact. The human toxicity impact per CRT device ranges from 4 to 60 times higher than that of FPD monitors/TVs and a tablet, respectively. Therefore, even if the increase in sorting and shredding efficiency and plastics recycling in screens entails higher toxicity impacts, they are lower than the current impact of CRT treatment.

The LCA results provided interesting insights into the impacts of the WEEE chain, but the results are influenced by data limitations. We believe that LCA can be used to assess the impacts and benefits of WEEE chain, but, as pointed out in section 5.6, the compliance schemes do not have the LCI data required to perform accurate studies. We suggest that the selection of e-waste providers by the compliance schemes, includes environmental requirements, together with other technical and economic already considered, so that life cycle inventory data can be monitored. This information could also be used to improve ESR database.

Finally, in section 5.7 we propose two indicators to be used by the compliance schemes based on LCA results considering the ESR database to assess the benefits of WEEE chains, and to assist in prioritizing the recovery of elements which are significant in terms of environmental performance.

Table 5-18 summarizes the environmental indicators and the sections in which the indicators are proposed and validated. This set of indicators aims to reply to the third research question of this thesis, presented in **Chapter 1,** regarding the metrics for evaluating the environmental impacts and benefits of the WEEE chain.

Table 5-18. Summary of the environmental indicators proposed in Chapter 5.

| | Economic indicators | Equations | Validation with case study |
|---|---|---|---|
| 1. | Environmentally weighted-based treatment rate | Equation 5-1 | Section 5.7.1 |
| 2. | Environmental impact avoidance opportunity | Equation 5-2 | Section 5.7.2 |

Together with previous studies published in the literature (Ardente et al., 2014; Haupt edet al., 2016; Huisman, 2003; Nelen et al., 2014; Parajuly et al., 2017; Tansel, 2017; Van Eygen et al., 2016), these indicators aim to include the environmental priorities related to raw materials, beside the current weight-based approach. Moreover, the WEEE chain capacity to supply secondary raw materials for the economy is also included in the second indicator.

As with the technical indicators proposed in the previous chapter, the feasibility of the environmental indicators relies on the availability of data to calculate them. In addition to the waste flows data addressed in **Chapter 4**, it requires environmental data that is partially covered by the ESR database, and information on the apparent consumption of raw materials in France. With the current datasets available in this database, only a few elements and materials can be assessed, and only for some of the products in WEEE categories. We suggest expanding the current ESR database, with more transparency on the scenarios and methodological choices.

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## Chapter 6 | Economic performance of WEEE chain

*"Se aprendesse qualquer coisa, necessitaria aprender mais, e nunca ficaria satisfeito.*[17] *"*
Graciliano Ramos, Vidas Secas (1938).

### 6.1 Assessment of WEEE chain economic performance

As discussed in the previous chapters, the Waste Electrical and Electronic Equipment (WEEE) regulation in the European Union (WEEE Directive) does not propose economic indicators and targets. At the national level, according to the specifications included in the Ministerial Orders for household and professional WEEE (JORF, 2015, 2014), the French compliance schemes must report indicators regarding their income and expenses to ADEME and the Ministry of the Environment.

This chapter presents the WEEE chain economic model, and details the economic indicators currently used by the French compliance schemes (section 6.2). An expansion of the current indicators based on the literature review is presented in section 6.3 and validated with a case study (section 6.4).

We propose new indicators to improve the monitoring and reporting of the costs and benefits of the WEEE chain, and an alternative approach for the weight-based indicators (e.g. collection and recycling rate). The potential users of the indicators are compliance schemes and policymakers. Nevertheless, the information reported by the indicators is useful for different stakeholders – including consumers.

### 6.2 WEEE chain economic model

The funding mechanism of e-waste take-back activities and the allocation of economic responsibilities along the material chain is complex and challenging (UNEP, 2011). It requires substantial and sustained dialogue between government, producers, compliance schemes and other relevant stakeholders (McCann et al., 2015).

There are few studies on the actual costs of WEEE schemes and the functioning of the financial system (Huisman et al., 2019a). The economic models identified in the current WEEE schemes around the world are briefly described below (Huisman et al., 2019b; McCann et al., 2015):

- Up-front fees paid by the producer when the product is placed on the market;
- Visible fees paid by final users to cover waste management costs of historic e-waste generated;
- End-of-life (EOL) fee paid by generators of e-waste (person or entity responsible for disposing of the e-waste) which covers the collection and recycling costs;

[17] If he learned anything at all, he would need to know more and would never be satisfied.

- Placed on the market (POM) based market share where operational costs of running the take-back system are allocated on the volume of product placed on the market in a given time frame;
- Return share compliance cost in which the financial responsibility is assigned based on return market share by auditing of the e-waste that is being returned through the take-back system.

In the European Union (EU), the WEEE legislation allows the Member States to decide on the allocation of financial responsibilities in Extended Producer Responsibility (EPR) regimes (European Commission, 2012). In most of the Member States, including France, producers pay a fee to a compliance scheme on a weight or unit basis, and a visible fee is paid by the consumers (Gregory et al., 2009). Regardless of the WEEE chain funding system, its overall profitability is essential. Materials recycled or recovered from e-waste provide revenues for the system and act as cost compensation (Wang, 2014).

According to UNEP, WEEE chain costs can be divided into technical (include the cost of collection, transportation, dismantling, recycling and disposal) and administrative (e.g. expenses related to fee collection, fund administration, and auditing) (UNEP, 2011). Li et al. (2016) identified that the logistics cost is of the greatest concern in WEEE treatment. Other activities with economic impact are dismantling and pretreatment fees, disposal fees, sales revenue, and labor costs. Conversely, fund audit, and collection costs are of less concern.

Several factors affect the functioning of the WEEE schemes, including economic factors, and may influence the overall performance of the e-waste chain. Table 6-1 summarizes some economic factors and their influence on treatment performance. Some factors are dependent on dynamics not under control of the WEEE chain, while others are directly linked to the compliance with legal requirements and distortions happening on the market (Magalini and Huisman, 2018; Wang, 2014).

In France, the compliance schemes monitor the costs and revenues of e-waste treatment and partially report them to ADEME. Treatment operators also track their expenses and profits, but this study does not evaluate these aspects in detail. Our analysis focuses on the economic performance of WEEE schemes from the viewpoint of the compliance schemes, as they are the key players in the French EPR system.

According to the specifications included in the Ministerial Orders (JORF, 2015, 2014), compliance schemes must provide annual indicators related to their revenues and expenses. There is no greater precision on the scope of the indicators. The current indicators reported to ADEME allow the monitoring of the total costs and incomes of the EPR system considering the end-of-waste status.

Table 6-1. Economic factors influencing the performance of e-waste treatment.

| Influential factor | Influence on treatment performance |
|---|---|
| Positive market value of certain WEEE components/materials/ Elements | Development of treatment operators and secondary market for the recycled materials;<br>Scavenging of products and components (complementary flows) which reduces the profits of the official schemes; |
| Quantity of e-waste collected | Economy of scale (some treatment requires a minimum mass to be profitable);<br>Overall economic performance; |
| Logistic costs | Cost of transposition from collection points to treatment facilities affects the economic performance; |
| Labor costs | Amount of manual work (such as dismantling and sorting) and other labor expenses entails lower profits; |
| Investment, treatment subsidy or service charges | Selection of treatment equipment and facilities (capital and operational expenses may affect technology selection both when implementing the facility and in daily operation);<br>Overall profitability of treatment; |
| Materials/elements price | Order of priority for materials recycling;<br>Selection of recycling techniques and equipment;<br>Overall revenue. |

Source: based on Magalini and Huisman, 2018 and Wang, 2014.

As detailed in **Chapter 3**, the current economic indicators quantify the costs of the compliance schemes and the revenues of the fractions commercialized for further material recovery. The amount collected by the visible fee of products placed on the market is also reported. The expenses include logistic and treatment costs, contribution to the municipalities for the collection of household e-waste, communication expenses, contribution to research and development (R&D) studies, amongst others.

The compliance schemes select the treatment operators through an open call for tenders. Thus, the logistic and treatment costs correspond to the price paid by the compliance schemes to the different companies in charge of e-waste collection, regrouping, and treatment. It includes equipment investments and the ongoing cost of running the treatment activities (e.g. operational expenses and labor costs), as well as the profits of the operators. As stressed by Wang (2014), treatment operators pursue the highest profitability of the system.

The indicators currently used by the compliance schemes monitor neither the costs of material recovery associated with the last stage of fractions recycling nor the price of the secondary raw materials. These aspects are left out of the existing assessment scope.

### 6.2.1. Visible fee and eco-modulation

As detailed in **Chapter 2**, in France, producers of household EEE bear costs for e-waste management by either joining a compliance scheme or setting-up a take-back system. They must also inform the consumers of the end-of-life treatment costs by indicating, on the sales price, the amount of the visible fee ('éco-participation' or 'éco-contribution' in French). It guarantees the financial security and transparency of the system, but it also represents an administrative burden to the producer (European Commission et al., 2018). Distributors transfer to producers, who pay a subscription to the compliance schemes, the exact amount of the visible fee they have collected from consumers (Gregory et al., 2009).

The visible fee was established to allow the treatment of equipment placed on the market before the set-up of the WEEE chain (August 15, 2005) and for which the producers are not responsible (Deprouw et al., 2018). When it was implemented, the legal text said that the fee would only be applicable until February 2013. Nevertheless, seeing the importance of this mechanism to finance the WEEE chain, as well as to foster the collection and recycling of WEEE this limit was postponed until January 2020 (Más, 2016). According to ESR, the amount recovered through the visible fee is used to cover operational costs (70%), to support e-waste collection (22%), to cover administrative expenses (5%) and with campaigns to promote e-waste treatment (3%)[18].

The visible fee varies according to the type of equipment, and the compliance scheme to which the producer is under contract. Since July 2010, the Ministry of the Environment imposed a modulation of the visible-fee considering its impact on the end-of-life (Deprouw et al., 2018). Globally, it aims at promoting financial responsibility for the true costs of the management of the products put on the market by the producers (Monier et al., 2014). These modulation criteria are linked with the suitability for repair and reuse, clean-up, recyclability and waste prevention. The producers must provide supporting documentation of the products concerned by the eco-modulation criteria.

Table 6-2 presents the modulation criteria established by the latest household EEE specifications for category II equipment (see columns 'criteria', 'modulation' and 'rules'), and the financial contribution in force from August 15, 2018 by Ecologic (column 'price without tax').

As can be noticed, two approaches were chosen by the legislator, according to the products and the modulation criteria applied to them. The first approach aims to apply best practices and implies a sanction if producers do not meet the modulation criteria. The second approach seeks to encourage more ambitious best practices and provides a bonus if the modulation criteria are respected.

The financial contribution set by the compliance schemes to products of the same (W)EEE category are similar, but in some cases, they define different technical criteria to set the price range of the visible fee. For example, in the case of monitors (category II), as presented in **Chapter 2,** two compliances schemes have been approved by the public authorities for the management of WEEE in France. Both set prices depending on the equipment weight, but ESR defines the prices for monitors below or above 5 kg, while Ecologic sets it below or above 6 kg.

[18] Available in: https://www.eco-systemes.fr/particuliers/recyclage/qui-finance-le-recyclage

Table 6-2. WEEE category II: visible-fee modulation criteria and financial contribution according to Ecologic.

| Device | Weight (kg) | Price without tax (€) | Criteria | Modulation | Rules |
|---|---|---|---|---|---|
| Tablets | N/A | 0.25 | Presence of plastic parts > 25g containing brominated flame retardants; **OR** Lack of mutually compatible software updates, essential for the basic use of the device | +100% | If one (or both) of the criteria apply to the EEE, the contribution is increased |
| Laptops | < 2 | 0.30 | Absence of paint and coatings incompatible with recycling and reuse on plastic parts >100g; **AND** Incorporation of post-consumer recycled plastic (minimum threshold of 10%); **AND** Product upgrade with standard tools, including memory drives, chips and cards | -20% | If the EEE simultaneously meets the 3 criteria, the contribution is reduced by -20% |
| | ≥ 2 | 0.42 | | | |
| Monitors | < 6 | 1.42 | Equipment not concerned by eco-design criteria | N/A | N/A |
| | ≥ 6 | 2.50 | | | |
| TVs | < 7 | 4.17 | Provision of technical documentation for electrically authorized repairers and essential parts for equipment use (electronic boards) for 5 years; **OR** Incorporation of post-consumer recycled plastic (minimum threshold of 10%) | -20% | If the EEE meets one of the 2 criteria or both, the contribution is reduced |
| | 7 ≥ x < 12 | 7.50 | | | |
| | 12 ≥ x < 17 | 9.17 | | | |
| | 17 ≥ x < 25 | 12.5 | | | |
| | ≥ 25 | 12.5 | | | |

Source: based on data from Ecologic, 2018; OCAD3E, 2015.

### 6.2.2. Collection and treatment costs from the outlook of French compliance schemes

There is limited information publicly available in France, as well as in other Member States, on the costs and revenues of the WEEE schemes. This is a consequence of the competitive market between the compliance schemes that, therefore, do not share detailed economic information (Monier et al., 2014). According to Favot et al. (2016), their study on the economic performance of WEEE chain in Italy was possible because they had access to additional data.

Thanks to the fact that ADEME is supporting this study, we had access to internal reports that contain data on the compliance schemes' revenues and costs. However, only the overall revenues and costs are reported by the compliance schemes, not allowing an assessment per category or waste stream. In the annual public report released by ADEME, only the total amount of visible fees received by the compliance schemes is given.

Table 6-3 summarizes the revenues reported from 2015 to 2018 for household WEEE. Similar monitoring is also performed for the professional e-waste chain. The visible fee represents around 70% of the revenue source of the e-waste chain. The total amount received from the visible fee increased by 80% from 2015 to 2018. The current fee modulation system is closer to a "true EOL cost", which may explain this change. Moreover, only half of e-waste generated is collected by the official schemes (collection rate was 49% in 2018).

Table 6-3. Household WEEE chain revenues from 2015 to 2018 (in k€).

| **Revenues** | **2015** | **2016** | **2017** | **2018** |
|---|---|---|---|---|
| Visible fee | 168,950 | 188,795 | 269,024 | 304,779 |
| Fractions selling | 81,071 | 72,490 | 92,579 | 101,227 |
| Products | 3,365 | 2,253 | 1,441 | 1,238 |
| Others | 59 | 143 | 18 | 42 |
| Total | 253,445 | 263,681 | 363,062 | 407,286 |

Sources: based on data from Fangeat, 2019, 2018, 2017, 2016.

In turn, Table 6-4 presents the costs reported by the compliance schemes from 2016 to 2018 for household WEEE. In France, compliance schemes are not-for-profit organizations. Thus, provisions are the difference between the total expenses (Table 6-4) and the revenues (Table 6-3). In 2018, the total expenses excluding provisions increased by 18% in comparison to 2015.

It can be noticed that, before 2017, the compliance schemes' income was lower than the total expenses. From 2017, this scenario has changed, resulting in positive provisions. The change in the visible fee modulation, and the limited collection by the official schemes (collection rate was 54% in 2018) may contribute to high revenues. In 2016, the taxes were a negative value potentially due to a tax credit (e.g. research tax credit).

Table 6-4. Household WEEE chain expenses from 2015 to 2018 (in k€).

| Expenses | 2015 | 2016 | 2017 | 2018 |
|---|---|---|---|---|
| Operational costs | 224,355 | 252,644 | 253,082 | 269,834 |
| Net operational costs | 143,284 | 180,154 | 160,503 | 168,607 |
| Support to collection actors (except communication) | 41,209 | 45,268 | 44,374 | 46,253 |
| Communication | 11,790 | 13,190 | 13,323 | 13,893 |
| R&D | 2,174 | 2,279 | 1,455 | 2,791 |
| Provisions | -39,349 | -61,619 | 37,078 | 61,723 |
| Operational fees | 11,766 | 12,030 | 12,097 | 10,677 |
| Taxes | 1,500 | -111 | 1,653 | 2,115 |
| Total expenses | 253,445 | 263,681 | 363,062 | 407,286 |
| Total expenses excluding provisions | 292,794 | 325,300 | 325 984 | 345,563 |

Sources: based on data from Fangeat, 2019, 2018, 2017, 2016.

## 6.3 Expansion of the current economic indicators

In this study, we define an indicator as a value representing the performance of a system or organization. It gives an idea of its state or level of compliance, and is ideally assessed by comparing the result with an agreed standard or target. According to Bell and Morse (2013), indicators are a means of capturing complexity into easily digestible bits of information. It may help non-technical minded specialists to make use of complex datasets.

The current economic indicators reported by the compliance schemes and ADEME are, in fact, data on incomes and expenses of the official schemes. They do not allow an understanding of the economic performance of WEEE systems. In order to interpret them, it is necessary to analyze other data from the performance of the chain (e.g. collection cost per tons of WEEE collected by the official schemes).

In this context, we searched indicators in the literature that could improve the current assessment of the economic performance of e-waste schemes. Table 6-5 summarizes the indicators selected and/or developed in this thesis (listed in the last column) organized by group of indicators (cost-effectiveness and economic benefits of the WEEE schemes). The intermediate columns present the indicators selected from the literature review (with their references) that support the development of the indicators suggested in this thesis.

Table 6-5. Indicators used as reference for the set of economic indicators proposed in this study.

| Set of economic indicators | Indicators currently used or identified in the literature review | | | Indicators suggested in this work |
|---|---|---|---|---|
| | **Reference** | **Name of the indicator** | **Main aspects identified** | |
| Cost effectiveness of WEEE schemes | Monier et al., 2014 | Fees per EEE collected | Ratio between the fees and WEEE collected | Visible fee per WEEE collected (section 6.3.1.1) |
| | Monier et al., 2014 | Fees per inhabitant per year | Ratio between the fees and the number of inhabitants | Visible fee per inhabitant (section 6.3.1.2) |
| | Rigamonti et al. 2016 | Costs indicator | Total costs (collection, treatment and final disposal) per ton of waste | Total costs per WEEE collected (section 6.3.1.3) |
| | Magalini and Huisman, 2018 | Costs for compliant and non-compliant recycling of WEEE product categories (€/million tons) | Total costs and per type (compliance, depollution, disposal of hazardous and non-hazardous substances and treatment) for each WEEE category | Total costs per WEEE treated (section 6.3.1.4)<br>Total costs per inhabitant (section 6.3.1.5) |
| Economic benefits of WEEE schemes | Cucchiella et al., 2015 | Recovery economic potential (€/kg of product and €/pieces) | Calculated as the product of the materials in WEEE and their market prices | Economic weighted-based collection rate (section 6.3.2.2) |
| | De Oliveira Neto et al. (2017) | Economic advantage index (EAI) | Quantifies the amount of dollars (US$) saved per material recycled from WEEE | Economic weighted-based recycling rate (section 6.3.2.2) |
| | Magalini and Huisman, 2018 | Losses (in euro and kilotons per year) related to scavenging of components for waste streams | Losses per category of WEEE based on WEEE generated approach | Economic value of the complementary flows (section 6.3.2.3) |
| | Magalini and Huisman, 2018 | Value (in millions €) of scavenged components in 2016 | Losses per type of component in all WEEE categories | |

The first two indicators are fully adopted from the literature ("visible fee per WEEE collected" and the "visible fee per inhabitant"), but the others are proposals based on improvements of indicators identified in the literature. For example, regarding the costs of WEEE schemes, our assessment quantifies the costs per ton of WEEE collected, treated and by the number of inhabitants.

Regarding the set of indicators grouped in the "economic benefits of the WEEE schemes", Cucchiella et al. (2015) considered the price of materials in WEEE composition in order to estimate the economic potential of the end-of-life of different types of WEEE products. Following this approach, we suggest calculating the economic weighted-based collection and recycling rate. Our scope and method of calculation is different, although based on the same approach (for more details see sections 6.3.2.1 and 6.3.2.2). The last indicator (economic value of the complementary flows) includes in addition to the components scavenging suggested by Magalini and Huisman, 2018, the economic potential dissipated in other complementary flows.

### 6.3.1. Cost-effectiveness of WEEE schemes

#### 6.3.1.1. Visible fee per EEE POM

As previously presented, the visible fee is the main source of revenue from the WEEE chain (75% in 2018) and is the only economic data publicly available in ADEME reports. However, this information is difficult to interpret without a comparison to data of the WEEE chain technical performance. We suggest assessing the ratio between the amount raised (in euros) from the visible fee and the number of items of equipment POM (Equation 6-1).

$$FC_f = \frac{\sum_{i=1}^{n1} VF_i}{\sum_{i=1}^{n1} WM_i}$$

$FC_f$ visible fee per EEE POM per category (€/tons or €/kg);

$n1$ number of different UNU-Keys related to the WEEE category;

$VF$ compliance schemes' revenues with the visible fee, in euros;

$WM$ amount of EEE placed on the market, in tons or kilos.

Equation 6-1. Visible fee per EEE POM per category.

For the consumers (EEE buyers/customers), this indicator can clarify the average amount of visible fee collected (in euros) per average weight of equipment placed on the market (in tons or kg). As discussed in section 6.2.1, the fee is modeled per type of equipment, but it may be useful to the consumers to have a rough overall view per WEEE category. This information, together with other indicators proposed in this study, can highlight the importance of the visible fee.

#### 6.3.1.2. Visible fee per inhabitant

Complementary to the previous indicator, the ratio between the visible fee and the population in a given year (Equation 6-2) can support consumer awareness of its contribution to the WEEE chain.

$$FI_f = \frac{\sum_{i=1}^{n1} VF_i}{N_{inh}}$$

$FI_f$ visible fee per inhabitant per category in a given year (€/inh/year);

$n1$ number of different UNU-Keys related to the WEEE category;

$VF$ compliance schemes' revenues with the visible fee, in euros;

$N_{inh}$ population of the country in a given year.

Equation 6-2. Visible fee per inhabitant per category.

#### 6.3.1.3. Total costs per WEEE collected

The total costs of the WEEE chain correspond to the expenses of the compliance schemes until the end-of-waste status is achieved for the recoverable fractions or disposal of non-recoverable fractions. It includes the expenses of the collection and treatment operators, as well as other costs related to the compliance schemes operation, as detailed in In turn, Table 6-4 presents the costs reported by the compliance schemes from 2016 to 2018 for household WEEE. In France, compliance schemes are not-for-profit organizations. Thus, provisions are the difference between the total expenses (Table 6-4) and the revenues (Table 6-3). In 2018, the total expenses excluding provisions increased by 18% in comparison to 2015.

It can be noticed that, before 2017, the compliance schemes' income was lower than the total expenses. From 2017, this scenario has changed, resulting in positive provisions. The change in the visible fee modulation, and the limited collection by the official schemes (collection rate was 54% in 2018) may contribute to high revenues. In 2016, the taxes were a negative value potentially due to a tax credit (e.g. research tax credit).

The ratio between the total costs and the weight of EEE collected (Equation 6-3) allows an estimation of the costs per WEEE collected in a given year by the official schemes. Monitoring the evolution of the total expenses (excluding the provisions) over the years can give interesting information on the evolution of the costs of the e-waste chain.

$$CC_f = \frac{\sum_{i=1}^{n1} TC_i}{\sum_{i=1}^{n1} WC_i}$$

$CC_f$ total costs per WEEE collected by the official schemes per category (€/tons or €/kg);

$n1$ number of different UNU-Keys related to the WEEE category;

$TC$ total expenses of the official schemes, in euros;

$WC$ weight of e-waste collected by the official schemes, in tons or kilos.

Equation 6-3. Total costs per WEEE collected by the official schemes per category.

#### 6.3.1.4. Total costs per WEEE treated

Considering that there is a gap between the quantities collected and treated in the same year, in addition to the previous indicator, we suggest assessing the ratio between the total expenses and the amount treated in a given year.

$$CT_f = \frac{\sum_{i=1}^{n1} TC_i}{\sum_{i=1}^{n1} WT_i}$$

$CT_f$ total costs per WEEE treated by the official schemes per category (€/tons or €/kg);

$n1$ number of different UNU-Keys related to the WEEE category;

$TC$ total expenses of the official schemes, in euros;

$WT$ weight of e-waste that is sent for the different types of treatment (recycling, reuse, energy recovery or disposal), in tons or kilos.

Equation 6-4. Total costs per WEEE treated by the official schemes per category.

The difference between this indicator and the previous one may not be significant, but it is important to monitor the evolution between the collection, treatment, and total costs of the official schemes.

#### 6.3.1.5. Total costs per inhabitant

In addition to the indicator 'visible fee per inhabitant' (Equation 6-2), we suggest monitoring the ratio between the total costs and the population in a given year (Equation 6-5). The analysis of these two indicators in parallel can support consumer awareness about the costs of treating e-waste and the importance of the visible fee for the e-waste chain.

$$CI_f = \frac{\sum_{i=1}^{n1} TC_i}{N_{inh}}$$

$CI_f$ total costs of the WEEE chain per inhabitant per category (€/inh).

$n1$ number of different UNU-Keys related to the WEEE category;

$TC$ total expenses of the official schemes, in euros;

$N_{inh}$ population of the country in a given year.

Equation 6-5. Visible fee per inhabitant per WEEE category.

We believe that the comparison between the two indicators could be interesting for the compliance schemes and the policy makers as well. Since the visible fee is the main source of funding, and it is modeled to account for the real costs of treatment, the results of both indicators should be similar.

### 6.3.2. Economic benefits of WEEE schemes

#### 6.3.2.1. Economic weighted-based collection rate

As pointed out in **Chapter 1**, economic benefits are one of the main drivers for e-waste treatment. Aiming to highlight the importance of improving collection to maximize profits, we suggest adopting economic weighted-based collection rate indicator (Equation 6-6). This indicator measures how differences in the types of e-waste generated and collected by the official systems can affect, from an economic perspective, the potential economic benefits of element recovery. It should be applied to measure two or more target elements.

The calculation is similar to the collection rate per target element (Equation 4-3) and includes the market price of each target material in the given year.

$$ECR_f = \frac{\sum_{l=1}^{n4} WC_{el} \times MP_{el}}{\sum_{l=1}^{n4} WG_{el} \times MP_{el}}$$

$ECR_f$ economic weighted-based collection rate per WEEE category;

$n4$ number of target elements considered in the assessment;

$WC_e$ weight of target element collected by the official schemes (calculated based on Equation 4-3 detailed in **Chapter 4**);

$MP_e$ market price of the target element;

$WG_e$ weight of target element generated (calculated based on Equation 4-3 detailed in **Chapter 4**).

Equation 6-6. Economic weighted-based collection rate for target elements in a WEEE category.

#### 6.3.2.2. Economic weighted-based recycling rate

This indicator aims to access the recycling rate considering the market price of target elements (Equation 6-7). The scope of the assessment is from the collection by the official schemes to the recycling of the elements. It aims to provide insights on the profitability of the recycling by the official schemes. It should be applied to measure two or more target elements.

The calculation is similar to the recycling rate per target element (Equation 4-9) and includes the market price of each target material in the given year.

$$ERR_f = \frac{\sum_{l=1}^{n4} WR_{el} \times MP_{el}}{\sum_{l=1}^{n4} WG_{el} \times MP_{el}}$$

| | |
|---|---|
| $ERR_f$ | economic weighted-based recycling rate per WEEE category; |
| $n4$ | number of target elements considered in the assessment; |
| $WR_e$ | weight of target element recycled by the official schemes (calculated based on Equation 4-9Equation 4-3 detailed in **Chapter 4**); |
| $MP_e$ | market price of the target element; |
| $WG_e$ | weight of target element generated (calculated based on Equation 4-3 detailed in **Chapter 4**). |

Equation 6-7. Economic weighted-based recycling rate for target element in a WEEE category.

#### 6.3.2.3. Economic value of the complementary flows

This indicator aims to quantify the potential value of generated e-waste that is dissipated in the complementary flows, including component scavenging. As described in Equation 6-8, the market price is included in the ratio of the target elements dissipated in complementary flows and the target elements generated.

$$ECF_f = \frac{\sum_{l=1}^{n4} (WG_{el} - WC_{el}) \times MP_{el}}{\sum_{l=1}^{n4} WG_{el} \times MP_{el}}$$

| | |
|---|---|
| $ECF_f$ | economic value of the complementary flows per WEEE category; |
| $n4$ | number of target elements considered in the assessment; |
| $WG_e$ | weight of target element generated (calculated based on Equation 4-3 detailed in **Chapter 4**); |
| $WC_e$ | weight of target element collected by the official schemes (calculated based on Equation 4-3 detailed in **Chapter 4**). |
| $MP_e$ | market price of the target element; |

Equation 6-8. Economic value of the complementary flows per WEEE category.

## 6.4 Caste Study: economic assessment of WEEE schemes in France

### 6.4.1. Overall expenses of WEEE treatment

As described in section 6.3.1, we suggest calculating the cost-effectiveness per WEEE category. Nevertheless, the data reported nowadays by the compliance schemes does not allow an assessment per category. Currently, only data on the overall costs and revenues of all e-waste categories is available, as presented in Table 6-3 and Table 6-4. Consequently, due to a lack of data, we could not validate the cost-effectiveness indicators focused on WEEE category II. The following paragraphs present the results of the cost-effectiveness indicators for all WEEE categories.

Figure 6-1 presents the total amount collected with the visible fees (current indicator represented in the secondary axis), and the indicator "visible fee per EEE POM" from 2015 to 2018 (primary axis).

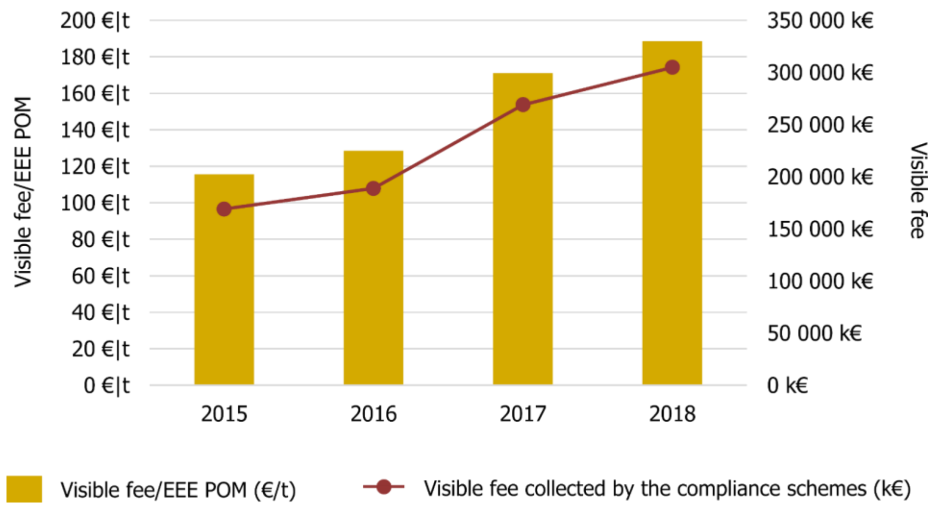



Figure 6-1. Visible fee and EEE placed on the market.

The amount of fees collected from 2015 and 2018 increased by 80% (from 168,950 k€ to 304,779 k€). With the "visible fee per EEE POM" indicator we could identify that there was a growth of 33% in the fees collected per ton of EEE POM from 2016 to 2017. In the same period there was also a rise in EEE POM (11%), and a new modulation criterion was adopted by the compliance schemes (detailed in section 6.2.1).For communication to the EEE consumers, we recommend using the unit €/kg to facilitate the comprehension based on the weight of individual devices. For example: *"in 2018, to support e-waste collection and treatment, 0.19 € were collected per kilogram of EEE placed on the market"*.

The indicator "visible fee per inhabitant", is complementary to the previous one. As presented in Figure 6-2, the contribution per inhabitant to e-waste treatment has increased in recent years. In 2015 it was 2.54 €/inh, and in 2018 it rose to 4.54 €/inhabitant (an increase of 44%). This is a consequence of population increase (in average 0.5%/year), and, as previously discussed, of consumption behavior and changes in the modulation criteria. The fee per inhabitant rose by 10% from 2015 to 2016 and from 2017 to 2018. The growth from 2016 to 2017 was higher (30%), mostly due to the change in the modulation criteria.

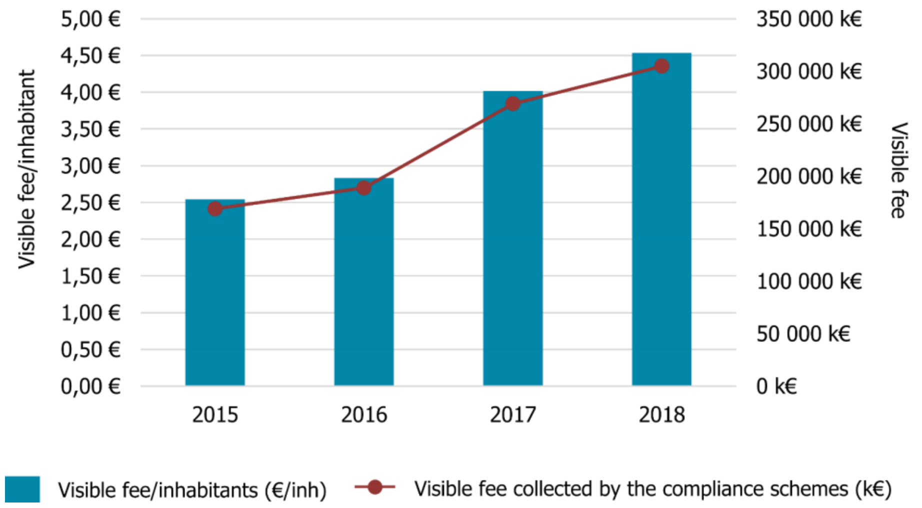



Figure 6-2. Visible fee and population in France.

Figure 6-3 presents the results for the indicator "total cost per WEEE collected" (in €/tons presented in the primary axis) and the information currently monitored by ADEME – total costs excluding provisions (secondary axis). As can be seen, the monitoring of the annual costs could be wrongly interpreted if the amount of e-waste collected is not taken into account.

From 2015 to 2018, the total costs increased by 18%. However, in the same period, the total e-waste collected by the official schemes increased by 26%. Consequently, even if there was an increase in the total expenses, the cost per ton of e-waste collected has decreased. The annual decrease was 4% from 2016 to 2015, and from 2017 to 2016. In 2018 there was a slight increase in the average cost in comparison to 2017 (less than 1%).

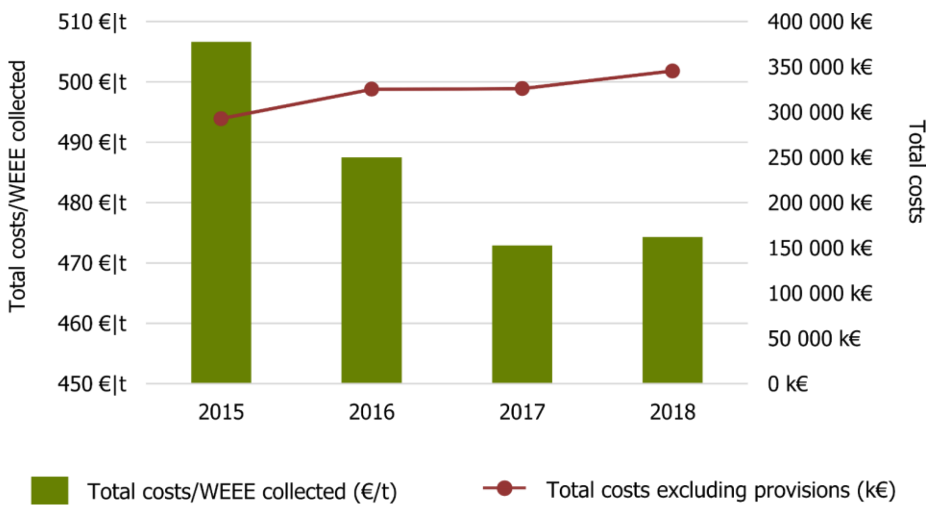



Figure 6-3. Total costs and WEEE collection by the official schemes.

Figure 6-4 presents the total costs per e-waste treated in a given year. This indicator is complementary to the previous one and considering that there is often a gap between the e-waste collected and treated in a given year, it is interesting to monitor both values.

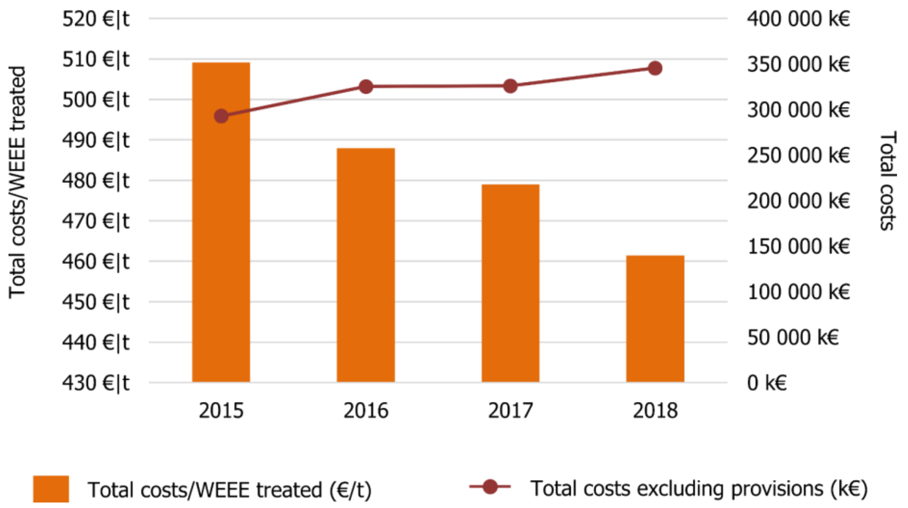



Figure 6-4. Total costs and WEEE treated by the official schemes.

For example, in 2018, 748,952 t of e-waste were treated, but only 728,569 t of e-waste were collected in the same year. Thus, about 20,000 of the e-waste treated had been collected in the previous year. In turn, in 2017 the amount collected was higher than the quantities treated (by about 9,000 t). Therefore, in 2018, the "total costs per WEEE collected" was slightly higher than in 2017 (472.90 € in comparison to 474.30 €), and the "total costs per WEEE treated" was lower (4,789.4 € and 461.40 € respectively).

As presented in Figure 6-5, the indicator "total costs per inhabitants" (calculated with Equation 6-5), is complementary to the indicator "visible fees per inhabitant" (calculated with Equation 6-2). It can be used to communicate to the consumers the extent to which the visible fee contributes to paying the total costs of the WEEE schemes. For example, *"the total cost of WEEE collection and treatment in 2018 was 5.14 €/inhabitant, of which the visible fee financed 4.54€/inhabitant or 88%".*

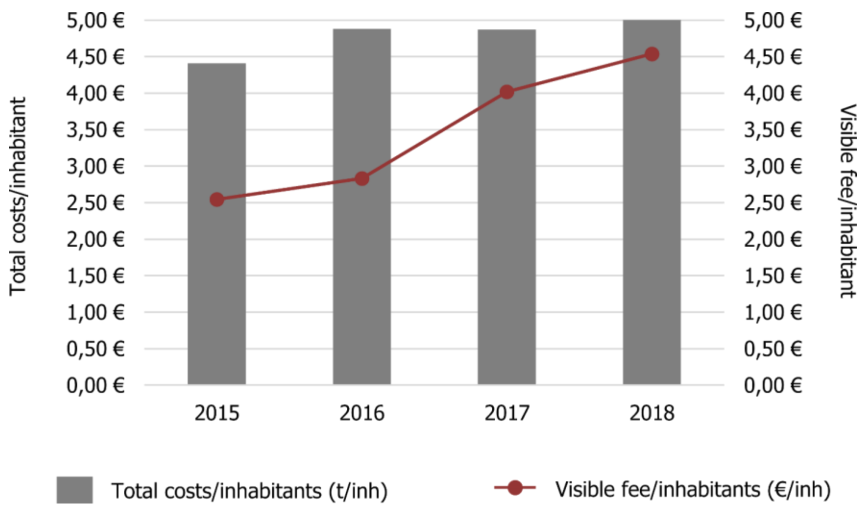
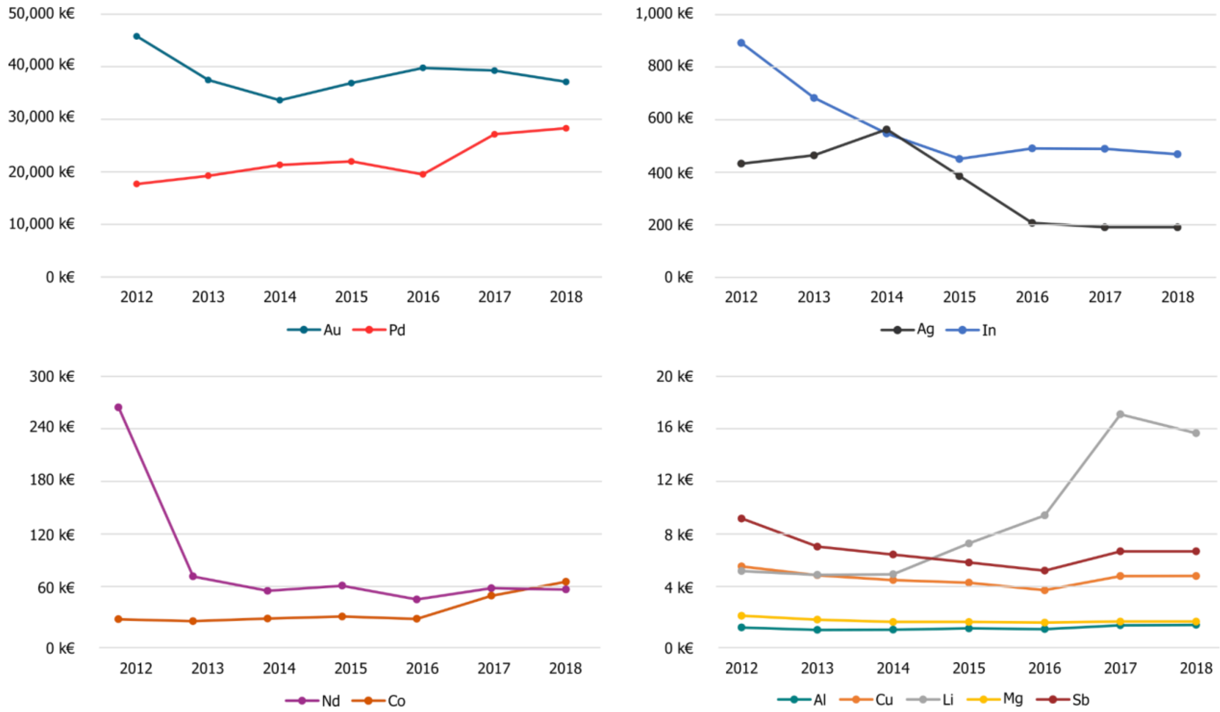



Figure 6-5. Total costs and fees collected per inhabitant.

### 6.4.2. Economic benefits of screens treatment

The screens category is selected as a case study, and the same treatment scenarios presented in **Chapter 4** are considered for the assessment of the economic benefits. Figure 6-6 presents the market price of the elements targeted in the case study from 2012 to 2018. It can be noticed, for certain materials, that there is high price volatility.




Ag, Au, Al, Cu, Li and Co: in 2018, the price is based on average prices from January to August.
In, Mg, Pd and Sb : due to lack of data, in 2018, we considered the same values as 2017.

Figure 6-6. Market price of the elements targeted in the study (k€/t).

Source: based on data from BRGM, 2018; Kitco, 2019.

Figure 6-7 presents the indicator "economic weighted-based collection rate", as well as the weight-based collection rate indicator (described in **Chapter 4**) for the targeted elements. Since WEEE batteries are treated by a different EPR chain, we present the results for the WEEE and batteries separately.

For the WEEE without batteries, when the price of the target element is considered in the collection rate assessment, the results are lower than the weight-based collection rate. For example, in 2018, the economic weighted-based collection rate was 23%, while the weight-based was 27%. This result is due to the low collection of certain target elements having a high market value, such as gold and palladium, which are present in high quantities in FPD displays. For the batteries, the impact of the economic weighted-based approach is lower because the target elements present in batteries have lower market values (e.g. Cu and Co).

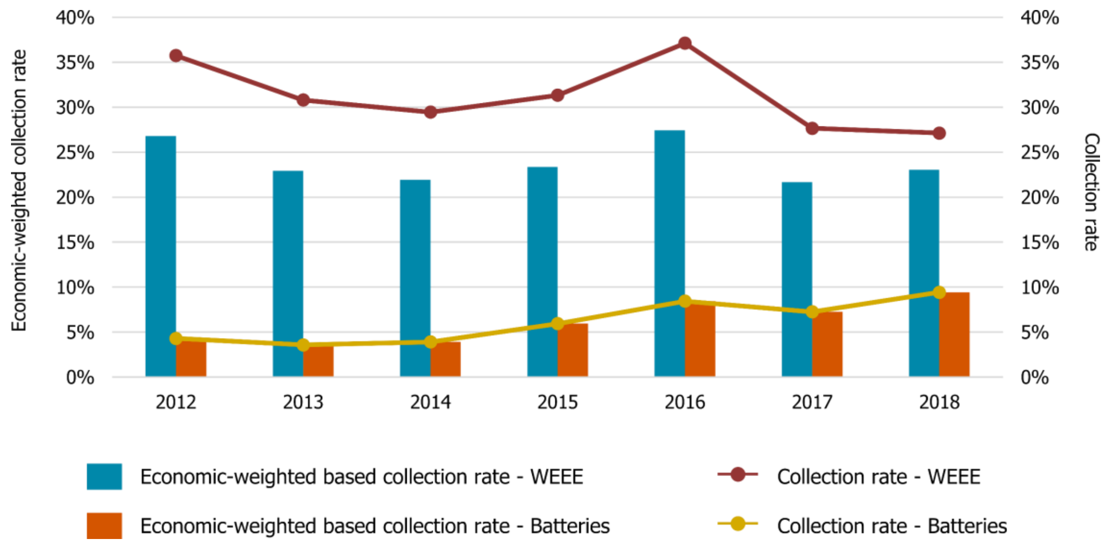



Figure 6-7. Economic weighted-based collection rate.

The results of the indicator "economic weighted-based recycling rate" are presented in Figure 6-8, together with the results of the weight-based recycling rate (described in **Chapter 4**). Unlike the previous indicator, the scope of the indicator is from the e-waste collected to the elements recycled. Thus, the amount not captured by the compliance schemes is not considered.

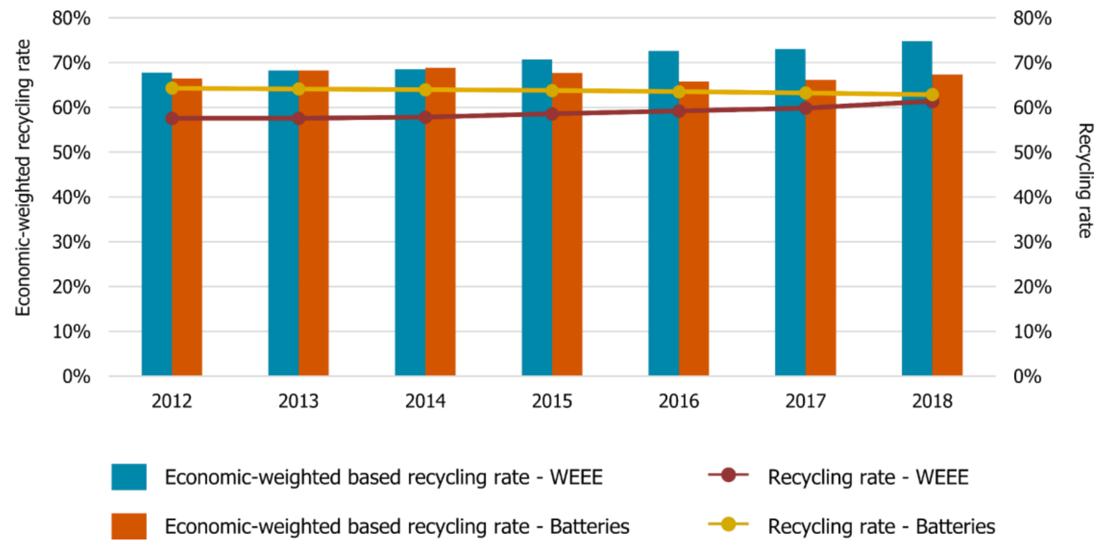



Figure 6-8. Economic weighted-based recycling rate.

The results of the "economic weighted-based recycling rate" for the target elements is higher than the weight-based recycling rate, again this is because of the high market value of certain materials. For example, Figure 6-9 and Figure 6-10 present the results of the weight-based and economic-weighted recycling rate for gold and copper. It also presents the elements share in terms of mass, and mass multiplied by the market value.

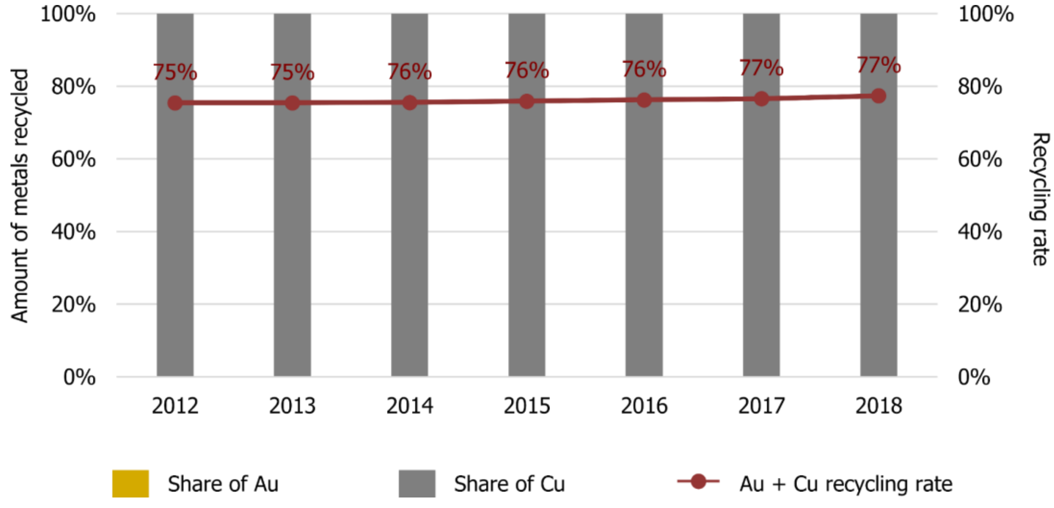



Figure 6-9. Gold and copper: weight-based recycling rate.

From a weight-based approach, copper accounts for 99% of the total mass. Nevertheless, when the price is considered, gold accounts for 63% of the share. The high market value together with an efficient performance of the final recycling operation, contribute to a high "economic-weighted recycling rate". This conclusion reinforces the results of **Chapter 4** that for some materials the most significant (economic and in terms of weight) losses are due to low collection.

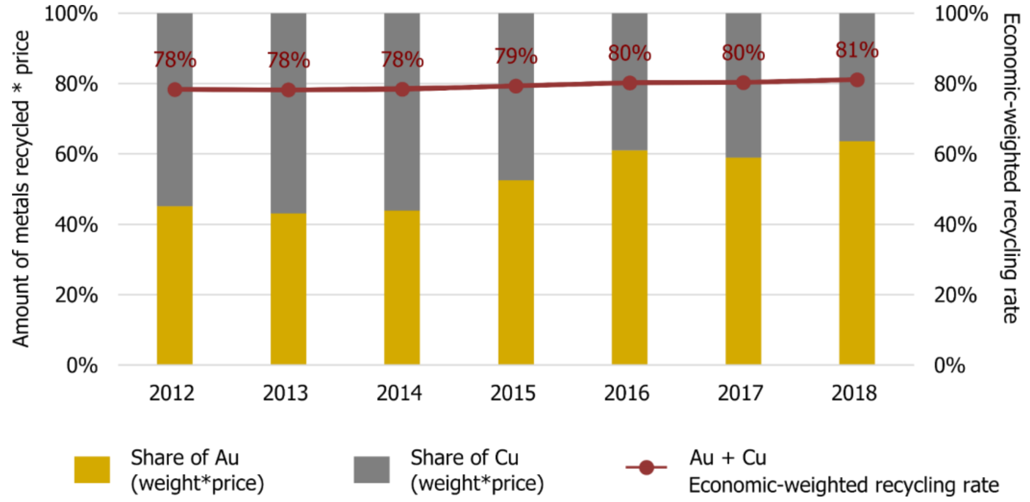



Figure 6-10. Gold and copper: economic-weighted recycling rate.

Lastly, Figure 6-11 presents the results for the final economic indicator: economic value of the complementary flows. This indicator aims to highlight the economic potential of the flows not captured by the compliance schemes.

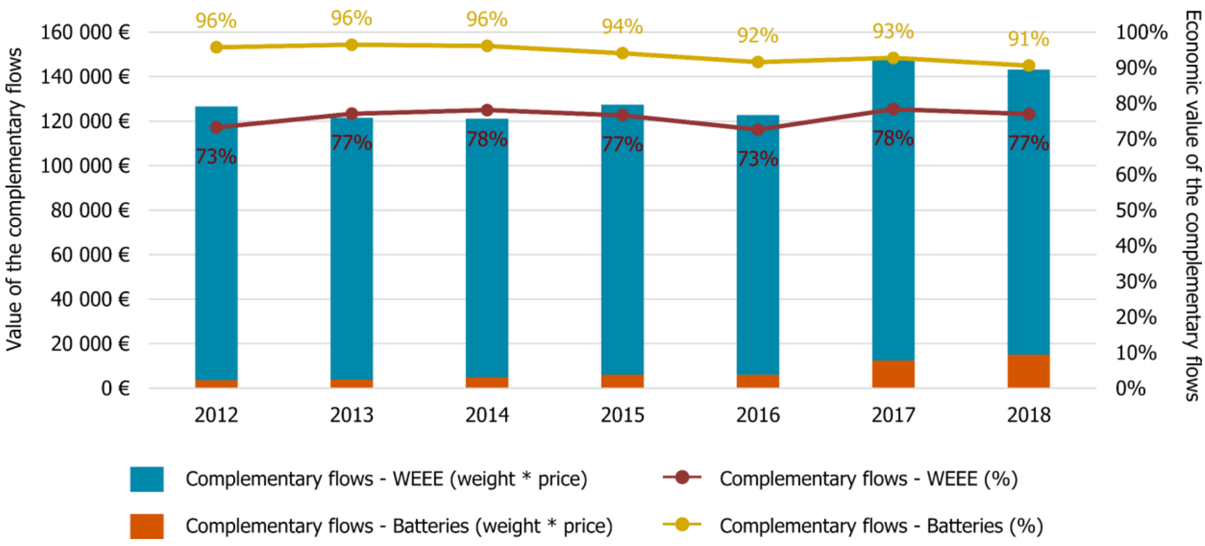



Figure 6-11. Economic value of the complementary flows.

In 2018, for the WEEE without batteries, 77% of the economic potential of the target elements was dissipated in complementary flows. The results were quite stable in the preceding years (from 73% to 77% depending on the amount collected and its composition). For the batteries the result is even higher: in 2018, 91% of the economic potential of the target elements was not collected by the official schemes. With the progressive increase of batteries collected, this result is expected to reduce.

## 6.5 Conclusions

This chapter presents eight economic indicators to improve monitoring of the WEEE chain, as well as the methods and data required. The first set of indicators (cost-effectiveness) intend to improve the monitoring and reporting of revenues and expenses. As presented in section 6.4, they can be used to improve the awareness of consumers regarding the costs of the WEEE chain and the importance of the visible fee.

The indicators regrouped in the second set of indicators (economic benefits) present a lifeline to the current weight-based approach, where the only materials targeted are those which are highly concentrated in terms of weight. This approach can be used for decision making, for example, to prioritize the recycling of materials considering not only their weight, but also their market value, and to define different types of targets. As discussed in the previous chapters, it is necessary to identify other metrics than the current weight-based approach to highlight the importance of collecting and recycling those elements present in smaller quantities.

Table 6-6 summarizes the indicators, as well as the sections in which the indicators are validated. This set of indicators aims to reply to the third research question of this thesis presented in **Chapter 1** regarding the metrics for evaluating the economic impacts and benefits of the WEEE chain.

Table 6-6. Summary of the economic indicators proposed in Chapter 6.

| | Economic indicators | Equations | Validation with case study |
|---|---|---|---|
| 1. | Visible fee per EEE placed on the market | Equation 6-1 | Section 6.3.1.1 |
| 2. | Visible fee per inhabitant | Equation 6-2 | Section 6.3.1.2 |
| 3. | Total costs per WEEE collected | Equation 6-3 | Section 6.3.1.3 |
| 4. | Total costs per WEEE treated | Equation 6-4 | Section 6.3.1.4 |
| 5. | Total costs per inhabitant | Equation 6-5 | Section 6.3.1.5 |
| 6. | Economic weighted-based collection rate | Equation 6-6 | Section 6.3.2.1 |
| 7. | Economic weighted-based recycling rate | Equation 6-7 | Section 6.3.2.2 |
| 8. | Economic value of the complementary flows | Equation 6-8 | Section 6.3.2.3 |

The feasibility of the indicators presented in this chapter relies on the availability of data to calculate them. The cost-effectiveness indicators (indicators 1 to 5) could not be assessed per category because, nowadays, the compliance schemes do not share with ADEME the revenues and cost data per category. As mentioned previously, the compliance schemes do not share all financial data because they are in competition with one another, although in fact, they monitor costs with a higher degree of detail than they report.

The general goal of cost-effectiveness indicators is to improve visibility of the costs and revenues of the WEEE chain for the different actors. Thus, we believe that these indicators could be adopted in the next national specifications included in the Ministerial Orders for household and professional WEEE. In order to not disclose internal data of the compliance schemes, we suggest reporting the average results of the compliance schemes, and not the individual financial performance of these organizations.

Regarding the economic benefit indicators (indicators 6 to 8), they rely on more precise data related to waste flow monitoring, composition, and treatment performance. The difficulties and strategies for obtaining this data are discussed in **Chapter 4**. Additionally, data on the market prices of materials and elements is also required. For that data, the difficulties are lower because there are a variety sources (NB most of them are not free of charge) that provide this type of data.

Despite the eventual difficulties in data acquisition to put the indicators into practice, we believe that their results can help to boost e-waste collection and treatment. As discussed previously, the economic benefits are one of the main motivations of e-waste treatment. Thus, quantifying the potential economic benefits can support the engagement of the different actors in the chain.

The economic weighted-based based indicators are an alternative approach for the weight-based indicators. By adding economic value to the current performance assessment scope, the interest in collection and recycling elements found in small quantities, but which have high market value, is highlighted. This can be an interesting input for future performance targets based on approaches other than overall weight.

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# Chapter 7 | Criticality performance of WEEE chain

*"Não podemos deixar de ser doutos. Doutores.*
*País de dores anônimas, de doutores anônimos.*
*O Império foi assim. Eruditamos tudo.*
*Esquecemos o gavião de penacho.*[19] *"*
Oswald de Andrade, Manifesto da Poesia Pau-Brasil (1924).

## 7.1. Criticality: a technical, economic or environmental assessment?

Following the approach of the previous chapters, in **Chapter 7,** we present another dimension of Waste Electrical and Electronic Equipment (WEEE) performance assessment as a lifeline to the current weight-based approach. Given the potential of WEEE as a supplier of secondary raw materials, we explore the combination of criticality assessment and metrics to evaluate the recovery efficiency of materials. Critical raw materials (CRMs) describe elements that, at a given time and within a certain economic system, are necessary to achieve performances and functions in a technology or product, and therefore their supply is crucial (Hofmann et al., 2018).

Hence, criticality assessment goes beyond the availability of mineral deposits and reserves (currently explored, and/or identified). It considers geopolitical, economic, and technical aspects related to the materials supply (Graedel et al., 2014). Although the adverse impacts of an eventual material disruption are analyzed from an anthropocentric dimension, economic downturns impact the affected population. In this study, we classify criticality as an environmental indicator because it aims to capture the potential of the WEEE chain as a supplier of CRM in the context of a circular economy.

Before presenting the new indicators (section 7.5), we introduce some aspects relevant to the understanding of the final calculation and the interest of these metrics:

- Criticality assessment (section 7.2): a brief overview of the current criticality assessment methodologies is presented with a focus on the European Commission (EC) methodology used by the indicators suggested in this thesis (section 7.2.1);
- CRM and WEEE (section 7.3): we discuss the importance of urban mining and to what extent WEEE recycling can mitigate supply disruption;
- Criticality as a parameter to assess WEEE recovery (section 7.4): we describe three studies that inspired our proposition, and the opportunities for improvement we identified leading to the metrics proposed in this thesis.

Lastly, as in the previous chapters, the proposed indicators are validated in a case study (section 7.6), followed by a conclusion (section 7.7).

[19] We can't help being erudite. Doctor of Philosophy. Country of anonymous ills, of anonymous doctors. The Empire was like that. We made everything erudite. We forgot the ornate hawk-eagle (a fairly large bird of prey from the tropical Americas).

## 7.2. Criticality assessment

The term *criticality* has been in use since the 40s, but in recent years, it has become a topical issue in different stakeholders' debates (politicians, industry, academy, etc.). Consequently, several definitions and metrics to assess material criticality have been published in the literature. Regardless of the definition and assessment metrics, two aspects are widely regarded as crucial for the identification of CRMs: supply risk potential (likelihood) and the vulnerability (consequence) of a system to a potential supply shortfall (Dewulf et al., 2016; Frenzel et al., 2017).

The first criticality evaluation methodology was released by the U.S. National Research Council (NRC, 2008). This study introduced a criticality matrix approach that was used as a reference for many subsequent studies. As presented in Figure 7-1, the matrix is based on two dimensions: in the vertical axis, the impact of supply restriction (vulnerability to disruption), and in the horizontal axis the material's availability (risk or likelihood) (Jin et al., 2016).

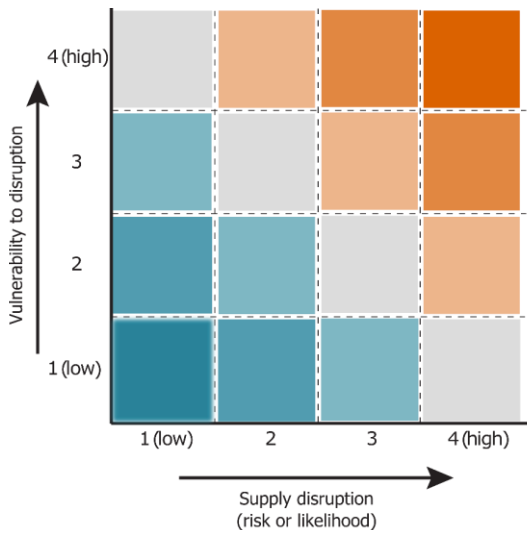


Figure 7-1. Criticality matrix approach.
Source: adapted from NRC, 2008.

The criticality matrix is a powerful tool for identifying and communicating economic vulnerabilities due to insecure raw material supply, and often leads to a ranking of the materials along the scale (Glöser et al., 2015). The scope of the vertical axis varies depending on the methodology considered and may be applicable to a region (e.g. EC methodology detailed in 7.2.1), a country (BRGM, 2014; Morley and Eatherley, 2008), or a company.

The most common procedure for criticality assessments is to compile sets of different indicators into aggregate scores for both the likelihood of supply disruptions and the consequence of such disruptions (economic importance). Then, the results are plotted against each other to delimit the field of critical raw materials (Frenzel et al., 2017; Løvik et al., 2018). Aggregation of the criticality axes into one single criticality indicator is rarely done (Dewulf et al., 2016).

Some studies include the environmental aspect in the criticality assessment. For example, Graedel et al. (2012) published a methodology comprised of three dimensions: supply risk, vulnerability to supply restriction and environmental implications. The environmental dimension is calculated based on the results of Life Cycle Assessment (LCA) calculated according to the ReCiPe end point method, with "world" normalization and "hierarchist" weighting based on Ecoinvent database (see **Chapter 5** for more information on LCA). According to Sonnemann et al. (2015), the information gathered in LCA databases, methods and studies allows contributes to the criticality assessment of resources, water and land. Nevertheless, the authors highlight the limitation of the current LCIA methodologies in not capturing socio-economic and geopolitical issues related to natural resources that are relevant for sustainability assessment.

A survey performed by Blengini et al. (2017a) identified more than 200 sources and organizations dealing with criticality assessments. According to the same study, within the EU, most of organizations involved in criticality studies adopt the methodology and CRM list developed by the EC.

The first version of the EC methodology for criticality assessment was published in 2010, and the first list of CRMs was made publicly available in 2011 (European Commission, 2010). Both methodology and list were updated in 2014 and 2017, and the EC is continuously working to keep it up to date at least every three years (European Commission, 2019). Considering that, in this study, we propose criticality-weighted based indicators to be used in the context of EU WEEE chain (see section 7.5), in this study we selected the EC methodology.

As discussed by some authors in the literature, criticality assessment methodologies depend upon subjective judgements and contain uncertainties. For example, in the choice and weighting of indicators to quantify the supply risk and the economic importance. Consequently, it is inevitable that different studies will arrive at different results (Lloyd et al., 2012).

### 7.2.1. European Commission methodology

EU Raw Materials Initiative (RMI) launched by the EC in 2008 has as its pillars ensuring sustainable supply of raw materials and boosting resource efficiency and recycling (Vidal-Legaz et al., 2018). The main motivation behind the RMI is to secure access to enough raw materials for the European economy and reduce reliance on imports (De Meester et al., 2019).

As part of the EU policy and strategy on raw materials, as previously mentioned, the EC decided to establish a criticality assessment methodology and a list of CRMs for Europe (Løvik et al., 2018). A material is considered as critical for the European Union (EU) if it has high economic importance and its supply is vulnerable to disruption (Blengini et al., 2017b).

It is important to remark that the criticality assessment is an analysis of the current situation based on the recent past (5 years) and does not include the future response of the systems due to an inadequate supply of a given material (resilience). The following subsections briefly describe the economic importance and supply risk calculations, and the 2017 CRM list based on the EC methodology (Blengini et al., 2017a, 2017b; Deloitte Sustainability et al., 2017a, 2017b, 2017c).

#### 7.2.2.1. Economic importance

High economic importance (EI) means that the material is of fundamental importance to industry sectors that create added value and jobs. The importance of a material to the EU economy is measured in terms of its end-use applications, and the value added (VA) by the corresponding sectors at the NACE Rev.2 (2 digital level)[20].

The economic importance is calculated with Equation 7-1. As can be seen, the EI is corrected by the substitution index ($SI_{EI}$). It is based on a matrix that compares the given material to a potential substitute in terms of technical performance and cost for each end-use application. The EC methodology considers only proven substitutes that are readily available for use. In this approach, a maximum 30% reduction of EI is assumed if all substitute materials offer similar performance and cost – an optimal scenario very improbable to happen.

$$EI = \sum\nolimits_S (A_S \times Q_S) \times SI_{EI}$$

| | |
|---|---|
| $EI$ | economic importance; |
| $S$ | denotes sector; |
| $A_S$ | share of end use of a raw material in a NACE Rev. 2 2-digit level sector; |
| $Q_S$ | NACE Rev. 2 2-digit level sector's value added; |
| $SI_{EI}$ | substitution index related to economic importance. |

Equation 7-1. Economic importance according to the EC methodology.

The two improvements proposed by the latest EC methodology update were: (1) a more detailed and transparent allocation of raw material uses to their corresponding NACE sectors and (2) the use of a raw material-specific substitution index. The economic importance is assumed to be proportional to the size of the economic sectors in which the raw material is used, but the importance of the raw material within the sector is left unaddressed (Løvik et al., 2018).

[20] NACE is the statistical classification of economic activities in the European Community. NACE Rev. 2 is the outcome of a major revision work of the international integrated system of economic classifications which took place between 2000 and 2007. Lastly, 2 digital level is one of the hierarchical levels of this system.
More informational can be found in: https://ec.europa.eu/eurostat/web/nace-rev2

#### 7.2.2.2. Supply risk

Vulnerability to supply disruption means there is a high risk that the materials supply is likely to be inadequate to meet EU industry demand. The supply risk (SR) is based on the concentration of primary supply from other countries and on their level of governance.

Following the recommendation of the 2014 criticality assessment, an initial bottleneck screening is performed to determine the weakest point in the supply chain at which to carry out the assessment (i.e. mining or refining). For example, for indium, the assessment is focused on the processing and refining, while for cobalt it is on mine production.

The supply risk is calculated using Equation 7-2, and its parameters are detailed in the following paragraphs:

$$SR = \left[ \left(HHI_{WGI,t}\right)_{GS} \times \frac{IR}{2} + \left(HHI_{WGI,t}\right)_{EUsourcing} \times \left(1 - \frac{IR}{2}\right) \right] \times (1 - EOL_{RIR}) \times SI_{SR}$$

| | |
|---|---|
| $SR$ | supply risk; |
| $HHI$ | Herfindahl-Hirschman Index; |
| $WGI$ | scaled World Governance Index; |
| $t$ | trade parameter adjusting WGI; |
| $GS$ | global supply; |
| $IR$ | import reliance; |
| $EUsourcing$ | actual sourcing of the EU supply; |
| $EOL_{RIR}$ | end-of-life recycling input rate; |
| $SI_{SR}$ | substitution index related to supply risk. |

Equation 7-2. Supply risk according to the EC methodology.

The Herfindahl-Hirschman Index (HHI) is the most popular means of assessing supply monopolies. In the SR calculation, it is used as a proxy for country concentration. HHI is calculated by squaring the supply percentages of countries producing a given material and then summing the resulting numbers. It results in a theoretical value of between 0 and 10,000, where a higher HHI score indicates a greater risk of supply restrictions.

World Governance Index (WGI) reports aggregate and individual governance indicators for over 200 countries and territories. In the SR calculation, it is used to weight the supply risk originating from country production concentration. It consists of six indices providing information about the perception of different dimensions of governance: (1) Voice and Accountability; (2) Political Stability and Absence of Violence/Terrorism; (3) Government Effectiveness; (4) Regulatory Quality; (5) Rule of Law (RL); and

(6) Control of Corruption. The average of these six dimensions is used as a proxy of the governance of the producing countries.

Hence WGI does not capture risks due to export restrictions, $HHI_{WGI}$ is adjusted by a trade-related variable (t). It is calculated based on three types of restrictions that may be imposed: (1) export tax (when applicable, it varies from 1.1 to 1.3 depending on the percentage of the export tax); (2) export physical quota (calculated as the country production of a given material minus the physical quota imposed divided by the world production); (3) export prohibition (represents an extreme version of the export physical quota). When there is more than one restriction, the highest score should be considered.

In the previous EC methodology (2014), only the global supply was considered, and the dependence of the EU on a combination of supplier countries was not included. The combination of global and EU supply mix provides a more representative measure of the supply risk for the EU. For example, more than half of the global refined indium is produced in China (57%). On the other hand, for indium EU imports, 41% comes from China, followed by 19% from Kazakhstan (which represent less than 1% of global production). For some materials as tungsten, this difference is even higher.

The import reliance (IR) is the ratio of net import to apparent consumption, and it balances the two measures of the supply risk (global supply and actual EU sourcing). Apparent consumption considers domestic production and imports, minus the exports. In the SR, if import reliance is equal to 100%, 50% of the risk is based on global supply and 50% on the EU sourcing.

The end-of-life recycling input rate ($EOL_{RIR}$) measures, for a given element, how much of its input into the production system comes from recycling of old scrap (Eurostat, 2016). This indicator is a good measure of the circular use of the raw materials and its contribution to meeting materials demand in the EU (Mathieux et al., 2017).

The $EOL_{RIR}$ for most of the elements (especially the critical ones) is still limited for different reasons. In the EC criticality methodology, $EOL_{RIR}$ refers only to functional recycling (recyclates used for the same functions and applications as when obtained from primary sources) (Talens Peiró et al., 2018). Figure 7-2 presents the $EOL_{RIR}$ for the elements targeted in this thesis.

The $EOL_{RIR}$ was already included in the previous EC methodology, and the effort of the last update was on improving data quality and defining the sources for calculating it. The data priority order for calculating the $EOL_{RIR}$ is: (1) EU raw material system analysis (MSA) data (BIO by Deloitte, 2015); (2) UNEP report (UNEP, 2011); (3) rates from the previous EC criticality reports or sectorial reports and; (4) an expert judgment.

Lastly, the substitution ($SI_{SR}$) is based on three parameters: (1) substitute production; (2) substitute criticality and; (3) substitute co-production. For the substitute production assessment, only substitutes that are available in large enough quantities in terms of annual production are included (it is considered as 0.8 higher than the given material, or 1 if equal or lower). Regarding the substitute criticality, it is considered as 1 if no substitute is available or the substitute is in the last EU list of CRM, or 0.8 if the substitute is not a CRM. Lastly, the substitute co-production aims to address the influence of the substitute by production – it is considered as 1 if the substitute is mined only as a by-product or if no substitute is available, 0.8 if the substitute is mined as a primary material and 0.9 if it is mined as primary and by-product.

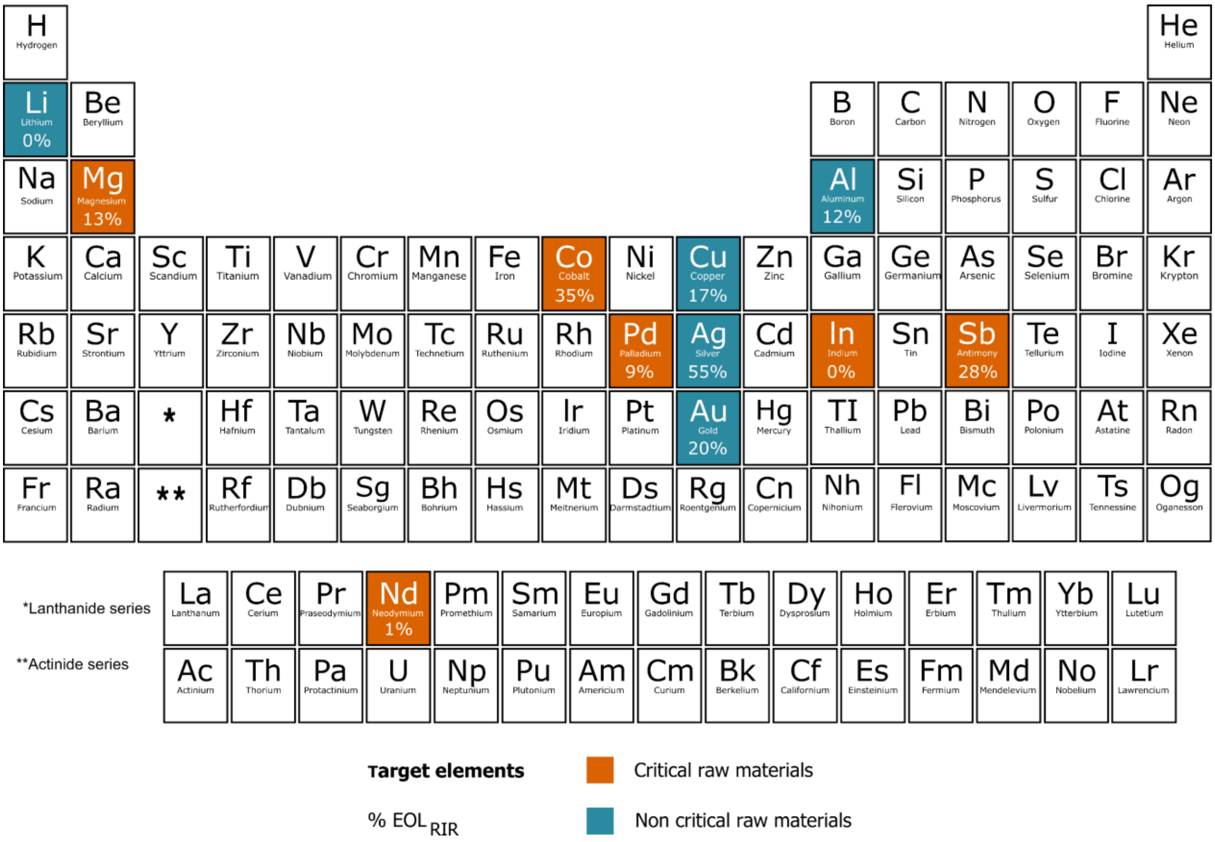


Figure 7-2. End-of-life recycling input rate ($EOL_{RIR}$): focus on elements targeted in the thesis
Source: data based on Vidal-Legaz et al., 2018.

As can be noticed from Equation 7-2, production of secondary raw materials ($EOL_{RIR}$) and substitution ($SI_{SR}$) are risk-reducing filters. Thus, the supply risk increases if the raw material is concentrated in countries with poor governance and decreases if there is a high end-of-life recycling input rate and if it can be easily substituted.

The four improvements proposed by the last EC methodology were: (1) take into account the actual supply to the EU and its import dependency; (2) incorporate trade barriers and agreements; (3) adopt a more systematic supply chain approach; (4) maintain a prominent role for recycling and improve the quality and representativeness of data for the EU.

### 7.2.2. CRM list 2017

The 2017 criticality assessment was carried out for 61 candidate materials – 58 individual materials and 3 material groups: heavy rare earth elements (HREE), light rare earth elements (LREE), platinum group metals (PGM), amounting to 78 materials in total. Figure 7-3 present the results of criticality assessment 2017. Among the raw materials assessed, 27 were identified as critical.

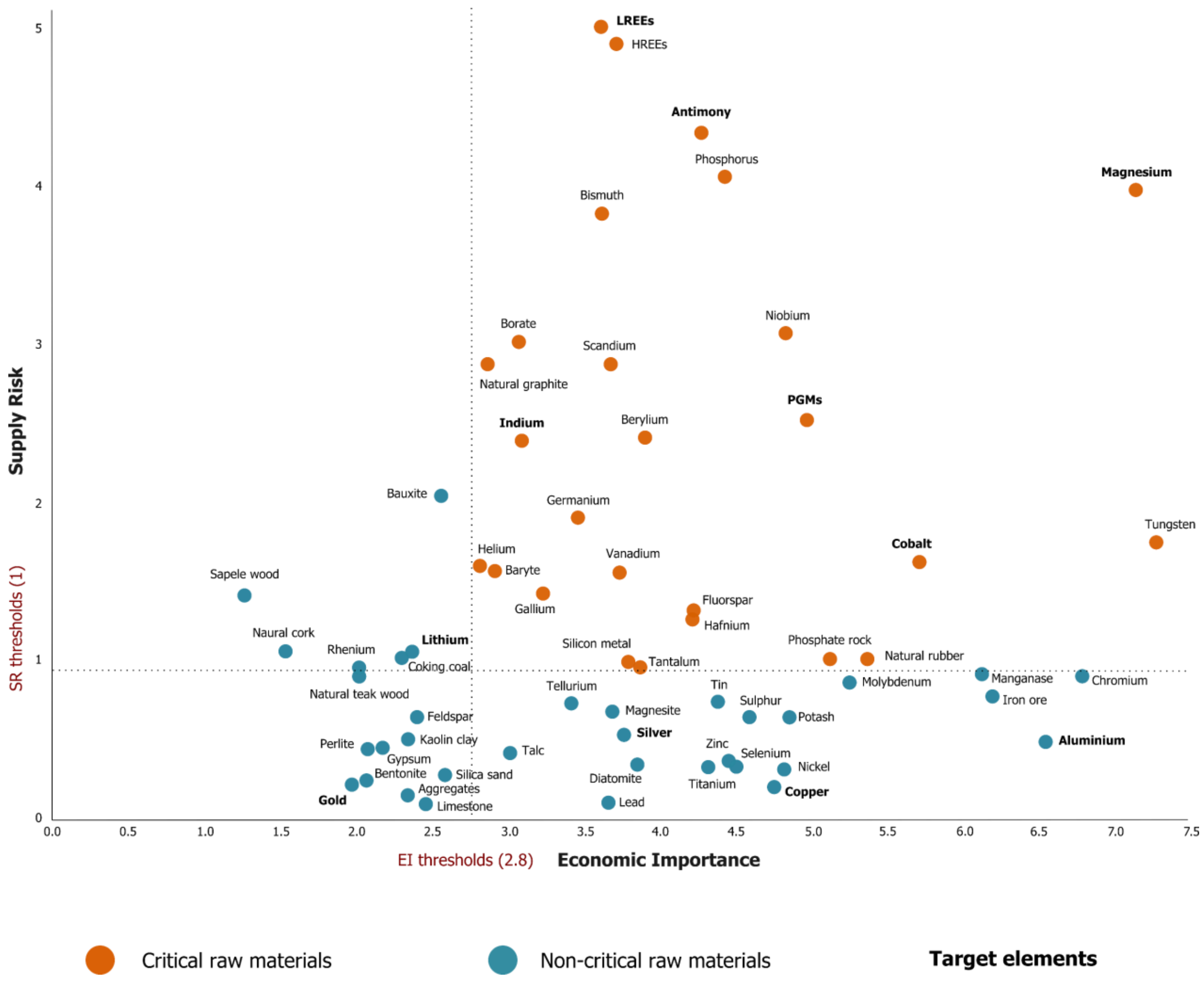


Figure 7-3. EC criticality assessment 2017: economic importance and supply risk results
Source: adapted from Deloitte Sustainability et al., 2017a.

The list of CRMs (European Commission, 2017) has an important role in supporting decision makers for strategic planning and decision-making in government and industry about possible impacts on security of supply in the short term. It is also applied more generally to increase awareness of potential raw material supply risks and related opportunities, and to foster efficient use and recycling of raw materials.

The EU methodology has fixed thresholds for both the economic importance and supply risk dimensions and defined a rectangular "critical zone" within the criticality matrix (SR ≥ 1 and EI ≥ 2.8). The CRM list does not classify the materials in terms of criticality level. In Figure 7-3 the CRMs are highlighted by orange dots, non-critical raw materials are represented by blue dots, and the CRM and non-CRM elements targeted in this thesis are highlighted in bold.

### 7.3. CRM and WEEE

The electric and electronic equipment (EEE) sector depends on a variety of elements, including CRMs, present in different materials and components. The material requirements for EEE will most probably continue to increase, driven by the population rise, wealth growth (mainly in emerging economies), and new technological developments (Peck et al., 2015). Furthermore, some of the metals used in high-tech EEE are produced in only a few countries (Nguyen et al., 2018).

Table 7-1 presents, for the elements targeted in this thesis, their importance within the EEE sector, and the import reliance of the EU.

Table 7-1. Share of target elements in the EEE sector and EU import reliance.

| EC criticality classification | Target elements | Share EEE sector | Supply sourcing | EU import reliance |
|---|---|---|---|---|
| Non-critical raw materials | Silver (Ag) | 24.5% | EU | 59% |
| | Gold (Au) | 11.0% | Global | n.a.[1] |
| | Aluminum (Al) | 7.0% | EU | 64% |
| | Copper (Cu) | 36.0% | Global | 82% |
| | Lithium (Li) | 1.4% | EU | 86% |
| Critical raw materials | Cobalt (Co) | 47.0% | EU | 32% |
| | Indium (In) | 95.6% | EU | 0% |
| | Magnesium (Mg) | 15.0% | EU | 100% |
| | Neodymium (Nd) | 43.4% | EU | 100% |
| | Palladium (Pd) | 10.1% | Global | 100% |
| | Antimony (Sb) | 73.3% | EU | 100% |

[1] meaningful import reliance cannot be calculated because of complexity of trade flows of gold in diverse forms and uncertainties in reported trade data.

Source: data based on Raw Materials Information System (RMIS)[21] and Deloitte Sustainability et al., 2017a.

Some elements, like antimony, reach the EEE sector indirectly – i.e. used to produce flame retardant for plastics (Mathieux et al., 2017). As can be noticed, the import reliance of the EU for some elements (especially the critical ones) is very high. A high dependency does not automatically imply a supply risk, yet as discussed in the previous section, it is a key element together with governance and supply concentration (Vidal-Legaz et al., 2018). Except for indium and cobalt, the EU import reliance for the other target CRMs is from 99 to 100%. China is the EU's leading supplier for magnesium, neodymium and antimony, accounting for, respectively, 94%, 40% and 90% of EU sourcing (European Commission, 2017).

In the EU, cobalt is produced in Finland, Belgium and France, but most of the material sources for refining is imported. The leading suppliers for the EU are Russia (ores and concentrates) and the

[21] RMIS is a comprehensive online repository of information on policies, activities and data related to the European raw materials sector. More information available in: https://rmis.jrc.ec.europa.eu/

Democratic Republic of the Congo (refined cobalt) (Deloitte Sustainability et al., 2017b). Indium is among the few elements that the EU has an import reliance equal to 0%. Belgium accounts for 63% of indium production in the EU, followed by France (37%). Despite that, indium is considered a CRM because the score is calculated based on the global supply risk due to uncertainty about Belgium production. China is the major global supplier of indium (57%) (Deloitte Sustainability et al., 2017b).

The challenges associated with primary resources pose a challenge to the valuable metals production system (Tesfaye et al., 2017). Thus, urban mining is an alternative source of metals, alongside virgin mining, to support the resource demand and contribute to the security of raw materials supply (Mathieux et al., 2017; Tunsu et al., 2015). As presented in section 7.2.2.2, recycling is a risk-reducing filter for supply risk.

WEEE recycling, at a certain scale, may support the conservation of primary ores, saving energy and reducing carbon footprint (Batinic et al., 2018; De Meester et al., 2019; Işıldar et al., 2018). Nonetheless, as for natural ores, an economic benefit is essential to ensure the extraction and processing of waste (Cossu and Williams, 2015).

Among the different (W)EEE products, IT and telecommunication equipment (categories II and VI) as well as lighting equipment (category III), are the richest in terms of precious metals and CRMs (Bigum et al., 2012; Mathieux et al., 2017). According to Işıldar et al. (2018), the most important parameter for treating e-waste without losing CRMs is to know in advance their initial concentrations. Nevertheless, there is little information on the CRM concentration in (W)EEE (Peñaherrera et al., 2018).

Improving the collection and recycling of CRM from WEEE requires tackling a variety of initiatives in the overall process: design of the EEE products, e-waste collection, pre-processing, and recycling. As presented in **Chapter 4**, there is a significant difference between the actual content of CRM in e-waste generated, and the amount collected and recycled by the official schemes.

The first challenge is that (W)EEE is dispersed and that, although element concentrations are high, often the content per device is not. Limited collection by the official schemes results in low recovery of most elements present in WEEE. However, as previously discussed, certain devices and components are more vulnerable to scavenging, which has a different impact on the elements.

Appropriate pre-treatment steps are crucial to prevent irrecoverable losses before final recovery (Hagelüken, 2014). However, current pre-treatment methods are generally not focused on CRMs, but rather on metals present in percentages by weight (e.g. steel, aluminum, copper, brass), and, also, non-metallic fractions, like plastic and glass. Thus, the majority of CRMs are lost, as they stay coupled to dominant metal output fractions, or end up in the dust from the process (Batinic et al., 2018; Tsamis and Coyne, 2015).

Regarding the technical and economic feasibility of recycling, Hagelüken (2014) regrouped the CRM into three main groups:

- Critical metals combined with precious metals that can be recycled economically using existing metallurgical processes (e.g. palladium);
- Critical metals, without associated 'paying' precious metals, that can be technically recycled using (new) dedicated processes (e.g. indium and cobalt);
- Critical metals as part of a complex material mix with thermodynamic constraints (e.g. rare earth elements).

Lastly, it is also necessary to develop a market that uses secondary raw materials to ensure the profitability of the WEEE chain.

### 7.4. Criticality as a parameter to assess materials recovery from WEEE

Given the potential of WEEE as a supplier of secondary raw materials, some authors recently proposed indicators combining criticality assessment with metrics to evaluate the recovery efficiency of materials.

Nelen et al. (2014) developed a set of four indicators to quantify the recycling benefits beyond the weighted-based objectives employed in the WEEE Directive. Among these indicators, the authors proposed a scarcity indicator named "recovery of scarce materials" based on the criticality assessment performed in 2014 (European Commission, 2014). The indicator considers the total weight of materials recycled divided by the total weight in the input of the recycling process, multiplied by the numeric values on economic importance and supply risk per material. The indicator is based on the total weight of the recycling process input. Thus, the full composition of the e-waste is required to calculate it. All materials present in WEEE, but not included in the EC CRMs assessment scope, are given an EI and SR value of zero. The indicator equals 1 in the hypothetical situation that all materials present in the input of the recycling process - whose supply is of concern to the EU - are recovered.

The authors validated the indicators with a case study focused on the treatment of LCD television. The "recovery of scarce materials" in LCD TVs equals to 0.81, which means, according to the authors, that "of the total amount of critical materials in the input, 81% could be recovered". However, this high value is due to the recycling of steel scrap. Although iron is not a CRM, it is among the raw materials assessed by the EC – thus, its SR and EI were not considered as zero. Iron has a low SR (0.8 in the last EC CRM assessment), but its EI is among the highest (6.2). Iron is used by several sectors (more than 50% in the construction and automotive sectors, but only about 3% in domestic appliances).

Van Eygen et al. (2016) assessed the performance of WEEE recycling for desktop and laptop computers in Belgium in 2013. The analysis included an indicator that expresses the recycling efficiency of CRMs. The authors adopted the same approach proposed by Nelen et al. (2014) and named it as "recycled material criticality indicator" (RMC). Firstly, the authors calculated the indicator for all materials in desktop and laptop computers, and the result was similar to Nelen et al. (2014): 87% for desktops and 89% for laptops. For the desktop computers, the elements that contributed the most to the result are ferrous metals (85%), aluminum (9%), and copper (3%) – all of them are non-CRMs. For laptops, magnesium (51%), a CRM available in the housing, followed by ferrous metals (23%) were the elements that contributed the most to the result.

Secondly, the authors calculated the indicator using only the materials deemed critical by the EC, that resulted in an RMC indicator of 43% for desktops, and 95% for laptops. This high value for laptops is due to the high shares of magnesium and cobalt in the input. As discussed in **Chapter 4**, magnesium is mostly recycled as an aluminum alloy. Moreover, it has a high SR (equals 4.0 in the last 2017 CRM assessment), and an even higher economic importance (7.1) due to its use in different sectors. Regarding the cobalt, it has lower criticality results than magnesium (1.6 and 7.1, respectively), but almost 50% of its end-use is for the EEE sector (mainly batteries). As highlighted by the authors, seeing the low recycling rates for most of CRMs, the high values of the RMC indicator should not be interpreted as no further efforts should be made.

Mohamed Sultan et al. (2017) proposed an approach to evaluating material recycling desirability of end-of-life products (not restricted to WEEE). One of the components of this index is the material security based on the materials' criticality. Unlike the studies previously introduced, it is not based on the EC criticality methodology. They use a rank of the 69 highly insecure materials for the United Kingdom (UK) economy in the calculation (Morley and Eatherley, 2008). In this list, the materials are ranked according to eight individual factor combinations that represent the material risk and supply risk. The material risk includes global consumption level, lack of substitutability, global warming potential, and total material requirement. In its turn, supply risk includes scarcity, monopoly supply, political instability and vulnerability to the effects of climate change in key supplying regions.

The recycling desirability, considering the material security index, is calculated as the multiplication of the weight of the given element in the product and its criticality, divided by the total weight of the product multiplied by the higher result of the list. The final index for a given product is equal to the sum of the results per element. This approach allows different products to be ranked according to their material security desirability.

From the analysis of these three studies, we identified four aspects to be explored and/or improved: (1) the scope of the indicator; (2) limitations of the economic importance approach; (3) elements and materials recycling should be addressed differently; and (4) EEE sector importance as a consumer and potential supplier of target elements should be considered. These points are explored further in the following paragraphs.

The scope of the indicators is limited to recycling, and as previously discussed, it is known that several CRMs are lost in the pre-processing treatments and due to limited collection. Thus, in addition to the recycling by the official schemes, we suggest assessing the e-waste collected and generated.

For the first two studies (Nelen et al., 2014; Van Eygen et al., 2016), the higher contribution of non-critical elements materials is mainly due to the economic importance of non-CRMs that also are significant in terms of weight (e.g. copper and iron). As pointed out in section 7.2.2.1, the economic importance calculation does not address the importance of the raw material within the sector. When taking a closer look at the EEE sector (Table 7-1), we notice that most of these elements are generally used in other sectors. Therefore, we suggest that, instead of using the economic importance, the market value of the elements is considered. As discussed in the previous chapters, the economic value is one of the main drivers for e-waste treatment.

Van Eygen et al. (2016) considered in the calculation scope both elements (e.g. gold) and materials recycling (e.g. Al-Mg alloys). Aluminum alloy content from WEEE may be used again in the form of casting alloys for non-structural applications (Nelen et al., 2014). In turn, the major end uses of Mg in the EU are transportation (58%), followed by packaging (16%) (Deloitte Sustainability et al., 2017b). Thus, Mg-Al alloys can partially substitute primary Mg depending on the technical requirements for its use. In this context, we suggest that when the *material* recycling is considered in the calculation, that technical constraints due to the quality and the secondary raw materials' use are also included. One possible solution is to consider the criticality parameters for the material it will substitute. For example, the Al-Mg alloy is an Al scrap thus, thus the supply risk and economic importance of aluminum should be adopted in the calculation.

WEEE recycling can partially contribute to resource demand and the security of raw materials' supply. However, it is limited by the total concentration of elements in (W)EEE. In this context, we suggest normalizing the supply risk by the share of elements within the EEE sector.

## 7.5. Criticality weighted-based indicators

Inspired by the studies previously introduced, in this section we suggest three indicators to be used by the compliance schemes to report the WEEE chain performance from the outlook of criticality. Like the approach presented in the previous chapters, the proposed indicators intend to assess the WEEE chain performance beyond the weight-based approach.

### 7.5.1. Criticality weighted-based collection rate

The goal of this indicator is to assess the collection rate by the official schemes, including the criticality of the target elements, in addition to their weight. The criticality-weight based collection rate is calculated with Equation 7-3. The scope of assessment is the elements collection, and it can be applied to analyze all target elements, or a group of critical and non-critical elements.

Differently to the studies presented in previous section, we consider only the supply risk according to the EC methodology, together with the market price of the elements. Moreover, since this indicator aims to capture the WEEE chain capacity to supply secondary raw materials for the economy, we include their importance within the EEE sector.

$$CCR_f = \frac{\sum_{l=1}^{n4} WC_{el} \times SR_{el} \times SH_{el} \times MP_{el}}{\sum_{l=1}^{n4} WG_{el} \times SR_{el} \times SH_{el} \times MP_{el}}$$

| | |
|---|---|
| $CCR_f$ | criticality weighted-based collection rate per WEEE category; |
| $n4$ | number of target elements considered in the assessment; |
| $WC_e$ | weight of target element collected by the official schemes (calculated based on Equation 4-3 detailed in **Chapter 4**); |
| $SR_e$ | supply risk of the element based on EC criticality methodology; |
| $SH_e$ | share of element used within EEE sector; |
| $MP_e$ | market price of the target element. |
| $WG_e$ | weight of target element generated (calculated based on Equation 4-3 detailed in **Chapter 4**); |

Equation 7-3. Criticality weighted-based collection rate.

### 7.5.2. Criticality weighted-based recycling rate

This indicator measures in addition to the technical efficiency (pre-processing and end-processing), the criticality of the target elements recycled by the official schemes (Equation 7-4).

Its scope comprises from the e-waste collected by the official schemes, to the elements recycling. It is complementary to the previous indicator and allows an overview of CRM recovery beyond the weight-based approach.

$$CRR_f = \frac{\sum_{l=1}^{n4} WR_{el} \times SR_{el} \times SH_{el} \times MP_{el}}{\sum_{l=1}^{n4} WC_{el} \times SR_{el} \times SH_{el} \times MP_{el}}$$

| | |
|---|---|
| $CRR_f$ | criticality weighted-based recycling rate per WEEE category; |
| $n4$ | number of target elements considered in the assessment; |
| $WR_e$ | weight of target element recycled by the official schemes (calculated based on Equation 4-9 detailed in **Chapter 4**); |
| $SR$ | supply risk of the element based on EC criticality methodology; |
| $SH_e$ | share of element used within EEE sector; |
| $MP_e$ | market price of the target element; |
| $WC_e$ | weight of target element collected by the official schemes (calculated based on Equation 4-3 detailed in **Chapter 4**). |

Equation 7-4. Criticality weighted-based recycling rate.

### 7.5.3. Potential contribution for apparent consumption

This indicator aims to motivate e-waste recycling to reduce the demand for raw materials. It is calculated as the ratio between the potential contribution of elements generated by WEEE and not recycled by the official schemes, and the apparent consumption in France (Equation 7-5).

$$PC_e = \frac{WG_e - WR_e}{AC_e}$$

| | |
|---|---|
| $PC_e$ | potential contribution of target elements generated and not collected by the official schemes for apparent consumption (per target element in the WEEE category); |
| $WG_e$ | weight of target element generated (calculated based on Equation 4-3 detailed in **Chapter 4**); |
| $WR_e$ | weight of target element recycled by the official schemes (calculated based on Equation 4-9 detailed in **Chapter 4**); |
| $AC_e$ | apparent consumption of the target element in France. |

Equation 7-5. Potential contribution for apparent consumption per target element.

## 7.6. Case study: criticality assessment of screens treatment in France

The screens category is selected as a case study. Firstly, based on the methodological approach presented in **Chapter 4,** it is necessary to quantify the target elements generated ($WG_e$), collected ($WC_e$) and recycled ($WR_e$) by the official schemes, and the flows diverted in the complementary flows ($WG_e - WC_e$). The sub-sections present the other data required, and the results obtained with the indicators introduced in the previous section.

### 7.6.1. Screens: collection rate and recycling by the official schemes

After quantifying the element flows within the screens stream, the following step is the correlation of the elements' weights, with their supply risks (Table 7-2), the share within the EEE market (Table 7-1) and the market value of the elements in the given year (Figure 6-6).

The results analysis was performed on two levels: initially, we assessed the overall results for the 11 target elements, and then we analyzed individually the group of elements deemed to be critical and non-critical according to the EC. Besides assessing the indicators' results, the share of the elements within the results is analyzed.

Afterwards, the criticality results are compared with the weight-based and economic weighted-based results with the same calculation metric (ratio between quantity collected and generated). The economic weighted-based considers the market value of the elements multiplied by the elements weight.

Table 7-2. Supply risk of the target elements according to EC methodology 2017.

| EC criticality classification | Target elements | Supply risk |
|---|---|---|
| Non-critical raw materials | Silver (Ag) | 0.5 |
| | Gold (Au) | 0.2 |
| | Aluminum (Al) | 0.5 |
| | Copper (Cu) | 0.2 |
| | Lithium (Li) | 1.0 |
| Critical raw materials | Cobalt (Co) | 1.6 |
| | Indium (In) | 2.4 |
| | Magnesium (Mg) | 4.0 |
| | Neodymium (Nd) | 4.9 |
| | Palladium (Pd) | 1.7 |
| | Antimony (Sb) | 4.3 |

Source: Deloitte Sustainability et al., 2017a, 2017b.

Table 7-3 presents the results for the indicator "criticality weighted-based collection rate" based on Equation 7-3 (column 4), as well as the weight-based and economic weighted-based results (respectively, columns 2 and 3). As can be noticed, when other parameters are considered in addition to the elements' weight, the overall collection rate decreases.

Figure 7-4 presents the share of target elements generated and collected by the official schemes for screens in 2018 considering the element weights. As discussed in the previous chapters, the elements significant in terms of weight are the most important within this approach.

Table 7-3. Collection rate based on waste generated approach: comparison between different metrics.

| Target elements | Weight | Economic | Criticality |
|---|---|---|---|
| Overall | 27.7% | 23.3% | 24.1% |
| Critical | 31.2% | 24.6% | 23.5% |
| Non-critical | 27.0% | 23.1% | 27.4% |

The overall collection rate considering the market price and the criticality approach introduced in section 7.5.1 is, respectively, 15.8% and 12.9% lower than the weight-based approach. When only the economic parameter is considered, as discussed in **Chapter 6**, the result is influenced mainly by the high market value of gold (non-critical) and palladium (critical), that are mostly concentrated in flat panel displays (FPD) that are not yet significantly collected by the official schemes.

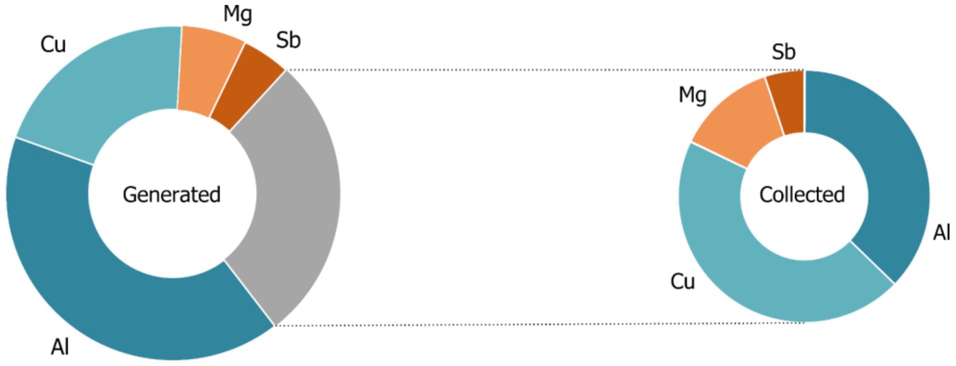


Figure 7-4. Weight-based collection rate.

The criticality result is higher than the economic weighted-based collection rate due to the collection of antimony (mostly concentrated in CRT displays) (see Figure 7-5). Moreover, within the criticality approach, other elements present in very small concentrations in screens, such as neodymium, have a greater importance and can be identified as key elements to be recovered in order to increase the performance in terms of criticality.

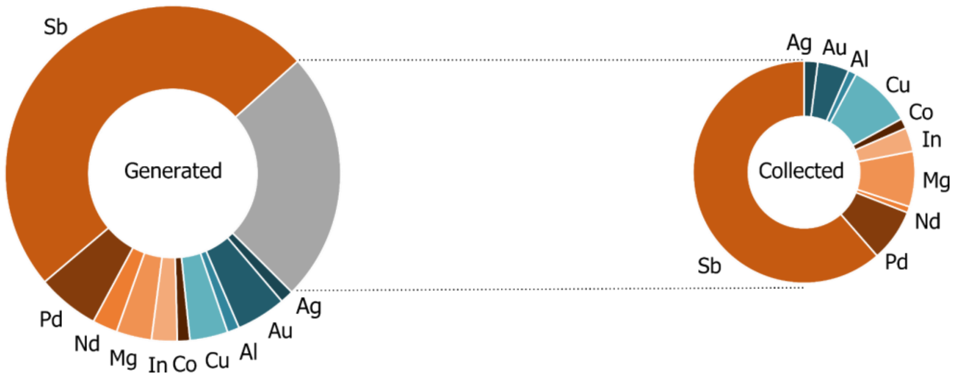


Figure 7-5. Criticality weighted-based collection rate.

Although antimony has a low weight in comparison to other elements as Cu and Al, and a low market price in comparison to Au and Pd, it has a high supply risk, and the EEE sector is among the major users of Sb in the EU. Consequently, WEEE could be a supplier of secondary material – however, its recycling is limited by technical and economic issues (see **Chapter 4**).

Regarding magnesium, from a weight perspective it is present in a concentration 3.5 times lower than copper, but in the criticality-weighted based approach, its recovery has almost the same importance as copper. This is because the supply risk of magnesium is 20 times higher than copper.

Taking a closer look at the target elements deemed to be critical by the EC methodology, the result is 24.7% lower than the weight-based approach, due to the low collection of elements highly present in FPD screens. The difference is negligible for the non-critical elements.

Following the same assessment approach, Table 7-4 presents the results for the "criticality weighted-based recycling rate". The higher result is achieved by the economic approach, motivated mainly by Au (49.6%), Cu (28.3%) and Pd (10.7%) recycling. The overall criticality result is 64.4% lower than the weight-based approach.

Table 7-4. Recycling rate based on waste generated approach: comparison between different metrics.

| Target elements | Weight | Economic | Criticality |
|---|---|---|---|
| Overall | 61.4% | 74.8% | 21.8% |
| Critical | 0.6% | 47.8% | 9.4% |
| Non-critical | 74.6% | 80.4% | 79.5% |

Based on the weight-based approach, Figure 7-6 presents, on the left, the share of elements recycled from the total screens collected by the official schemes (in grey) and the recycling losses per target element (the critical in orange and the non-critical in blue). Then, on the right, from the target elements recycled, the elements' distribution in terms of weight. It clearly illustrates the aspect discussed in the previous chapters and highlighted in the collection rate results for the CRM and non-CRM: current recycling is focused on non-CRM that are prevalent in terms of weight and less than 1% of critical elements is recycled.

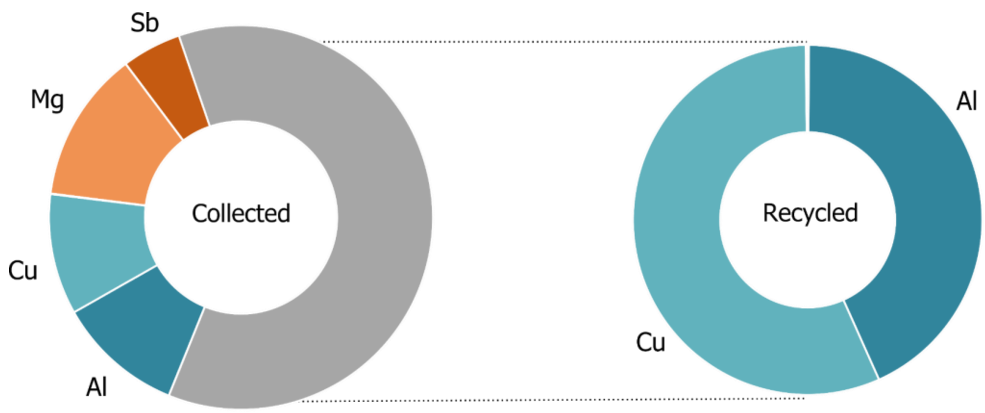


Figure 7-6. Weight-based recycling rate.

Figure 7-7 presents the results for the criticality weight-based approach. The percentage of WEEE recycled is presented in grey on the left, and its composition is detailed on the right. The criticality weighted-based overall result is mainly influenced by antimony and magnesium (21.8%). In this study we considered that only the Sb present in PCB is partially recycled, and that magnesium is not recycled. As discussed in section 7.4, Mg is recycled as Al-Mg alloys (material) and is not in the scope of the indicator.

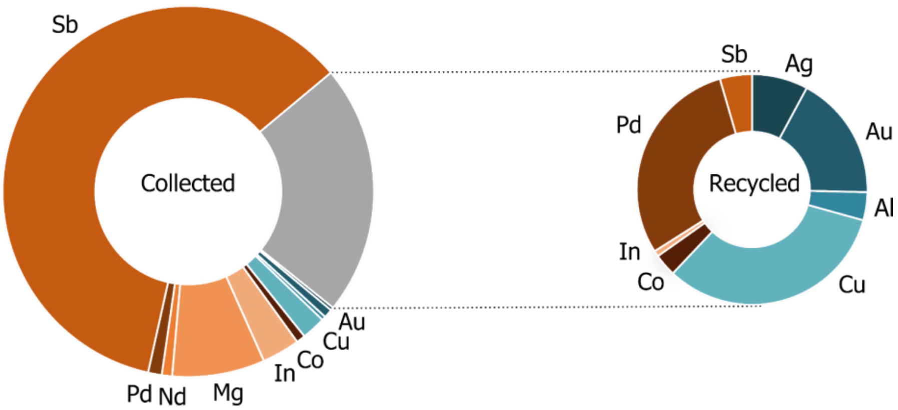


Figure 7-7. Criticality weighted-based recycling rate.

Among the elements recycled by the official flows, the most significant from the criticality approach are palladium (that has a small weight but is a precious metal) and copper. Due to the result of Pd recycling, when assessing the indicator focused on the elements deemed critical by the EC the result is higher than the weight-based approach.

### 7.6.2. Screens: potential contribution for target elements demand

This indicator aims to quantify the potential of elements currently not recycled by the official systems as suppliers of the overall consumption of raw materials in France. The apparent consumption in France is calculated based on the approach presented in **Chapter 5**. The results and parameters are presented in Table 7-5.

Table 7-5. Potential of WEEE screens to supply the apparent consumption of target elements in France.

| Target elements | Quantity of elements not recycled by the official schemes (t) | Apparent consumption in the EU (t) | Estimated apparent consumption in France (t) | Potential contribution (French scope) |
|---|---|---|---|---|
| Silver | 8.28 | 8,827.39 | 509.24 | 1.63% |
| Gold | 2.07 | 98,730.61 | 5,695.66 | 0.04% |
| Aluminum | 6,789.21 | 6,119 769.96 | 353,043.07 | 1.92% |
| Copper | 5,081.31 | 4,661 804.64 | 268,934.59 | 1.34% |
| Lithium | 0.05 | 1,807.63 | 104.28 | 0.02% |
| Cobalt | 9.54 | 1,807.63 | 104.28 | 9.14% |
| Indium | 2.66 | 22.20 | 1.28 | > 100% |
| Magnesium | 1,496.64 | 118,281.03 | 6,823.51 | 21.93% |
| Neodymium | 7.37 | 571.47 | 32.97 | 22.36% |
| Palladium | 0.44 | 115,097.24 | 6,639.84 | 0.01% |
| Antimony | 940.74 | 18,162.10 | 1,047.75 | 88.79% |

These results, together with an assessment of the technical and economic limitations of the current processes, can be used to boost and support the development of recycling processes to allow the recovery of secondary materials.

This assessment can be performed focused on the French scale, as well as considering the EU boundaries: elements generated and treated across the different Member States, and the total EU apparent consumption (column 3).

## 7.7. Conclusions

This chapter explores the potential of e-waste as an alternative supplier for the raw materials demand. We present a complementary approach for the technical (**Chapter 4**), environmental (**Chapter 5**) and economic (**Chapter 6**) aspects based on the criticality to assess the performance of WEEE schemes. The European Commission (EC) criticality assessment methodology is used as the background for our proposal.

Table 7-6 summarizes the indicators, as well as the sections in which the indicators are validated. This set of indicators aims to reply to the third research question of this thesis presented in **Chapter 1** regarding alternative metrics than the weight-based approach for evaluating the impacts and benefits of the Waste Electric and Electronic (WEEE) chain.

Table 7-6. Summary of the criticality indicators proposed in Chapter 7.

| Criticality indicators | | Equations | Validation with case study |
|---|---|---|---|
| 1. | Criticality weighted-based collection rate | Equation 7-3 | Section 7.6.1 |
| 2. | Criticality weighted-based recycling rate | Equation 7-4 | Section 7.6.1 |
| 3. | Potential contribution for apparent consumption | Equation 7-5 | Section 7.6.2 |

Indicators 1 and 2 include criticality priorities (supply risk and market value) related to raw materials, besides the current weight-based collection and recycling rate approach. We concluded that when other parameters are considered in addition to the elements' weight, the overall collection rate decreases. Moreover, within the criticality approach, other elements present in minimal concentrations in screens, such as neodymium, have greater importance. The criticality weighted-based recycling rate is mainly influenced by antimony (Sb), magnesium (Mg), and palladium (Pd). In this study, we considered that only the Sb present in PCB is partially recycled and that magnesium is not recycled. As discussed in the previous sections, Mg is sorted as aluminum (Al) scrap and is recycled as Al-Mg alloys (as a material and not an element) and is not in the scope of the indicator.

The third indicator assesses the potential of the elements not currently recycled by the official schemes to meet the current demand for raw materials in France. We estimated that some of the CRMs targeted in our study could supply more than 20% of the current raw material demand, highlighting the importance of e-waste as a source of urban mining. The results of the case study reinforce what other authors have defended in recent years that the targets should not be based on overall weight, but on the recovery of individual materials and elements (Bigum et al., 2012; Huisman et al., 2008; Van Eygen et al., 2016). In addition, it illustrates the real interest of indicators that can monitor the performance of the e-waste chain as a supplier of quality secondary (critical) raw materials.

The indicators presented in this chapter conclude the indicators' propositions of this thesis. As presented in section 7.6, the criticality approach is complementary to the technical and economic analysis. In **Chapter 8** we go further in the multi-dimensional analysis.

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## Chapter 8 | Conclusions and recommendations

*"O que importa não é a verdade intrínseca das coisas,*
*mas a maneira como elas vão ser contadas ao povo.*[22] *"*
Chico Buarque e Ruy Guerra, Calabar: o elogio da traição (1973).

### 8.1. Synthesis of the research contributions

The core of this thesis was to establish an indicator dashboard offering a more holistic performance assessment of the WEEE chain, including technical, economic, environmental, and criticality aspects. The results of this work intend to improve visibility on the progress of the official WEEE schemes in a circular economy. Moreover, the thesis aims to support the development of the future specifications provided by the French Ministry of Environment in the next agreement period for the compliance schemes, that will come into force from 2021.

The social impacts of the WEEE chain are not the focus of this thesis, but according to the current specifications provided by the Environmental Code, the compliance schemes must also provide information on employment and integration in the official e-waste chain. As pointed out in section 3.3, we identified in the literature review some propositions that can be further explored by the official schemes in order to report data on working conditions (e.g. income, level of education, equal opportunities for men and women).

The indicators are presented and validated in the previous chapters with a case-study considering data and particularities of the e-waste chain in France. Nevertheless, we believe that the metrics introduced in this study could be used in the other Member States. Moreover, seeing that e-waste treatment (official and the complementary flows) is not limited to country boundaries, it is important to monitor and develop strategies to improve WEEE recovery in the whole European Union (EU) context. EU monitoring is even more relevant when looking e-waste as a source of a secondary resource.

Besides the set of indicators itself (summarized in Table 8-1), the methodological approach adopted in this work is also a contribution to future research in the field, and goes hand in hand with discussions presented by other authors. The main parameters adopted in this study are briefly presented in section 8.2.1.

As described in **Chapter 1**, the structure of this thesis follows the research questions. The main findings that grounded the response to the research questions are summarized in the following sub-sections.

[22] What matters is not the intrinsic truth of things, but the way they're going to be told to the people.

Table 8-1. Set of indicators proposed in this thesis.

| Type of indicator | Indicators | Section |
|---|---|---|
| Technical | 1. EEE placed on the market per inhabitant | 4.5.1 |
| | 2. Collection rate based on the WEEE generated approach | 4.5.2 |
| | 3. Collection rate per target element | 4.5.3 |
| | 4. Rate of departments complying with the WEEE collection target | 4.5.4 |
| | 5. Rate of plastics that may contain brominated flame retardants in WEEE collected | 4.5.5 |
| | 6. Reuse rate by the official schemes per WEEE category | 4.5.6 |
| | 7. Recycling rate by the official schemes per WEEE category | 4.5.6 |
| | 8. Treatment rate based on WEEE generated approach | 4.5.7 |
| | 9. Recycling rate per target element | 4.5.8 |
| Environmental | 10. Environmental impact avoidance opportunity | 5.7.1 |
| | 11. Environmental impact avoidance opportunity | 5.7.2 |
| Economic | 12. Visible fee per EEE placed on the market | 6.3.1.1 |
| | 13. Visible fee per inhabitant | 6.3.1.2 |
| | 14. Total costs per WEEE collected | 6.3.1.3 |
| | 15. Total costs per WEEE treated | 6.3.1.4 |
| | 16. Total costs per inhabitant | 6.3.1.5 |
| | 17. Economic weighted-based collection rate | 6.3.2.1 |
| | 18. Economic weighted-based recycling rate | 6.3.2.2 |
| | 19. Economic value of the complementary flows | 6.3.2.3 |
| Criticality | 20. Criticality weighted-based collection rate | 7.5.1 |
| | 21. Criticality weighted-based recycling rate | 7.5.2 |
| | 22. Potential contribution for apparent consumption | 7.5.3 |

### 8.1.1. Indicators currently used by the compliance schemes and policy-makers

The WEEE Directive, together with other regulations at the national and EU level, was a significant breakthrough in e-waste management. Its indicators and targets (collection, recycling and reuse rate, and recovery rate – detailed in section 2.2.2) allowed and fostered Member States in the development of WEEE chain systems, but they provide a limited overview based on the overall weight-based results. Although the Directive presents an alternative approach for the collection rate calculation based on the e-waste generated it is not yet widely adopted by the Member States, (the most commonly used approach is based on the average weight of equipment placed on the market in the three previous years).

The WEEE generated approach allows a better assessment of the real collection performance (including equipment lifetime and storage), as well as of the complementary flows. As presented in section 3.2, at the national level, the compliance schemes and ADEME monitor other indicators than the ones imposed by the regulation. These indicators aim to track the equipment placed on the market, the collection set-up, and e-waste treatment until the end-of-waste status is achieved.

Besides the technical indicators, we identified some economic indicators reported by the compliance schemes. Nonetheless, we concluded that these indicators are, in fact, only data on incomes and expenses of the official schemes. They do not allow any understanding of the economic performance of WEEE systems. In order to interpret them, it is necessary to analyze other data from the performance of the chain (e.g. collection cost per ton of WEEE collected by the official schemes).

Moreover, both technical and economic indicators are reported as overall results of the WEEE chain (e.g. collection rate, the amount in euros collected with the visible fee) or, in some cases, per category (e.g. recycling and reuse rate). The monitoring of the quality of e-waste collected and treated (the type of products generated and collected, components scavenging, materials and elements recovered) is not currently practiced.

According to the most recent specifications included in the Ministerial Orders for household and professional WEEE (JORF, 2015, 2014), the French compliance schemes must also report other indicators related to the employment, integration and environmental impacts in the context of the official WEEE chain. However, neither the method nor the scope for calculating the indicators is detailed by the authorities.

### 8.1.2. Indicators and metrics to assess WEEE chain technical performance

The technical indicators presented in **Chapter 4** are based on the monitoring of e-waste from WEEE generation (equipment placed on the market and its consequent disposal after a few years) until the recycling of some target elements.

The methodological approach is based on five assessment levels: flows (WEEE categories or e-waste streams), products, components, materials and elements (see Figure 8-1). The current approach only monitors the macroscopic level, and we believe that to improve material and element recovery it is necessary to monitor e-waste flows with a higher level of detail.

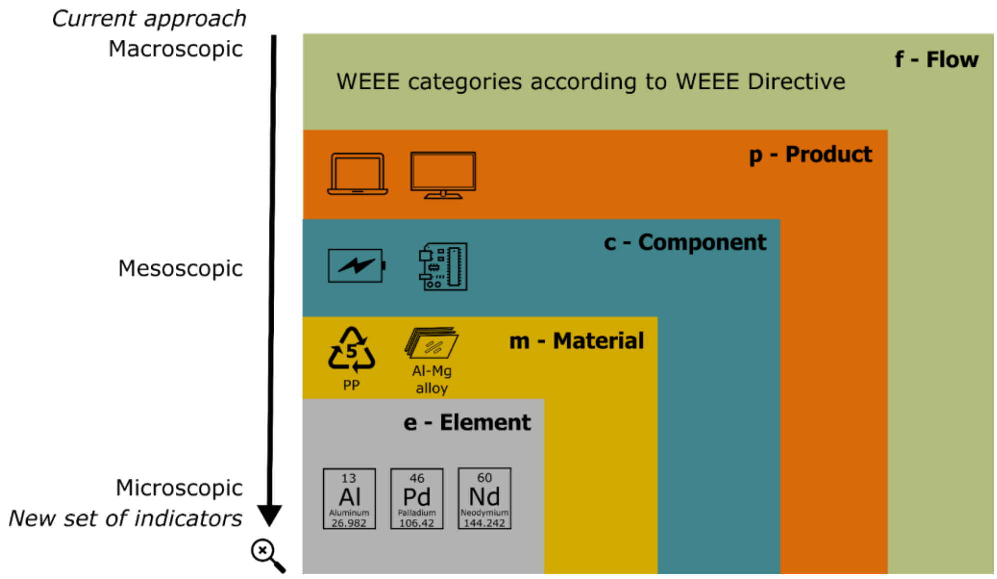


Figure 8-1. Levels of e-waste monitoring to support a better performance assessment.
Source: adapted from Huisman et al., 2017; Zeng et al., 2017.

Figure 8-2 presents the nine technical indicators, their assessment level and summarizes the results of the French case study discussed in section 4.6. Among the nine technical indicators proposed in this work, four are focused on the macroscopic assessment per category ("Rate of departments complying with the WEEE collection target", "Reuse rate by the official schemes per WEEE category", "Recycling rate by the official schemes per WEEE category" and "Treatment rate based on WEEE generated approach").

Two indicators are based on a microscopic assessment per element ("Collection rate per target element" and "Recycling rate per target element"). For the element recycling, the information on the composition of each material, component and product (mesoscopic level) is mandatory to estimate the treatment efficiency. In addition, one indicator ("Rate of plastics that may contain brominated flame retardants in WEEE collected") is focused on the material level (rate of plastic that contains brominated flame retardants).

Indicators "EEE placed on the market per inhabitant" and "Collection rate based on the WEEE generated approach" provide a result per category, but we suggest monitoring the products (UNU-keys) placed on the market, generated and collected in order to increase the understanding of the actual and future types of devices that will be treated by the official schemes. In the case of screens, we observed a significant difference in the collection performance of the cathode ray tube (CRT) and flat panel display (FPD) screens. In 2018, FPD represented 51.7% of WEEE generated. However, 77% of the screens collected by the official schemes were CRT. Consequently, the collection rate for CRT and FPD is completely different: 62% and 17%, respectively.

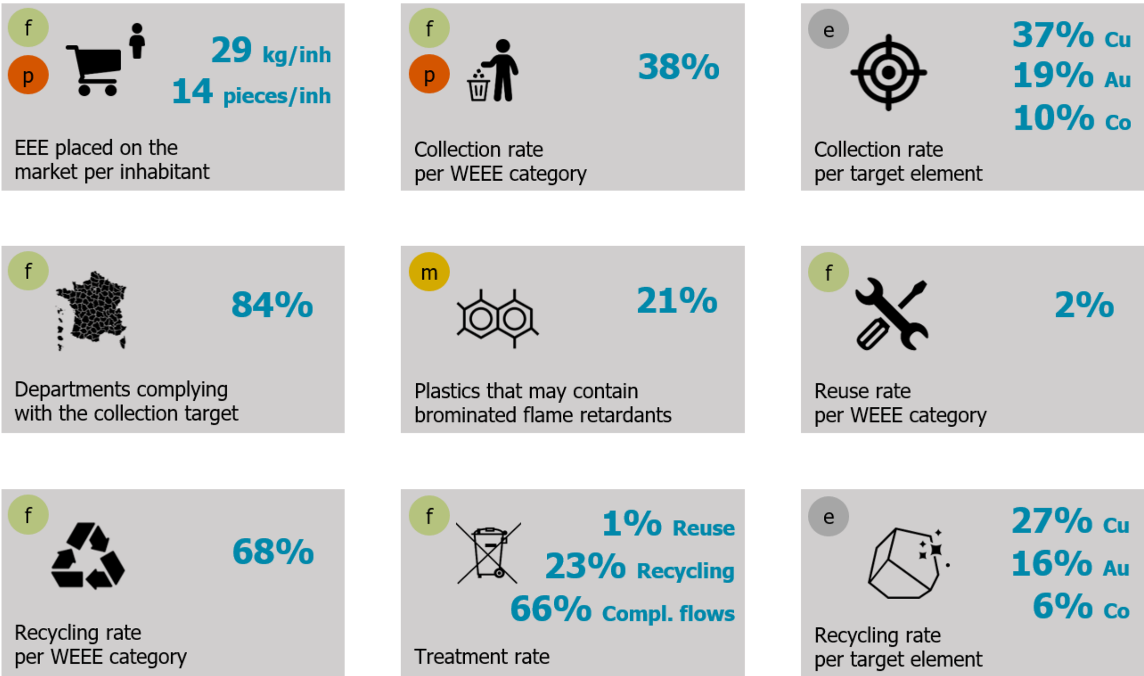


Figure 8-2. Dashboard of technical indicators.

CRT and FPD have different compositions depending on the type, functionalities, and producers of the devices. Thus, changes in the type of screens generated and collected impact collection rate based on the overall weight, and also per target element. A 17-inch CRT screen has an average weight of 20 kg, in which approximately 50% is CRT glass, 15% metals, 15% plastics and 6% printed circuit boards (PCB) (Eco-Systèmes, 2013a; Hischier et al., 2007). A laptop has an average weight of 2 kg, in which the leading materials and components, in terms of weight, are plastics (from 20 to 40%), metals (25%) and PCB (15%) (Eco-Systèmes, 2013b; Hischier et al., 2007; HP Development Company, 2018).

In the case study, we estimated that the collection and recycling rate of the main elements present in FPD, and more specifically in tablets and laptops', is three to four times lower than elements present in high percentages in CRT devices. Thus, it is important to have indicators to monitor the quality of e-waste collected and recycled by the official schemes.

This approach allowed us to quantify WEEE chain performance with a level of detail that was not possible before. Figure 8-3 presents some of the results obtained using the technical indicators we had validated. It illustrates results obtained with indicators "Collection rate per target element" and "Recycling rate per target element", in addition to providing data on the grade and size of complementary flows, and losses because of recycling limits.

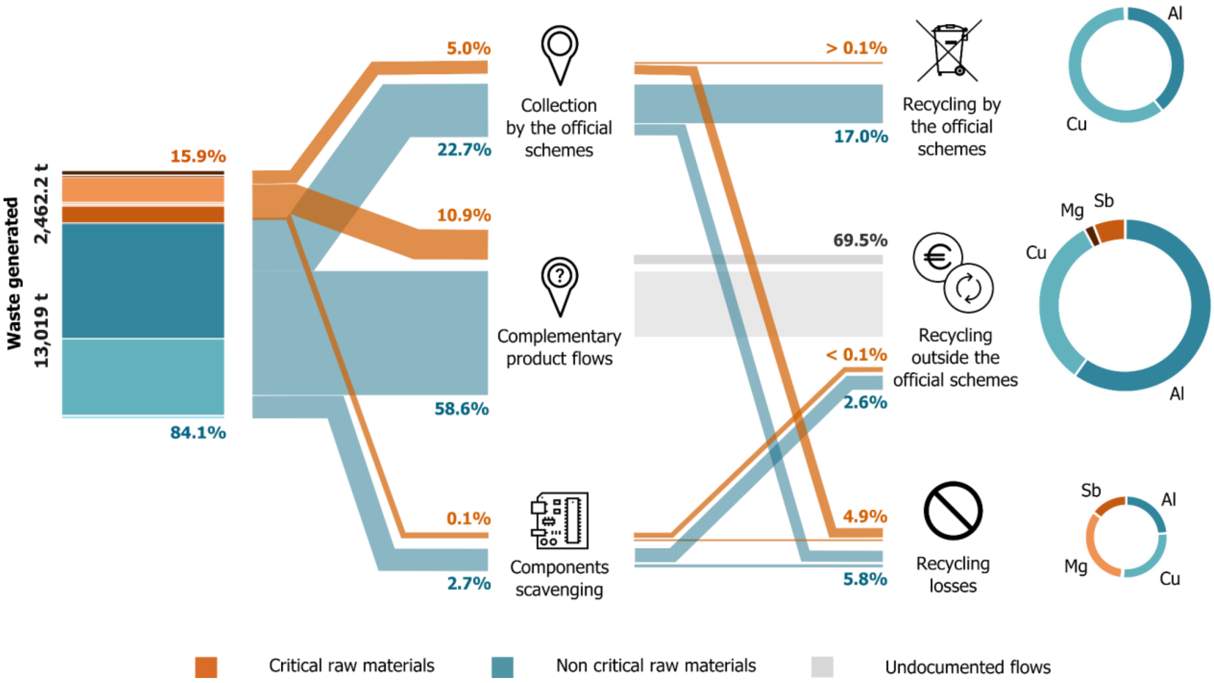


Figure 8-3. WEEE screens flows: target materials generated, collected and recycled (weight-based approach).

From a weight-based perspective, in 2018, non-critical raw materials account for 84% of target elements generated from screens in France, mostly due to high aluminum (Al) and copper (Cu) content in screens. Regarding the collection, a significant amount of e-waste is diverted into complementary product flows and components scavenging. Considering the size of complementary flows (around 70% of the total weight of target elements generated in screens), they cannot be neglected, highlighting the interest of the waste generated approach.

The results of the case study confirm that monitoring the performance per target element in comparison to overall weight indicators allows an overview of the volume and the grade of potential secondary resources. It also allows the identification of the main challenges to improve materials recovery towards a circular economy.

### 8.1.3. Indicators and metrics for assessing WEEE chain environmental and economic performance

The search for metrics to evaluate the WEEE chain beyond the weight-based approach was divided into three chapters: environmental (**Chapter 5**), economic (**Chapter 6**), and criticality (**Chapter 7**). This assessment followed the same five assessment levels presented in the previous section.

#### Environmental assessment

In the review performed to identify the indicators proposed in the literature, among the studies that proposed environmental indicators, the majority used Life Cycle Assessment (LCA). Thus, we explored the potential of LCA to provide results on the impacts and benefits of the WEEE chain. Based on the assessment of several LCA studies for WEEE products (section 5.3) and the case study (section 5.5), we confirm that LCA can provide interesting insights into the impacts and benefits of the WEEE chain. Among other issues, we concluded that the total global warming and human toxicity impacts are decreasing over the years, with the reduction in the number of CRT monitors and TVs collected. However, the treatment scenarios and the results are influenced by data limitations (sections 5.4 and 5.6).

In this context, we recommend developing LCA studies with a cradle-to-gate approach by e-waste treatment providers (who can gather primary data), for example, to compare recycling paths from an environmental outlook (Grimaud et al., 2017). We concluded, due to data limitations and the complexity of the method, that LCA is not suitable for reporting overall results by the official chain treatment (i.e. it is relevant from the outlook of research, but not for on-the-ground application). Thus, instead of suggesting an indicator based on the assessment of impacts per ton of WEEE treated (information which is difficult to quantify based on current compliance schemes), we suggest two environmental indicators based on the inventory of e-waste treatment published by the ESR database (section 5.4.2).

Figure 8-4 presents the two environmental indicators proposed in this work, their assessment level and summarizes the results of the French case study discussed in section 5.7.

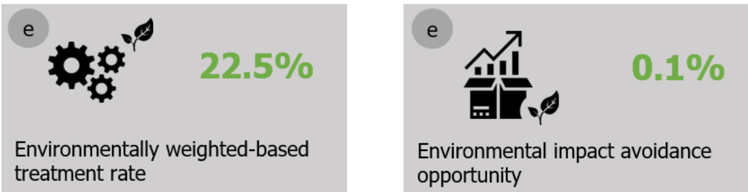


Figure 8-4. Dashboard of environmental indicators.

The indicator "Environmentally weighted-based treatment rate" (see section 5.7.1) uses data on the composition of e-waste generated and collected by the official schemes, information on the avoided impacts by the WEEE treatment based on ESR database and life cycle impact assessment (LCIA). This indicator can support the shift from the current approach of focusing on elements present in higher weights, to boost the collection and treatment of elements with greater environmental interest. As presented in Figure 8-5, when including the environmental aspect in the assessment, the share between the target elements changes due to the different environmental performance of the target elements.

The main limitations of the indicator are related to the database's scope. Currently, it contains a limited number of products (e.g. CRT are not included), for a limited range of materials and elements, and the treatment scenarios are not disclosed.

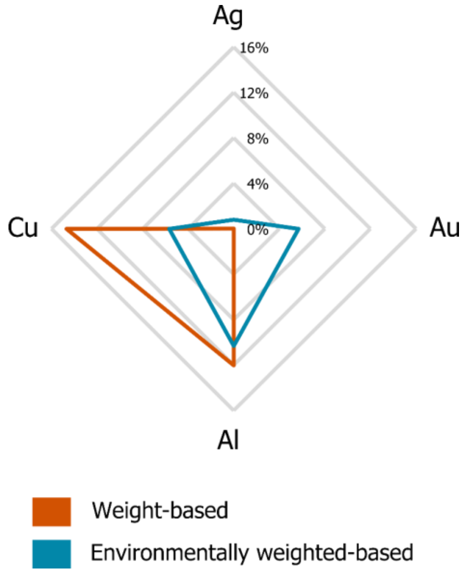


Figure 8-5. FPD treatment rate: weight-based and environmentally-based approaches.

In turn, indicator "Environmental impact avoidance opportunity" (section 5.7.2) estimates the opportunity of the complementary flows (e-waste dissipated from the official schemes), to avoid the environmental impacts of the current consumption of target elements by the French economy.

**Economic assessment**

After looking more deeply into the economic model of the WEEE chain, as well as identifying its costs and profits from the viewpoint of the compliance schemes (section 6.2), we developed eight indicators, regrouped in two sets of economic indicators. The eight economic indicators, their assessment level and the results of the French case study discussed in section 6.4 are presented in Figure 8-6.

The first set, named as cost-effectiveness, regroup five indicators ("Visible fee per EEE placed on the market", "Visible fee per inhabitant", "Total costs per WEEE collected", "Total costs per WEEE treated", "Total costs per inhabitant") that aim to improve monitoring and reporting of revenues and expenses (section 6.3.1). These indicators can be used to improve the reporting of costs and revenues and to increase the awareness of consumers regarding the costs of the WEEE chain and the importance of the visible fee. In the case study, we could not assess a specific WEEE category because the compliance schemes do not disclose this data. We recommend reporting per category in order to allow a better understanding of the particularities per type of e-waste. Magalini and Huisman (2018), estimated that the costs and revenues per categories can differ significantly.

The second set of indicators, named as economic benefits (section 6.3.2), present an alternative approach to the current weight-based indicators and regroup three indicators ("Economic weighted-based collection rate", "Economic weighted-based recycling rate", "Economic value of the complementary flows"). This approach can be used for decision making, for example, to prioritize the recycling of materials considering not only their weight but also their market value.

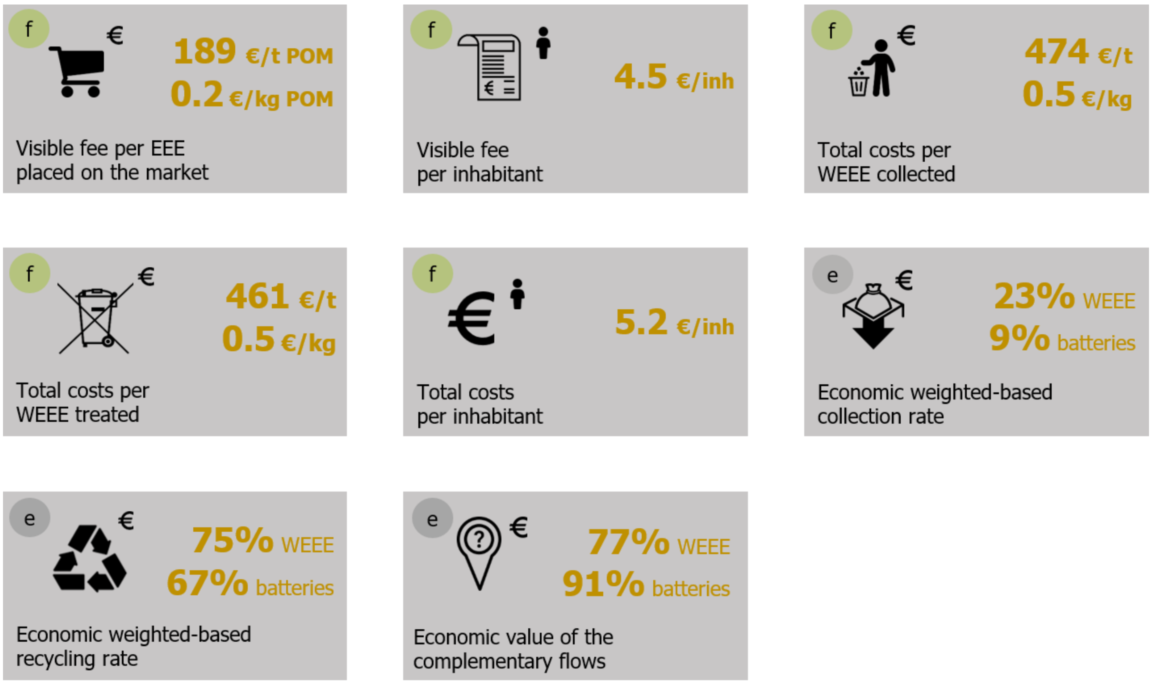


Figure 8-6. Dashboard of economic indicators.

By adding economic value to the current performance assessment scope, the interest of collecting and recycling elements found in small quantities, but with a high market value (e.g. gold and palladium), is highlighted. This is illustrated by Figure 8-7.

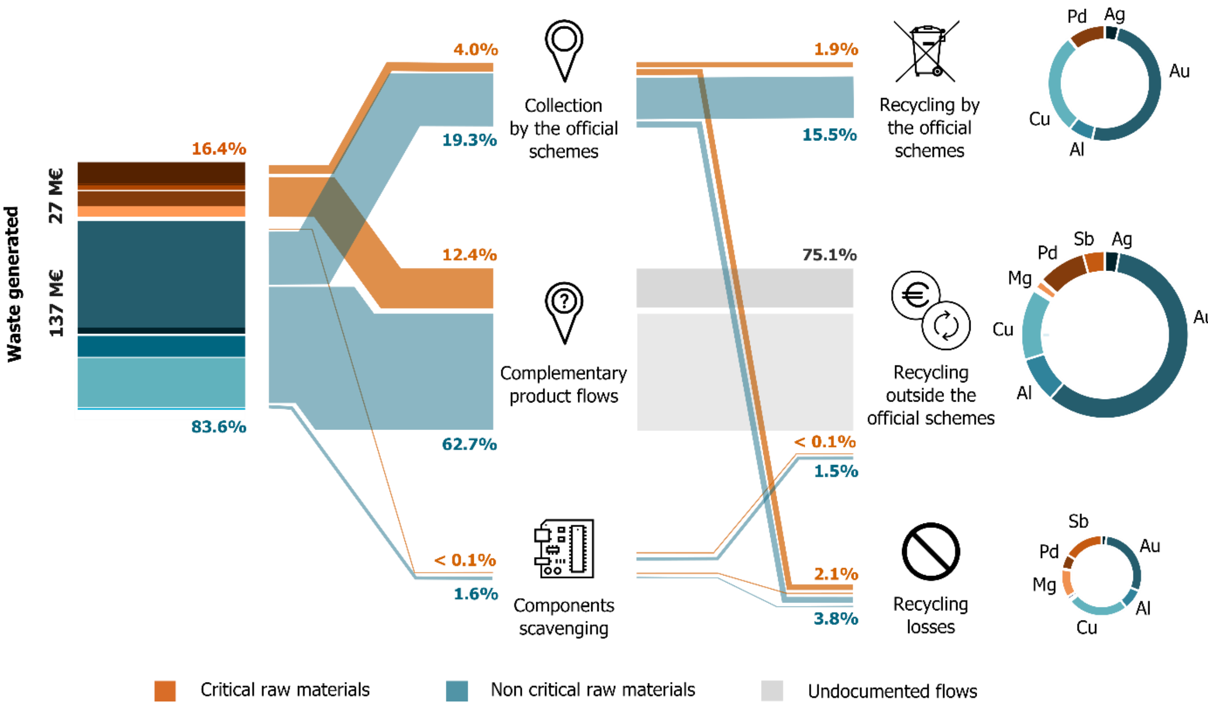


Figure 8-7. WEEE screens flows: target materials generated, collected and recycled (economic-based approach).

The intrinsic value of complementary flows represents more than four times the size, in euros, of target elements recycled by the official schemes. Even if, potentially, some of the diverted screens are reused or stored longer than the lifetime model predicted (section 4.2.1), it is interesting to estimate the economic value of these flows to stress their potential and the need to improve collection. Gold accounts for about 60% of the intrinsic value of complementary flows. It is present mostly in PCBs of laptops and tablets (56%) that still have a low collection rate in the official schemes.

Concerning the elements recycled by the official schemes, a difference between the share of CRM and non-CRM based on the intrinsic value (Figure 8-7) and based on the weight (Figure 8-3) can be noticed. From a weight-based approach, target CRM accounts for 0,3% of elements recycled (considering it as the ratio between recycled and total generated), while from an economic approach, it accounts for 1.9%. The recycled CRMs with the highest intrinsic value are palladium and cobalt. In 2018, indium, a CRM also targeted in this study, had higher sale prices than cobalt (ton/€), but as stressed in **Chapter 4**, it has limited recycling on an industrial scale.

### Criticality assessment

Given the potential of WEEE as a supplier of secondary raw materials, we explored the combination of criticality assessment and metrics to evaluate the recovery efficiency of materials. However, before proposing an indicator, we looked at the concepts related to this approach (section 7.2), and more specifically, in the criticality assessment method developed by the European Commission (EC) (section 7.3).

Inspired by previous studies identified in the literature review (Mohamed Sultan et al., 2017; Nelen et al., 2014; Van Eygen et al., 2016), we suggest three indicators to be used b-y the compliance schemes to report the WEEE chain performance from the outlook of criticality (Figure 8-8).


Figure 8-8. Dashboard of criticality indicators.

Indicators "Criticality weighted-based collection rate" and "Criticality weighted-based recycling rate" present a complementary result to the weight-based metric. Our approach considers only the supply risk according to the EC methodology, together with the market price of the elements. Moreover, since this indicator aims to capture the WEEE chain capacity to supply secondary raw materials for the economy, we include their importance within the EEE sector.

Similar to other indicators proposed in this thesis, these indicators aim to present a complementary metric to the current weight-based approach. Figure 8-9 summarizes part of the results obtained with the criticality approach suggested in this thesis.

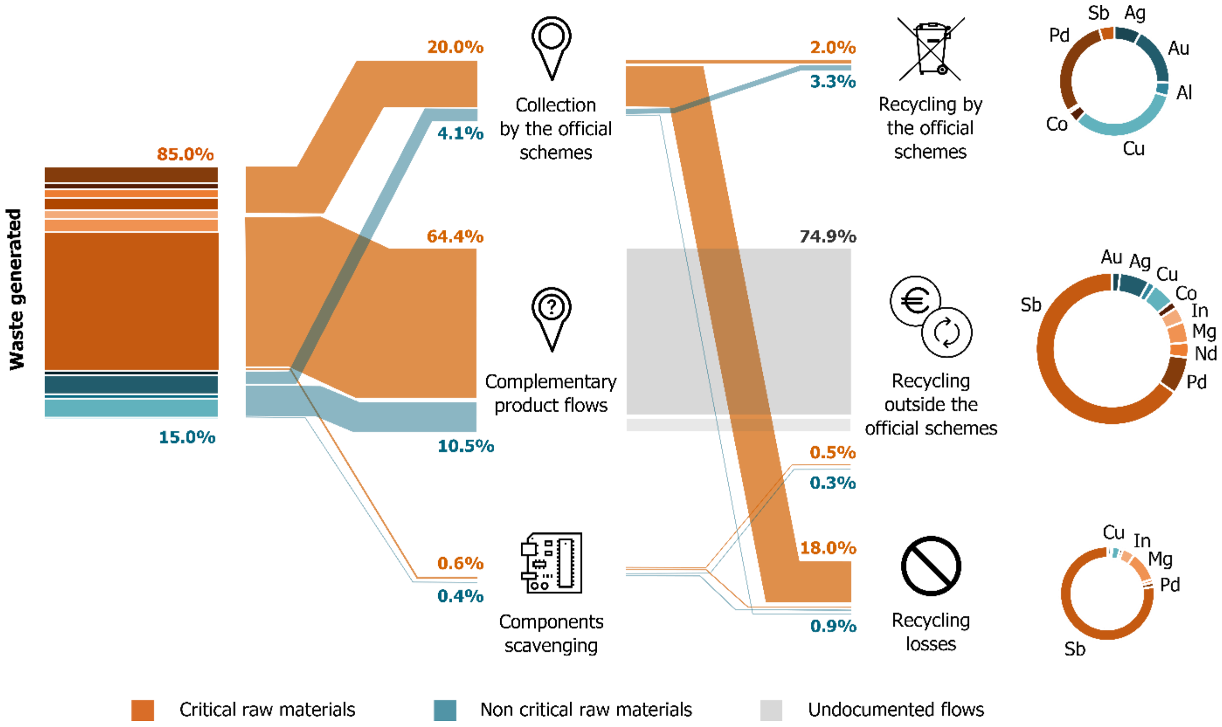


Figure 8-9. WEEE screens flows: target materials generated, collected and recycled (criticality weighted-based approach).

We identified from all the complementary approaches to the weight-based collection rate presented in this thesis, that when other parameters are considered in addition to the element weight, the overall result (including CRM and non-CRM) decreases. Moreover, within the criticality approach, other elements present in minimal concentrations in screens, such as neodymium and cobalt, have greater importance. The criticality weighted-based results are mainly influenced by antimony (Sb), magnesium (Mg), and palladium (Pd). Among the elements recycled by the official flows, the most significant from the criticality approach are copper (non-CRM) and palladium (both precious and a critical material).

Lastly, the indicator "Potential contribution for apparent consumption" estimates the contribution of complementary flows and recycling losses to the apparent consumption of materials in the EU. We estimated that some of the CRMs targeted in our study - focused only on screens - could supply more than 20% of the current raw material demand, highlighting the importance of e-waste as a source of urban mining.

### 8.1.4. Towards a more sustainable and circular WEEE chain

The last research question regarding the possibility of proposing improvements towards a more sustainable and circular (W)EEE chain based on results of multi-dimensional indicators, is answered by the combination of the results presented in **Chapter 4** to **7**. As stressed by Haupt and Hellweg (2019): "what gets measured gets managed". Thus, we believe that more detailed monitoring of e-waste flows will improve the recovery of target elements in following years, as well as improvement of the current collection and treatment management.

The individual results of the case study for each indicator allow a more detailed overview of the performance of the e-waste chain: from macro to microscopic level. For example, when quantifying the e-waste flows in **Chapter 4**, we identified the main difficulties in improving collection and recycling for the target elements. The assessment of recycling and reuse rates separately can boost new solutions to increase the reuse chain. From the environmental outlook, in **Chapter 5**, we identified in a group of target elements, those that could contribute most to increasing avoided impacts by treatment in official schemes. The set of economic indicators in **Chapter 6** provides more precise information on the expenses and revenues of the WEEE chain and an estimation of the economic potential in euros of e-waste flows collected, recycled, and dissipated in complementary flows. This data can motivate both the collection and development of more accurate pre-processing technologies to reduce losses of elements present in small quantities, but with high market value. The results of the criticality indicators in **Chapter 7** allow an overview of the collection and recycling rate based on criticality and the potential of the e-waste chain to reduce the demand for raw materials.

The time series of WEEE chain performance presented for some of the parameters and indicators used in this work (e.g. Figure 4-8 that presents data on screens placed on the market from 2010 to 2018) allow the identification of changes in the products, and consequently on the composition of the elements generated, collected and recycled by the official schemes over time. Monitoring this information can support the development of WEEE official schemes, and investments and/or research into development of new treatment processes adapted to estimated future e-waste flows.

Although it may be challenging for compliance schemes and policy-makers to continuously monitor and interpret results for 22 indicators for different WEEE categories, we do not believe is feasible to suggest either an overall single score indicator (index) calculated by weighting and summing all indicators, or for each type of indicator (technical, economic, environmental and criticality). Our reasoning is grounded in four arguments:

- The indicators assess different aspects of the WEEE chain treatment, and sometimes have a different assessment scope, which makes aggregation difficult;
- We believe that important data would be lost. As presented in section 2.2.2, in the EU, we currently already have three main indicators: collection rate (for all WEEE categories), recycling and preparation for reuse rate (for each WEEE category), and recovery rate (for each WEEE category). One of the main limitations of these indicators is the absence of a macroscopic and general overview of the WEEE chain performance;
- As presented in section 3.2, the compliance schemes and ADEME are used to monitor a significant number of indicators;
- Lastly, the criticality weighted-based collection and recycling rate (indicators 20 and 21) are already, in some ways, an aggregation of different parameters: weight, economics,

and supply risk. Nevertheless, as presented in section 7.6.1, it is interesting to monitor this indicator in comparison to weight-based and economic-weighted based metrics.

Some of the indicators suggested in this work propose monitoring of different target elements, which may also be challenging for the compliance schemes and policy-makers to monitor. A way to increase feasibility is to reduce the scope of the indicators to only a few selected (critical) raw materials per WEEE category. We recommend that the selection of target elements be done after a first screening of the composition (for example with data of the Urban Mine Platform) in order to identify elements relevant for each WEEE category considering that e-waste is very heterogeneous.

## 8.2. New set of indicators to assess the performance of WEEE schemes

### 8.2.1. Main parameters

This work proposes an extension of the current indicators to assess WEEE schemes' performance. Our method is based on five main parameters summarized below. As illustrated in Figure 4-1 (**Chapter 4**), the scope of the indicators is variable and may range from the waste generated to the collection and treatment by the official schemes, as well as the parallel flows which divert secondary resources from the official schemes.

#### E-waste flow tracking

The starting point of our approach is the mapping of e-waste flows from waste generation until element recycling. This approach is based on Material Flow Analysis (MFA) and involves its basic steps: (1) determination of the system model, including processes and materials, (2) calculation of material flows, and (3) interpretation of results (Clarke et al., 2019).

The e-waste generated is quantified with the WEEE Calculation Tool (section 4.2.1), that considers the amount of EEE placed on the market (POM) in the preceding years, and the corresponding product life spans (Baldé et al., 2017). Subsequently, the e-waste collected is quantified according to data from the official schemes (section 4.2.2). The complementary flows (including both scavenging of products and components) can be calculated as the difference between these two flows (section 4.2.3).

Finally, the e-waste treatment is estimated based on data of the current e-waste handling by the official schemes. Seeing that recycling is nowadays the most common treatment performed in France, it is the major focus of this study, but our mapping includes the other treatment paths (e.g. reuse, energy recovery, etc.).

#### Composition

The compliance schemes and ADEME monitor collection per WEEE category, but our approach includes not only the macroscopic level (categories), but also the meso (products) and microscopic levels (components, materials and elements). Therefore, in the case study (section 4.3), we complemented the WEEE chain data with ProSUM data on the *quality* of e-waste collected.

**Recycling**

As previously mentioned, this thesis focuses on recycling (section 4.4). Nevertheless, it goes beyond the current end-of-waste status and includes pre-processing and end-processing efficiency. The information required to calculate the recycling efficiency is detailed in section 4.6.3 with the case study on screens.

**Avoided impacts**

The avoided impact is an approach used in Life Cycle Assessment (LCA) in the context of allocating environmental burden in the presence of recycling, reuse, and energy regeneration. The environmental impacts of collecting, treating and recycling WEEE are most closely linked to avoiding new material production due to recycling (Huisman, 2003; Seyring et al., 2015). Following this approach, the environmental indicators proposed in this thesis are based on the avoided impacts of elements treated by the French official schemes. The indicator is based on LCA results and data of the ESR database.

**Market (intrinsic) value**

The market value of certain components and elements in WEEE is one of the main motivations for its treatment. Thus, we suggest assessing the value of e-waste collected, recycled and complementary flows as a new approach to measure WEEE chain economic performance (see section 6.3.2 and Figure 6-6). This parameter is also used by the criticality indicators.

**Criticality**

A material is considered as critical for the EU if it has high economic importance and its supply is vulnerable to disruption (Blengini et al., 2017). Aiming to consider the criticality of materials as a relevant aspect in the recovery of secondary materials, we suggest considering the supply risk indicator according to the EC and the share of the elements within the electronic sector (section 7.5). Vulnerability to supply disruption means there is a high risk that the material supply is likely to be inadequate to meet EU industry demand. In turn, the share within the sector shows inform, at the same time, the electronic sector demand for materials, and the grade and size of secondary raw materials within the e-waste. This approach is complementary to the environmental and economic metrics that attempt to provide alternative metrics to assess the current weight-based collection and treatment performance.

### 8.2.2. Analysis of the indicators' set

In this section, we present an analysis of the indicators summarized in Table 8-1 according to the different criteria other than the classification per type of indicator.

We identified four groups of indicators according to their readiness of adoption based on the data required:

1. Immediate adoption because the parameters necessary to calculate the indicators are currently monitored – indicators "EEE placed on the market per inhabitant", "Rate of departments complying with the WEEE collection target", "Rate of plastics that may contain brominated flame retardants in WEEE collected", "Reuse rate by the official schemes per WEEE category" and "Recycling rate by the official schemes per WEEE category".

2. Adoption with limitations by some of the WEEE categories, elements and/or to communicate an overall result of the WEEE chain – indicators "Visible fee per EEE placed on the market", "Visible fee per inhabitant", "Total costs per WEEE collected", "Total costs per WEEE treated" and "Total costs per inhabitant".

3. Medium-term adoption because, beside requiring the adoption of waste generated calculation with the WEEE Calculation Tool, it also requires better monitoring of e-waste collection per type of product (UNU-keys) – indicators "Collection rate based on the WEEE generated approach" and "Treatment rate based on WEEE generated approach".

4. Medium term adoption because required data on the composition of the products is not currently monitored – indicators "Collection rate per target element", "Recycling rate per target element", "Environmentally weighted-based treatment rate", "Avoided environmental impact opportunity", "Economic weighted-based collection rate", "Economic weighted-based recycling rate", "Economic value of the complementary flows", "Criticality weighted-based collection rate", "Criticality weighted-based recycling rate" and "Potential contribution for apparent consumption".

Regarding the potential use of the indicators we identified three groups that regroup indicators of all types (technical, economic, environmental and criticality), based on the level of assessment:

1. Macroscopic efficiency of the system: it presents an overview of the WEEE chain performance – indicators "EEE placed on the market per inhabitant", "Collection rate based on the WEEE generated approach", "Rate of departments complying with the WEEE collection target", "Reuse rate by the official schemes per WEEE category", "Recycling rate by the official schemes per WEEE category" and "Treatment rate based on WEEE generated approach", "Visible fee per EEE placed on the market", "Visible fee per inhabitant", "Total costs per WEEE collected", "Total costs per WEEE treated" and "Total costs per inhabitant".

2. Mesoscopic and microscopic efficiency of the system: this presents more detailed information regarding materials, components and elements – indicators "Collection rate per target element", "Rate of plastics that may contain brominated flame retardants in WEEE collected", "Recycling rate per target element", "Environmentally weighted-based treatment rate", "Economic weighted-based collection rate", " Economic weighted-based recycling rate", "Criticality weighted-based collection rate" and "Criticality weighted-based recycling rate".

3. Benefits of e-waste treatment: these assess the potential benefits of WEEE chain – indicators "Avoided environmental impact opportunity", "Economic value of the complementary flows" and "Potential contribution for apparent consumption".

Lastly, we identified that some indicators, besides providing data on the progress of the WEEE chain performance, can also be used in decision making and planning of future strategies to develop the treatment paths and boost recovery of target elements. These indicators represent slightly more than 50% of the dashboard: indicators "EEE placed on the market per inhabitant", "Collection rate per target element", "Rate of plastics that may contain brominated flame retardants in WEEE collected", "Recycling rate per target element", "Environmentally weighted-based treatment rate", "Avoided environmental impact opportunity", "Visible fee per EEE placed on the market", "Economic weighted-based collection rate", "Economic weighted-based recycling rate", "Economic value of the complementary flows", "Criticality weighted-based collection rate", "Criticality weighted-based recycling rate" and "Potential contribution for apparent consumption".

### 8.2.3. Data availability and robustness

The feasibility of compliance schemes and policy-makers running the indicators suggested in this work on a regular basis depends on the availability of data to calculate them. Consequently, it is necessary to define a strategy to solve problems of data availability.

For the indicator that requires data on waste generated, we suggest that the French WEEE official chain adopts the WEEE Calculation Tool. The adoption of this tool by France and the other Member States will allow regular monitoring of collection rates based on the waste generated. Considering that the parameters are duly detailed in the tool, it would be possible to develop an interface between this tool and the current databases used by ADEME and compliance schemes to calculate the waste generated, based on updated data on the EEE placed on the market in France.

In France, the compliance schemes perform annual characterization campaigns to quantify the amount of WEEE collected per category, as well as the average composition per waste stream. An assessment of the share of equipment (e.g. UNU-keys) in screens collection, as well as the components scavenged, could be included in the annual characterization. As discussed by Baxter et al. (2016), future regulations should include flow monitoring via sampling and measurement. That would allow the changes in the type of equipment collected by the official schemes to be monitored, and the materials potentially available for recycling to be quantified.

Regarding (W)EEE composition, a collaborative work between producers and recyclers to find a cost-effective and efficient way of producing and sharing product and component data would support data gathering (Downes et al., 2017). However, this is not an issue that can be solved in the short term. A short/medium-term solution is to use ProSUM data. Nevertheless, for the long-term another solution is needed because the composition of the electronic devices changes rapidly with advances in technology.

It is necessary to establish compulsory mechanisms to enforce waste treatment providers to report data to the compliance schemes on e-waste treatment efficiency (sorting and final recycling operation), e.g. through requirements in the national specifications included in the Ministerial Orders for household and professional WEEE. To achieve materials circularity, the WEEE chain needs to be more transparent, and this includes improving WEEE flows monitoring.

In order to determine apparent consumption in France, we used data on EU consumption quantified in the criticality study of the EC (Deloitte Sustainability et al., 2017a, 2017b), together with an average of the overall percentage of metals consumed in France according to Eurostat statistics (EUROSTAT, 2019). We suggest using more accurate data on consumption in France estimated by the Statistical Studies Department of the General Commission for Sustainable Development (Calatayud and Mohkam, 2018). We did not have access to this data for the case study, but we believe that the WEEE official chain can obtain this data from the Ministry of the Environment.

As stressed in section 5.4.2, ESR developed a Life Cycle Inventory (LCI) database for End-of-Life (EOL) of e-waste in France. It was a pioneer project in the field of EOL treatment, and since ESR accounts for around 75% market share of compliance schemes, it is representative of the on-the-ground treatment. So far, ESR has decided not to make public the individual reports with details on the data used and further assumptions of the modeling. Additionally, the datasets available in this database only cover a few elements and materials, and only for some of the products in WEEE categories (e.g. CRT is not included). We suggest expanding the current ESR database (eventually with data from other compliance schemes), with more transparency on the scenarios and methodological choices.

The cost-effectiveness indicators proposed in section 6.3.2 cannot be assessed per category because, nowadays, the compliance schemes do not share with ADEME the revenues and cost data per category. We believe it is important to report the results per category, but we understand that the compliance schemes do not want to disclose sensitive data. Thus, we suggest the public reporting of the average results of the compliance schemes by ADEME, and not the individual financial performance of these organizations.

The economic benefits indicators proposed in section 6.3.2 require data on the market prices of materials and elements. This data can be obtained from different sources, although most of them are not free of charge.

Lastly, the two parameters are related to the criticality indicators: supply risk and share of the EEE sector. This data can be obtained in EC publications (RMIS website and Deloitte Sustainability et al., 2017a, 2017b). The EC is continuously working to keep it up to date at least every 3 years (European Commission, 2019). Nevertheless, the data is limited to the candidate materials on which the criticality assessment was carried out.

## 8.3. Closing remarks

Together with previous studies published in the literature (Ardente et al., 2014; Haupt et al., 2016; Huisman, 2003; Nelen et al., 2014; Parajuly et al., 2017; Tansel, 2017; Van Eygen et al., 2016; Wang, 2014), this work can support the development of indicators evaluating different environmental, economic and criticality priorities related to raw materials. Future policies could adopt indicators beyond the weight-based approach and better practices in WEEE management in order to improve the e-waste tracking and recovery of (critical) raw materials. By doing so, more targeted WEEE management activities have the potential to extend their scope from waste and hazardous substances management to enhance the supply of quality secondary (critical) raw materials.

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## Chapitre 9 | De la gestion des déchets à l'approvisionnement de matières secondaires

*"Brasil, meu dengo / A Mangueira chegou*
*Com versos que o livro apagou*
*Desde 1500 tem mais invasão do que descobrimento*
*Tem sangue retinto pisado atrás do herói emoldurado*
*Mulheres, tamoios, mulatos*
*Eu quero um país que não está no retrato.*[23] *"*
História para ninar gente grande, Samba-enredo Mangueira (2019).

### 9.1. Contexte général

La croissance de la population et le développement des nouvelles technologies vont de pair avec une augmentation de la consommation d'énergie et de matières premières qui entraîne des coproduits tels que des déchets et des émissions (Schandl et al., 2016). À l'échelle humaine, les ressources naturelles sont limitées et majoritairement non renouvelables aussi pour répondre aux besoins croissants de matières, il est nécessaire de trouver des solutions pour assurer l'approvisionnement futur de ces matières (Hofmann et al., 2018). Dans l'Union Européenne (UE), l'approvisionnement sûr et durable en matières premières revêt une importance cruciale en raison de l'importance des matières premières pour l'industrie européenne et de sa dépendance vis-à-vis des importations (Løvik et al., 2018).

Dans ce contexte, l'économie circulaire est considérée comme un élément vital pour parvenir à une meilleure gestion des ressources en prolongeant et en fermant les cycles de matériaux, tout en réduisant les déchets (Zeller et al., 2019). Ce nouveau modèle économique est une alternative à l'économie linéaire « extraire, fabriquer, jeter » (Parajuly and Wenzel, 2017).

Les concepts d'économie circulaire sont promus au niveau européen par le biais du plan d'action de l'UE pour l'économie circulaire (European Commission, 2015). Cette transition est perçue comme une occasion de réduire la dépendance à l'égard des ressources et le volume des déchets, ainsi que de favoriser l'emploi et la croissance local (Ellen MacArthur Foundation, 2015). En France, la loi sur la transition énergétique pour une croissance verte, promulguée en 2015, a fixé des objectifs et des actions visant à encourager la circularité, parmi lesquels la réduction de moitié de la quantité de déchets mis en décharge et le recyclage de 60 % des déchets d'ici 2025, ainsi que des incitations à la conception de produits donnant la priorité aux matières premières secondaires (JORF, 2015a). En 2018, le gouvernement français a publié la feuille de route pour l'économie circulaire. Avec l'amélioration de la production et de la consommation, la gestion des déchets est présentée comme une clé importante pour boucler la boucle des matériaux. Plusieurs mesures et objectifs ont été définis pour favoriser l'écoconception, la réutilisation et le recyclage (Ministry for an Ecological and Solidary Transition and Ministry for the Economy and Finance, 2018).

[23] Brésil, mon amour. La Mangueira (une école de samba de Rio) est arrivée avec des versets effacés des livres. Depuis 1500, il y a plus d'invasions que de découvertes. Il a du sang nègre derrière le héros encadré. Femmes, tamoios, mulâtres. Je veux un pays qui n'est pas sur la photo.

Boucler la boucle des matériaux est au cœur du modèle d'économie circulaire (voir Figure 9.1), en soulignant que les ressources contenues dans les déchets doivent être récupérées et réutilisées autant que possible (Cossu and Williams, 2015). Le processus de récupération des matériaux et des composants des déchets est connu sous le nom de mine urbaine (Gutberlet, 2015). Les Déchets d'Équipements Électriques et Électroniques (DEEE) font partie des principaux flux de mines urbaines en raison de leur composition variée et de la forte augmentation de leurs volumes (Koutamanis et al., 2018 ; Zeng et al., 2018). Les Véhicules Hors D'usage (VHU) et les déchets du bâtiment et des travaux publics (BTP) sont aussi des flux importants mis en évidence dans la littérature. Outre l'extraction primaire, l'exploitation des mines urbaines est une source alternative de matière (Tunsu et al., 2015).


Figure 9.1. Perspective de l'économie circulaire : des déchets aux ressources secondaires.

Dans ce contexte, les acteurs du traitement des déchets jouent un rôle crucial dans la mise en place de l'économie circulaire par la gestion et la réorientation des flux de déchets vers des traitements visant à prolonger le cycle de vie des produits et des matériaux. Néanmoins, une transition vers une économie circulaire exige d'aller au-delà du traitement et du recyclage des déchets et de s'orienter vers le développement durable et l'utilisation de matériaux et de produits grâce à des stratégies de circularité (p. ex. prolonger la durée de vie des produits et augmenter le recyclage à haute teneur) (Potting et al., 2017).

### 9.1.1. Les déchets : un changement de paradigme

Lorsque les premières discussions sur la mise en œuvre des politiques de gestion des déchets ont commencé, l'accent était mis sur la lutte contre la pollution afin de réduire les risques environnementaux et sanitaires. En Europe, l'élaboration de politiques en matière de déchets a commencé dans les années 70 avec la première Directive 75/442/CEE sur les déchets (European Commission, 1975). Après sa publication, la directive a été modifiée à plusieurs reprises afin d'améliorer son efficacité et de tenir compte des questions les plus récentes liées à la gestion des déchets (Más, 2016).

Dans les années 90, le concept de hiérarchie des déchets a été introduit. Comme le montre la Figure 9.2, la prévention est considérée comme l'option la plus souhaitable, suivie de la préparation des déchets en vue de leur réutilisation, de leur recyclage et de leur valorisation (p. ex., récupération d'énergie). La mise en décharge occupe le bas de la hiérarchie, cette option étant considérée comme la moins souhaitable pour l'environnement (Manfredi and Goralczyk, 2013). En 2006, la Directive 75/442/CEE a été codifiée en un nouveau texte qui a remplacé toutes les versions précédentes et, en 2008, la directive (révisée) sur les déchets est entrée en vigueur.

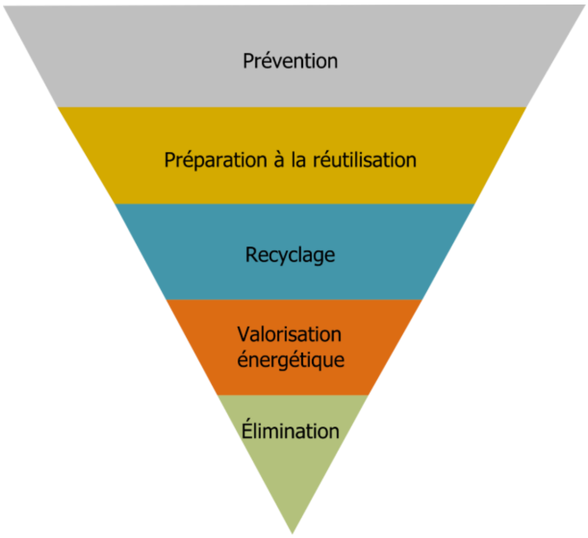


Figure 9.2. La hiérarchie des déchets.
Source: Adapté de European Commission, 2008.

Dans le cadre de sa stratégie globale de gestion des déchets, la Commission européenne (CE) a défini plusieurs flux de déchets prioritaires (par exemple, les déchets d'emballages, les VHU, les piles et les DEEE). En conséquence, une série de directives et d'actes juridiques s'appliquant spécifiquement à ces flux de déchets ont été publiés (European Parliament, 2015a).

Aujourd'hui, la politique de gestion des déchets de l'UE repose d'une part sur des stratégies politiques globales de réduction des déchets et d'optimisation de la fin de vie des produits. Et d'autre part, sur des directives et des actes juridiques s'appliquant à des flux de déchets spécifiques. Deux principes s'appliquent à tous les flux de déchets : la « hiérarchie des déchets » mentionnée précédemment et le « principe du pollueur-payeur » (ceux qui produisent la pollution doivent supporter les coûts de gestion et de traitement). L'obligation pour les producteurs d'assumer la responsabilité opérationnelle ou financière de la phase de fin de vie de leurs produits est appelée Responsabilité Élargie du Producteur (REP), et elle est utilisée dans plusieurs filières de déchets dans l'UE (European Parliament, 2015b).

En France, le principe de la REP existe en droit depuis 1975 et est codifié dans l'article L. 541-10 du Code de l'environnement. Le premier système de REP a été mis en place sur les emballages ménagers en 1992, et il existe aujourd'hui une vingtaine de filières REP pour différents produits, à différents stades de développement (ADEME, 2017). Certaines filières étant volontaires, uniquement françaises ou européennes.

Chaque filière REP a sa propre organisation et ses propres caractéristiques, néanmoins il existe des principes récurrents pour mettre en œuvre la responsabilité élargie du producteur, comme l'établissement d'un objectif minimal pour la réutilisation, le recyclage et la récupération, les exigences réglementaires en matière de financement et l'application d'une redevance lorsque les produits sont mis sur le marché. Les principaux acteurs de la chaîne REP en France, et leurs rôles, sont détaillés ci-dessous (ADEME, 2015) :

**Les pouvoirs publics**

Ils définissent le cadre réglementaire (objectifs, répartition des responsabilités entre les acteurs, agréments, etc.), s'assurent de la bonne mise en œuvre du dispositif (observation de la filière : quantités mises sur le marché, quantités collectées et traitées, etc.), contrôlent la conformité des actions des éco-organismes avec leur agrément et sanctionnent le cas échéant les contrevenants au dispositif. Il s'agit notamment des différents acteurs gouvernementaux qui contribuent à l'élaboration des politiques et à la mise en place du système, ainsi que de l'Agence de l'Environnement et de la Maîtrise de l'Énergie (ADEME) qui assure le pilotage et le suivi des chaînes REP.

**Producteurs**

Ils doivent participer à la gestion de la filière soit individuellement soit collectivement au travers d'éco-organismes. Il comprend les fabricants, les distributeurs et les importateurs.

**Éco-organismes**

Dans le cas d'un système collectif, les éco-organismes organisent, au nom des producteurs, la collecte et le traitement des déchets. Dans le cas d'une filière REP réglementaire, il est agréé par les pouvoirs publics sur la base d'un cahier des charges qui fixe l'ensemble de ses obligations.

**Organisme coordonnateur de la filière**

Agréé par les pouvoirs publics et à but non lucratif, il garantit des conditions de concurrence équitables et des niveaux de service uniformes pour tous les éco-organismes. Dans la filière DEEE, OCAD3E est également responsable de l'équilibrage des opérations des éco-organismes en fonction de leur part de marché. Une autre mission est de signer des accords contractuels avec les collectivités locales et de leur verser une compensation financière pour la collecte.

**Consommateurs (détenteurs des déchets)**

Ils doivent trier leurs déchets et les faire prendre en charge dans le cadre d'un dispositif adapté. Elle comprend les détenteurs de déchets ménagers et professionnels.

**Distributeurs**

Ils doivent informer le consommateur des conditions de bonne gestion des produits une fois usagés et ont l'obligation de reprendre gratuitement les produits usagés sans obligation d'achat ou lors de l'achat d'un produit neuf équivalent.

**Opérateurs de traitement des déchets**

Ils travaillent sous contrat pour fournir des services de gestion des déchets, y compris la collecte, le transport, le stockage, la préparation en vue de la réutilisation, la récupération des matériaux et l'élimination finale.

En conséquence de l'évolution des exigences légales et des technologies, la gestion des déchets a évolué progressivement dans l'UE (WEEElogic, 2019). Les récents efforts pour passer à une économie circulaire dans l'UE ont souligné l'importance de transformer les déchets en ressources (European Parliament, 2014). Pour atteindre cet objectif, il est essentiel d'améliorer les performances des filières REP.

### 9.1.2. (D)EEE : un flux complexe et intéressant

Les Équipements Électriques et Électroniques (EEE) sont des produits multi matériaux complexes dans lesquels une variété d'éléments sont reliés entre eux par des connexions constructives et chimiques (Van Schaik and Reuter, 2014). Si, dans les années 1990, un téléphone mobile utilisait une douzaine de métaux majeurs, les smartphones modernes se composent d'une soixantaine d'éléments, dont divers alliages métalliques (Nguyen et al., 2018).

Chaque année, des millions de tonnes d'EEE sont mis sur le marché dans le monde, ce qui entraîne une production massive de déchets. En 2018 en France, 28,8 kg d'EEE par habitant ont été mis sur le marché, soit une augmentation de 15 % par rapport à 2012. La quantité de déchets électroniques augmente au niveau mondial, avec une croissance annuelle d'environ 5 % (Ibanescu et al., 2018). En 2016, 44,7 millions de tonnes (Mt) de DEEE ont été produites dans le monde. L'Europe est le deuxième continent producteur de déchets électroniques (12,3 Mt), seule l'Asie en produit davantage (18,2 Mt) (Baldé et al., 2017).

En plus d'être le flux de déchets dont la croissance est la plus rapide, les DEEE sont particulièrement complexes en raison de leur composition matérielle (Vadoudi et al., 2015). Bien qu'il contienne certains matériaux de grande valeur (tels que l'or, le cuivre, les éléments des terres rares et le palladium), il comprend également certains matériaux toxiques (p. ex. le mercure, le plomb et les retardateurs de flamme bromés).

Le périmètre réglementaire des déchets électroniques varie considérablement d'un pays à l'autre, tout comme le niveau de développement des systèmes de gestion des DEEE (Tansel, 2017). Dans l'UE, la gestion des DEEE est régie par la Directive DEEE (Directive 2012/19/UE) (European Commission, 2012). En France, elle a été transposée en droit français, en même temps que la Directive RoHS (Directive 2002/95/CE) (principalement par décret 2005-829) (JORF, 2005), pour réglementer la composition des EEE et gérer les DEEE. Elle a ensuite été complétée par d'autres textes législatifs et par des ajouts au code français de l'environnement.

La Directive DEEE vise à prévenir la production de déchets électroniques, ainsi qu'à améliorer les performances des opérations de traitement pour la réutilisation, le recyclage et d'autres formes de valorisation (Wäger and Hischier, 2015). Elle introduit le concept de REP, ce qui signifie que les producteurs d'EEE (fabricants et importateurs) sont responsables de la fin de vie des appareils qu'ils ont mis sur le marché. Pour faciliter l'organisation des REP collective les éco-organismes ont vu le jour.

L'Europe est le continent où le taux de collecte est le plus élevé, notamment grâce aux Directives, règlements et systèmes mis en place pour les DEEE depuis 1990. Néanmoins, les performances varient considérablement d'un état membre à l'autre. En 2016, par exemple, le taux de collecte était de 67 % en Suède, 45 % en France et 10 % à Malte (Eurostat, 2018).

Ces variations s'expliquent par le niveau de vie, le stade de développement des filières officielles, ainsi que le manque d'harmonisation de la méthodologie de collecte des données pour calculer les taux de performance entre les États membres de l'UE (Baldé et al., 2017 ; Salhofer et al., 2016).

### 9.1.3. Principaux vecteurs du traitement des déchets électroniques

En raison de la composition des déchets électroniques, leur traitement est motivé par trois principaux facteurs : économique, environnemental et de santé publique. D'un point de vue économique, les DEEE contiennent des métaux précieux (or, argent et palladium) et d'autres métaux de valeur (cuivre, aluminium et fer) (Ibanescu et al., 2018).

Du point de vue des ressources, le recyclage des déchets électroniques, à une certaine échelle, diminue la pression causée sur l'environnement par l'extraction des matières premières des gisements minéraux (Meshram et al., 2019). Du point de vue de la durabilité, nous devrions rationaliser l'utilisation des matières premières en tenant compte des besoins des générations futures. Le recyclage peut également contribuer à la sécurité de l'approvisionnement en matières premières et à l'évolution vers une économie plus circulaire (Mathieux et al., 2017). Néanmoins, du point de vue de la santé publique, ce type de déchets contient également des substances dangereuses qui présentent des risques pour la santé humaine et l'environnement et qui nécessitent un traitement spécifique (p. ex. mercure et plomb) (Lixandru et al., 2017). On sait que la chaîne des DEEE a des incidences sur l'environnement (par exemple, émissions liées au transport, émissions de substances, consommation d'énergie et de ressources pour la réutilisation et le recyclage, etc.). Toutefois, on s'attend à ce que les impacts soient inférieurs à ceux de la production de Matières Premières Recyclées (MPR).

Le cuivre et les métaux précieux (par exemple l'or, le palladium et l'argent) constituent le principal moteur économique du recyclage des déchets électroniques, étant donné leur concentration significative par rapport aux minerais (Awasthi et al., 2018 ; Kumar et al., 2017). Par exemple, dans certains téléphones mobiles, l'or représente 0,04 % du poids total, soit une concentration de 400 g/t — 200 fois supérieures au minerai d'or typique (1-3 g/t), et 14 fois supérieures à celle du gisement à plus forte concentration connue (28 g/t) (Charles et al., 2017). À l'inverse, les polymères sont le plus souvent une charge financière et sont source de nombreuses problématiques de fin de vie.

En raison de son potentiel de mine urbaine, les DEEE ont été identifiés comme une activité lucrative tant dans les pays développés que dans les pays en développement (Sinha et al., 2016). Néanmoins, entre la collecte des déchets électroniques par les filières réglementaires et la production effective de MPR, il existe une chaîne complexe de processus motivés par des intérêts économiques et environnementaux (par exemple, le contrôle de substances comme le plomb et le mercure) (Valero Navazo et al., 2013). La production de déchets électroniques, combinée à l'existence de filières parallèles non officielles, entraîne des coûts de collecte élevés et une collecte modeste par les filières réglementaires. En raison de la complexité et de l'hétérogénéité des déchets électroniques, différentes techniques de dépollution, de démantèlement, de broyage, de tri et de recyclage doivent être combinées afin de récupérer les matières premières secondaires, en fonction de leur type. Ce traitement a un coût significatif lorsqu'il est réalisé dans le respect de toutes les normes environnementales et sanitaires. En conséquence, malgré la valeur intrinsèque des matières contenues dans les DEEE, la rentabilité ne compense pas toujours correctement les coûts d'investissement et d'exploitation (Cesaro et al., 2018). Certains matériaux recyclés, comme les résines polymères, peuvent être plus coûteux à traiter que les matières premières équivalentes (IEA, 2014).

### 9.1.4. Analyse de la performance de la filière DEEE

Les indicateurs sont un moyen de transformer des données complexes en informations clés pour les décideurs et les spécialistes non techniques (Cifrian et al., 2015). En tant qu'outil de pilotage de la filière de traitement des déchets, les indicateurs servent à quantifier les impacts et les avantages de la filière DEEE, ainsi qu'à suivre les progrès au fil du temps et à comparer les performances entre un ou plusieurs systèmes (Giljum et al., 2011). Ils peuvent être utilisés pour aider à la prise de décision, en plus de communiquer les résultats aux différentes parties prenantes.

Actuellement, dans l'UE, la performance de la filière DEEE est principalement évaluée au moyen d'indicateurs techniques qui visent à garantir la conformité du système aux objectifs de collecte et de valorisation fixés par la directive DEEE. Dans cette étude, les indicateurs sont classés comme techniques lorsqu'ils visent à quantifier les flux de déchets électroniques (par exemple, les déchets produits, collectés et recyclés) ou à mesurer l'efficacité d'un processus ou d'un système de déchets (par exemple, le taux de collecte et de recyclage).

La Directive DEEE établit trois indicateurs techniques pour surveiller l'efficacité du système DEEE (article 3 de la Directive DEEE) :

- Le taux de collecte
- Le taux de recyclage et de préparation à la réutilisation
- Le taux de valorisation.

Outre la Directive DEEE, d'autres indicateurs sont proposés par les parties prenantes du traitement des déchets électroniques (par exemple, les décideurs politiques, les systèmes de conformité et les associations) pour évaluer la performance du traitement des déchets électroniques dans un pays ou pour comparer la performance entre différents pays. Par exemple, le tableau de bord des matières premières publié par la CE compare les performances de la filière DEEE dans les États membres avec des indicateurs basés sur la masse des DEEE collectés (total et par ménage), recyclés et réutilisés, divisée par le nombre d'habitants (kg par habitant) (Vidal-Legaz et al., 2018).

Les indicateurs et objectifs existants sont mesurés selon une approche globale basée sur la masse, sans tenir compte des pertes de traitement, et en excluant les flux non collectés par les filières réglementaires, mais capturés par des flux complémentaires (Haupt et al., 2016 ; Parajuly et al., 2017). Les DEEE non collectés par les filières réglementaires finissent dans d'autres filières non documentées, incluant les équipements réutilisés, les déchets électroniques éliminés dans le flux de déchets municipaux, ainsi que les exportations non documentées soit comme déchets (illégaux), soit comme articles réutilisables (Bigum et al., 2013 ; Huisman et al., 2015).

En outre, la Directive DEEE ne prévoit pas d'objectifs spécifiques pour certains matériaux, en particulier ceux contenus en faibles quantités ou à l'état de traces, dont les Matières Premières Critiques (CRM) (Van Eygen et al., 2016). Il y a un manque d'indicateurs et d'objectifs opérationnels clairs et solides couvrant les aspects de durabilité liés à la gestion des DEEE (Nelen et al., 2014). En outre, compte tenu du potentiel des DEEE en tant que mine urbaine, il est nécessaire de disposer d'indicateurs permettant de suivre les performances de la chaîne des déchets électroniques en tant que fournisseur de matières premières secondaires (critiques) de qualité.

### 9.1.5. Contexte industriel de cette thèse : focus sur la filière française

La filière française DEEE professionnels est opérationnelle depuis 2005, et 2006 pour les e-déchets ménagers. Selon les données fournies par l'ADEME, en 2018, la filière DEEE a atteint l'objectif de collecte fixé par la directive DEEE (45 %). Toutefois, elle n'a pas respecté l'objectif de collecte fixé par la réglementation française (59 %) (Fangeat, 2019). Le **Chapitre 2** présente plus de détails sur les taux de l'UE et de la France, ainsi que sur la performance de la filière française DEEE.

De 2015 à 2020, trois éco-organismes sont approuvés par les pouvoirs publics pour la gestion des DEEE en France : Ecologic, ESR[24] (anciennement Eco-Systèmes et Récylum) et PV Cycle. En tant qu'éco-organisme approuvé, Ecologic, cofinanceur de ces travaux de thèse, est confronté à divers défis pour développer le secteur des DEEE, parmi lesquels :

- L'augmentation du taux de collecte, compte tenu des objectifs ambitieux fixés par la réglementation française pour les années à venir (65 % en 2019 et 2020).
- L'amélioration du taux de recyclage des matériaux contenus dans les déchets électroniques.
- Le développement du recyclage des plastiques et des matières premières critiques.
- Le respect des exigences de la législation française et des objectifs fixés par la Directive DEEE.

Les éco-organismes approuvés doivent être conformes aux cahiers des charges fournies par le Code de l'Environnement, qui présente des lignes directrices et des objectifs nationaux (annexes du décret du 2 décembre 2014 relatif à la procédure d'approbation des DEEE ménagers et du décret du 20 août 2015 relatif aux DEEE professionnels). Selon ces cahiers des charges, chaque année, les systèmes de conformité doivent rendre compte à l'ADEME et au Ministère de la Transition Écologique et Solidaire sur la base d'indicateurs relatifs à la gestion des DEEE domestiques et professionnels (JORF, 2015b, 2014). La liste des indicateurs de suivi demandés est présentée dans le cahier des charges domestiques et professionnel. Toutefois, ni la méthode ni les critères à utiliser pour calculer les indicateurs, ni leur périmètre, ne sont détaillés par les pouvoirs publics.

Par conséquent, cette recherche peut appuyer l'élaboration de futurs cahiers des charges en suggérant un ensemble d'indicateurs, y compris les méthodes de calcul, ainsi qu'une approche pour l'acquisition de données. Elle peut également améliorer les connaissances concernant les fractions de production du traitement des déchets électroniques (composition et utilisation secondaire), ainsi que l'évaluation des incidences et des avantages environnementaux des filières réglementaires.

## 9.2. Objectif et questions de recherche

L'objectif principal de cette thèse, aligné sur les besoins des partenaires de cette recherche (ADEME et Ecologic) est :

***Établir un ensemble solide d'indicateurs couvrant les aspects de développement durable (techniques, économiques, environnementaux et de criticité) liés à la collecte et au***

[24] ESR est devenu ecosystem en octobre 2019. Pendant la rédaction de cette thèse, le nom de l'éco-organisme (sur son site et dans sa base de données) était ESR. Pour plus d'informations sur le changement de nom de l'éco-organisme : https://www.actu-environnement.com/ae/news/dechets-electrique-electronique-ecosystem-energie-recylum-recyclage-34218.php4

***traitement des DEEE, afin d'améliorer la visibilité des progrès de la filière réglementaire de DEEE dans une économie circulaire.***

Dans ce contexte, la question générale de recherche de cette thèse est :

***La performance de la filière des DEEE pourrait-elle être améliorée grâce à un ensemble d'indicateurs qui renforcent son évaluation et son suivi ?***

Cette thèse de doctorat a développé des méthodes et des indicateurs pour atteindre cet objectif et répondre à la question générale de recherche. La question de recherche générale a été subdivisée en quatre questions de recherche (QR) :

***QR1. Les indicateurs actuels fournissent-ils des informations, au-delà des performances techniques, sur les progrès de la filière des DEEE ?***

Cette question est liée à la cartographie et à la classification des indicateurs actuels selon leur dimension d'analyse : technique, économique, environnementale et sociale. Une analyse documentaire des indicateurs actuellement utilisés par le secteur des DEEE, ainsi que des indicateurs publiés dans la littérature, est présentée au **Chapitre 3**.

***QR2. Quels sont les indicateurs et méthodes pertinents pour évaluer les performances techniques de la filière des DEEE ?***

Cette question porte sur l'identification des indicateurs techniques les plus appropriés pour appuyer la quantification de la performance de la collecte et du traitement, sans limiter l'évaluation à une approche globale fondée sur la masse. Les indicateurs techniques choisis dans la documentation et certains nouveaux indicateurs sont présentés au **Chapitre 4**. La faisabilité des indicateurs est évaluée par une étude de cas basée sur les flux d'écrans en France.

***QR3. Comment évaluer les performances environnementales et économiques de la filière des DEEE ?***

Cette question de recherche examine les méthodes disponibles pour évaluer les impacts et les bénéfices environnementaux et économiques de la chaîne des DEEE. Les méthodes et indicateurs identifiés sont présentés respectivement aux **Chapitres 5** et **6**. Compte tenu de l'importance de certaines matières premières pour l'économie de l'UE, une approche fondée sur la criticité comme paramètre d'évaluation de la valorisation des DEEE est examinée au **Chapitre 7**. Une étude de cas basée sur les flux d'écrans en France est présentée à la fin de chaque chapitre afin d'illustrer les résultats obtenus avec les indicateurs retenus.

***QR4. Sur la base des résultats des indicateurs multidimensionnels, est-il possible de proposer des améliorations vers une filière (D)EEE plus durable et circulaire ?***

Cette question de recherche porte sur l'utilisation d'indicateurs pour suivre les progrès au fil du temps et soutenir la prise de décision dans la filière (D)EEE. Ce sujet est présenté au **Chapitre 8**, ainsi qu'une discussion sur les paramètres essentiels et l'approche d'acquisition des données pour permettre la faisabilité des indicateurs.

Cette recherche se concentre sur le système français des DEEE et vise à améliorer le suivi des flux de déchets électroniques par les acteurs des filières réglementaires (ADEME et éco-organismes, par exemple). Ainsi, les indicateurs sont validés dans le cadre d'une étude de cas prenant en compte les données et les particularités de la filière des déchets électroniques en France. Le cadre réglementaire européen est pris en compte dans la recherche depuis que la France a transposé la Directive DEEE dans sa législation nationale, et il doit être conforme aux objectifs et programmes européens. En outre, nous pensons que les résultats de cette étude peuvent être utilisés dans d'autres états membres pour soutenir l'évolution des chaînes de DEEE, de traitement des déchets à fournisseurs de matières premières secondaires.

Cette thèse est structurée autour des quatre questions de recherche, et l'approche et les étapes pour y répondre sont présentées dans les sections suivantes.

## 9.3. Approche méthodologie

La complexité des déchets électroniques implique une approche multidisciplinaire pour assurer une compréhension holistique des défis associés à leur gestion. Outre les aspects techniques liés aux technologies et aux impacts économiques (coûts et bénéfices), le cadre réglementaire ne peut être négligé dans la gestion des déchets. Cette filière interagit également avec l'environnement ; par conséquent, les questions sociales et environnementales doivent également être prises en compte.

Le cadre réglementaire des déchets électroniques en Europe, et plus précisément en France, est le fil conducteur des discussions et de cette thèse. Ainsi, le **Chapitre 2** présente le cadre réglementaire des DEEE, ainsi que la structure et les différents acteurs de la filière DEEE en France.

Bien que l'objectif principal de cette étude soit de proposer un ensemble approprié d'indicateurs, l'évaluation des indicateurs existants constitue le point de départ de la recherche et est présentée au **Chapitre 3**. Cette évaluation comprend également la méthode utilisée pour classer les indicateurs identifiés dans l'analyse documentaire, ainsi que les limites et les lacunes des indicateurs actuels.

La quantification des flux de déchets électroniques est cruciale pour le calcul des performances de la filière. Ainsi, l'Analyse des Flux de Matières (MFA) est la principale méthodologie utilisée pour évaluer les performances techniques de la filière des DEEE présentée au **Chapitre 4**. La MFA est utilisée dans cette étude pour suivre le bilan massique des flux, des produits, des composants, des matériaux et des substances dans les déchets électroniques. Afin d'estimer les flux à toutes les étapes de fin de vie, différents domaines d'expertise sont nécessaires, tels que la gestion des déchets et le génie des procédés. Cette dernière englobe la connaissance des différents procédés d'ingénierie nécessaires à la manipulation des déchets pour obtenir les matières premières secondaires. Certains de ces processus sont décrits à la section 4.4. Ces procédés permettent de récupérer différents polymères et métaux à partir de flux hétérogènes de déchets électroniques composés de produits complexes multimatériaux.

L'Analyse du Cycle de Vie (ACV) est la principale méthodologie utilisée pour évaluer les impacts environnementaux des déchets électroniques. Une revue des études ACV, suivie d'une étude de cas appliquée au recyclage des écrans en France, est présentée au **Chapitre 5**. La portée et les choix méthodologiques de l'évaluation de l'impact et des avantages de l'ACV sur la filière des DEEE, ainsi que l'efficacité et les difficultés d'utilisation des résultats pour soutenir la prise de décision dans la gestion des DEEE, sont examinés à la section 5.6. Enfin, la section 5.7 présente des indicateurs environnementaux pour rendre compte des impacts et des avantages environnementaux de la chaîne des DEEE.

Comme mentionné précédemment, la valeur des matières premières secondaires est l'un des moteurs de la filière des DEEE. Ainsi, une évaluation du potentiel des DEEE du point de vue de la valeur intrinsèque (également appelée valeur économique) des matériaux est présentée au **Chapitre 6** (section 6.3.2). Cependant, avant que les matières premières secondaires soient prêtes à être utilisées dans un nouveau produit, plusieurs étapes de traitement doivent être mises en œuvre, ce qui implique des coûts de traitement. Du point de vue des éco-organismes, sur la base de l'analyse des coûts des étapes de fin de vie, les coûts et revenus associés au traitement des déchets électroniques en France sont convertis en indicateurs économiques dans la section 6.3.1.

La criticité des matières premières est l'un des nouveaux paramètres présentés dans cette étude pour évaluer la performance de récupération des matières issues des déchets électroniques (section 7.5). Il est basé sur l'évaluation de la criticité développée par la Commission européenne (Blengini et al., 2017). Les indicateurs développés au **Chapitre 7** et une partie des indicateurs présentés au **Chapitre 4** ont été développés au cours d'un séjour de recherche de cinq mois au Joint Research Center de la Commission Européenne.

L'ensemble des indicateurs développés dans la thèse, ainsi que les discussions sur la manière dont ils pourraient soutenir différents acteurs de la chaîne des DEEE, sont présentés au **Chapitre 8**. Cette discussion vise à soutenir Ecologic et l'ADEME dans l'élaboration des cahiers des charges qui seront publiés en 2021, tels que présentés dans la section 1.1.5.

Les informations concernant le traitement des DEEE et la disponibilité des matières premières secondaires sont utiles, par exemple, pour les décideurs politiques européens et nationaux, les éco-organismes, les recycleurs et les concepteurs. Ainsi, cette recherche est pertinente pour une large communauté de lecteurs en plus de la communauté scientifique dans le domaine et pourrait encourager de futures améliorations dans la législation DEEE ainsi que la recherche dans ce domaine.

### 9.4. Structure de la thèse

La structure de cette thèse suit les questions de recherche et l'approche décrites dans les sections précédentes. Les chapitres sont organisés en six thématiques :

- Contexte réglementaire (**Chapitre 2**) ;
- Évaluation des indicateurs actuels (**Chapitre 3**) ;
- Cartographie et caractérisation des flux de DEEE (**Chapitre 4**) ;
- Développement d'indicateurs multidimensionnels et expérimentation : techniques (**Chapitre 4**), économiques (**Chapitre 6**), environnementaux (**Chapitre 5**), et basés sur la criticité (**Chapitre 7**) ;
- Mise en place des nouveaux indicateurs et discussions (**Chapitre 8**).

Après la présentation des indicateurs aux **Chapitres 4**, **5**, **6** et **7**, une étude de cas portant sur le traitement des écrans en France est présentée pour valider les indicateurs. Cette organisation a pour but de faciliter la compréhension de la contribution de cette thèse et aussi de suivre les questions de recherche.

### 9.5. Synthèse des contributions de la thèse

L'objectif de cette thèse était d'établir un tableau de bord d'indicateurs couvrant une évaluation plus globale des performances de la filière des DEEE, incluant les aspects techniques, économiques, environnementaux et de criticité. Les résultats de ces travaux visent à améliorer la visibilité sur les progrès des systèmes officiels de DEEE dans une économie circulaire. En outre, il vise à soutenir le développement des cahiers des charges fournies par le Ministère de l'Environnement qui entreront en vigueur à partir de 2021 au cours de la prochaine période d'agrément pour les éco-organismes.

Les impacts sociaux de la filière des DEEE ne sont pas l'objet de cette thèse, mais selon les cahiers des charges actuels, les éco-organismes doivent également fournir des informations sur l'emploi et l'intégration dans la filière. Comme indiqué à la section 3.3, nous avons identifié dans la revue de la littérature quelques pistes qui peuvent être explorées davantage par les éco-organismes afin de rapporter des données sur les conditions de travail (par exemple, revenu, niveau d'éducation, égalité des chances pour les hommes et les femmes).

Les indicateurs sont présentés et validés dans les chapitres précédents avec une étude de cas portant sur les données et les particularités de la filière des déchets électroniques en France. Néanmoins, nous pensons que les mesures introduites dans cette étude peuvent être utilisées dans les autres États membres. En outre, étant donné que le traitement des déchets électroniques (flux officiels et flux complémentaires) ne se limite pas aux frontières nationales, il est important de suivre et de développer des stratégies pour améliorer la valorisation des DEEE dans le contexte de l'UE. Le suivi de l'UE est encore plus pertinent lorsque l'on considère les déchets électroniques comme une source importante de ressource secondaire.

Outre l'ensemble d'indicateurs lui-même (résumé dans le Tableau 9-1), l'approche méthodologique adoptée dans ce travail constitue également une contribution à la recherche dans ce domaine et va de pair avec les discussions présentées par d'autres auteurs.

Tableau 9-1. Ensemble d'indicateurs proposés dans cette thèse.

| Type d'indicateur | Indicateurs | Section |
|---|---|---|
| Techniques | 1. EEE mis sur le marché par habitant | 4.5.1 |
| | 2. Taux de collecte basé sur l'approche générée par les DEEE | 4.5.2 |
| | 3. Taux de collecte par élément cible | 4.5.3 |
| | 4. Taux de conformité des départements à l'objectif de collecte des DEEE | 4.5.4 |
| | 5. Taux de plastiques pouvant contenir des retardateurs de flamme bromés dans les DEEE collectés | 4.5.5 |
| | 6. Taux de réutilisation par catégorie de DEEE | 4.5.6 |
| | 7. Taux de recyclage par catégorie de DEEE | 4.5.6 |
| | 8. Taux de traitement basé sur l'approche des DEEE générés | 4.5.7 |
| | 9. Taux de recyclage par élément cible | 4.5.8 |
| Environnementaux | 10. Taux de traitement pondéré par l'impact évité | 5.7.1 |
| | 11. Opportunité d'impact environnemental évité | 5.7.2 |
| Économiques | 12. Eco-participation par EEE mis sur le marché | 6.3.1.1 |
| | 13. Eco-participation par habitant | 6.3.1.2 |
| | 14. Coût total par DEEE collecté | 6.3.1.3 |
| | 15. Coût total par DEEE traité | 6.3.1.4 |
| | 16. Coût total par habitant | 6.3.1.5 |
| | 17. Taux de collecte pondéré par l'impact économique | 6.3.2.1 |
| | 18. Taux de recyclage pondéré par l'impact économique | 6.3.2.2 |
| | 19. Valeur économique des flux complémentaires | 6.3.2.3 |
| Criticité | 20. Taux de collecte pondéré par la criticité | 7.5.1 |
| | 21. Taux de recyclage pondéré par la criticité | 7.5.2 |
| | 22. Contribution potentielle à la consommation apparente | 7.5.3 |

Les principaux paramètres retenus dans cette étude sont brièvement présentés à la section 9.6.2. Les principales constatations qui ont fondé la réponse aux questions de recherche sont résumées dans les sous-sections suivantes.

### 9.5.1. Indicateurs actuellement utilisés par les éco-organismes et les décideurs politiques

La Directive DEEE, ainsi que d'autres réglementations au niveau national et européen, a constitué une avancée significative dans la gestion des déchets électroniques. Ses indicateurs et ses objectifs (taux de collecte, de recyclage et de réutilisation et taux de valorisation - détaillés à la section 2.2.2) ont permis d'inciter les États membres à faire évoluer les filières DEEE, mais ils fournissent un aperçu limité basé sur des résultats globaux pondérés par le poids. Bien que la Directive présente une approche alternative pour le calcul du taux de collecte basée sur les déchets électroniques générés (l'approche la plus couramment utilisée est basée sur la moyenne des poids des équipements mis sur le marché au cours des trois années précédentes), elle n'est pas encore largement adoptée par les États membres.

L'approche basée sur les DEEE générés permet une meilleure évaluation des performances réelles de collecte (dont la durée de vie et le stockage des équipements), ainsi que des flux complémentaires. Comme indiqué à la section 3.2, au niveau national, les éco-organismes et l'ADEME contrôlent d'autres indicateurs que ceux imposés par la réglementation. Ces indicateurs visent à suivre les équipements mis sur le marché, la mise en place de la collecte et le traitement des déchets électroniques jusqu'à ce que le statut de fin de vie des déchets soit atteint.

Outre les indicateurs techniques, nous avons identifié quelques indicateurs économiques rapportés par les éco-organismes. Néanmoins, nous avons conclu que ces indicateurs sont en fait des données sur les revenus et les dépenses des éco-organismes. Ils ne permettent pas de comprendre les performances économiques de la filière DEEE. Pour les interpréter, il est nécessaire d'analyser d'autres données relatives aux performances de la filière (par exemple, le coût de collecte par tonne de DEEE collectés par les éco-organismes).

En outre, les indicateurs tant techniques qu'économiques sont présentés comme des résultats globaux de la filière DEEE (par exemple, le taux de collecte, le montant en euros collecté avec l'éco-participation) et, dans certains cas, par catégorie (par exemple, le taux de recyclage et de réutilisation). Un suivi en termes de qualité des déchets électroniques collectés et traités (type de produits générés et collectés, composants, matériaux et éléments récupérés) n'est actuellement pas effectué.

Selon les derniers cahiers des charges des DEEE ménagers et professionnels (JORF, 2015, 2014), les éco-organismes français doivent également présenter des indicateurs relatifs à l'emploi, l'intégration et les impacts environnementaux de la filière réglementaire. Toutefois, ni la méthode ni la portée du calcul des indicateurs ne sont détaillées par les pouvoirs publics.

### 9.5.2. Indicateurs et méthodes d'évaluation des performances techniques

Les indicateurs techniques présentés au **Chapitre 4** sont basés sur le suivi des déchets électroniques issus de la génération de DEEE (équipements mis sur le marché et leur élimination après quelques années en considérant la durée de vie) jusqu'au recyclage de certains éléments cibles. L'approche méthodologique repose sur cinq niveaux d'évaluation : flux (catégories de DEEE ou flux de déchets électroniques), produits, composants, matériaux et éléments (voir Figure 9.3). L'approche actuelle ne surveille que le niveau macroscopique, et pour améliorer la récupération des matériaux et des éléments, il est nécessaire de surveiller les flux de déchets électroniques avec un niveau de détail plus élevé.

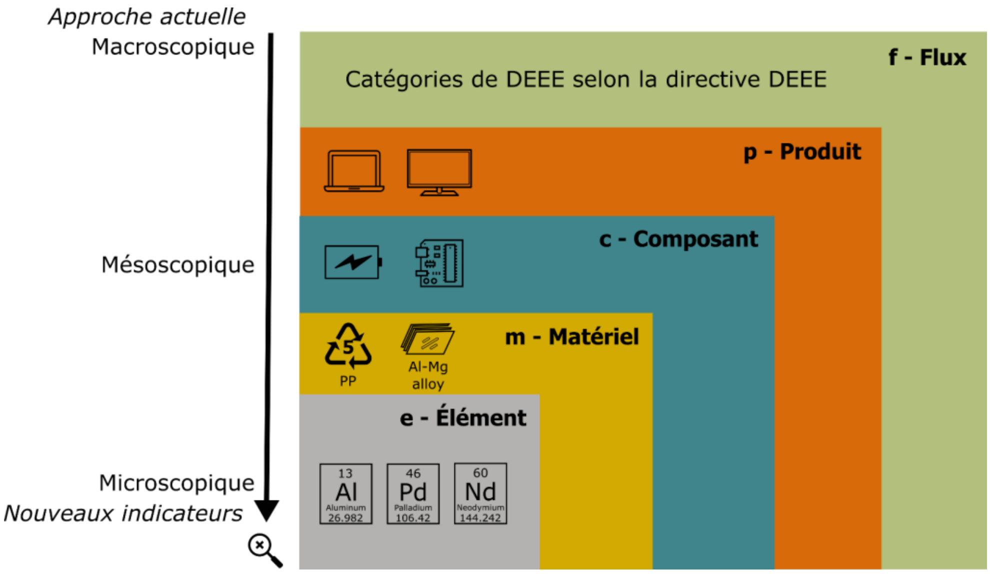


Figure 9.3. Niveaux de surveillance pour permettre une meilleure évaluation des performances de traitement.
Source : adapté de Huisman et al, 2017 ; Zeng et al, 2017

La Figure 9.4 présente les neuf indicateurs techniques, leur niveau d'évaluation et les résultats de l'étude de cas présentés à la section 4.6. Parmi les neuf indicateurs techniques proposés dans ce travail, quatre sont centrés sur l'évaluation macroscopique par catégorie (« Taux de conformité des départements à l'objectif de collecte des DEEE », « Taux de réutilisation par catégorie de DEEE », « Taux de recyclage par catégorie de DEEE » et « Taux de traitement selon l'approche DEEE générés »).

Deux indicateurs sont basés sur une évaluation microscopique par élément (« Taux de collecte par élément cible » et « Taux de recyclage par élément cible »). Pour le recyclage de l'élément, l'information sur la composition de chaque matériau (niveau microscopique), composant (niveau mésoscopique) et produit (niveau macroscopique) est obligatoire pour estimer l'efficacité du traitement. De son côté, un indicateur (« Taux de plastiques pouvant contenir des retardateurs de flamme bromés dans les DEEE collectés ») est axé sur le niveau du matériau (taux de plastiques contenant des retardateurs de flamme bromés).

Les indicateurs « EEE mis sur le marché par habitant » et « Taux de collecte basé sur l'approche DEEE générés » fournissent un résultat par catégorie, mais nous suggérons de surveiller les produits (UNU-Keys) mis sur le marché, générés et collectés afin de mieux comprendre les types actuels et futurs de dispositifs qui seront traités par les filières réglementaires. Dans le cas des écrans, nous avons observé une différence importante dans le rendement de collecte des écrans à tube cathodique (CRT) et des écrans plat (EP). En 2018, les EP représentaient 51,7 % des DEEE produits. Cependant, les écrans collectés par les dispositifs officiels sont composés à 77 % de CRT. Par conséquent, le taux de collecte des CRT et des EP est complètement différent : 62 % et 17 %, respectivement.

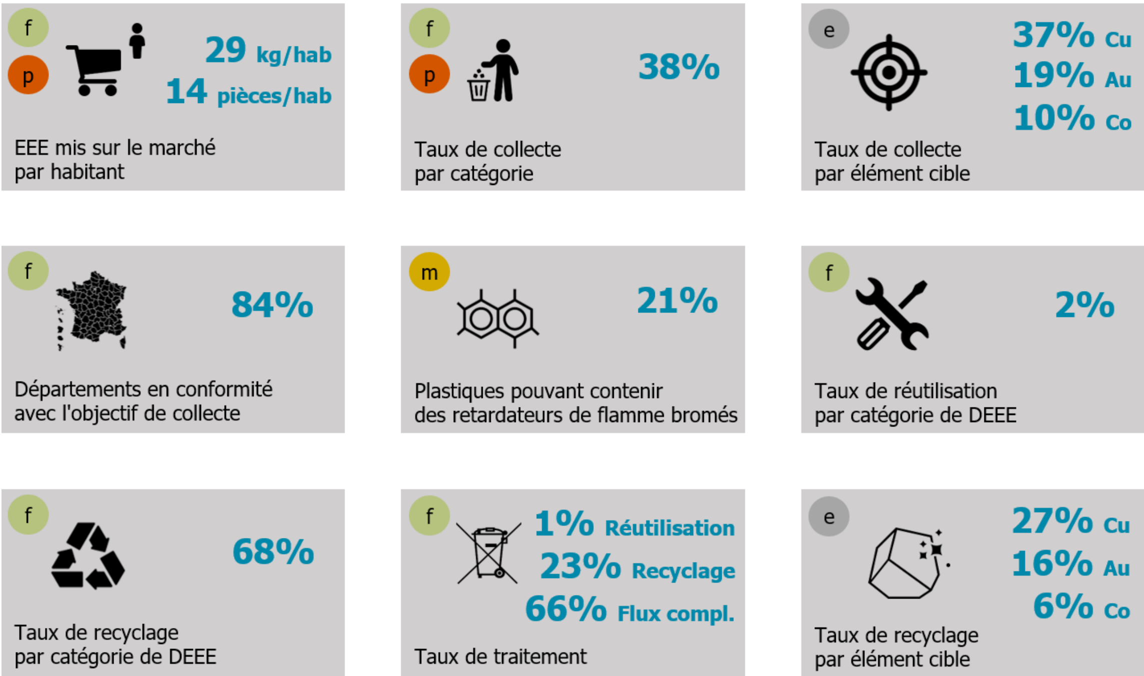


Figure 9.4. Tableau de bord des indicateurs techniques.

Les CRT et les EP ont des compositions différentes selon le type, les fonctionnalités et les fabricants des appareils. Ainsi, les changements de type d'écrans générés et collectés impactent le taux de collecte basé sur le poids global, et aussi par élément cible. Un écran CRT de 17 pouces a un poids moyen de 20 kg, dont environ 50 % de verre CRT, 15 % de métaux, 15 % de plastiques et 6 % de circuits imprimés (PCB) (Eco-Systèmes, 2013 a ; Hischier et al., 2007). Un ordinateur portable a un poids moyen de 2 kg, dont les principaux matériaux et composants, en termes de poids, sont les plastiques (de 20 à 40 %), les métaux (25 %) et les PCB (15 %) (Eco-Systèmes, 2013b; Hischier et al., 2007; HP Development Company, 2018).

Dans l'étude de cas, nous avons estimé que le taux de collecte et de recyclage des éléments les plus présents dans les EP, et plus particulièrement dans les tablettes et les ordinateurs portables, est trois à quatre fois inférieur à celui des éléments les plus présents dans les appareils à tube cathodique. Il est donc important de disposer d'indicateurs pour contrôler la qualité des déchets électroniques collectés et recyclés par les filières réglementaires.

Cette approche nous a permis de quantifier les performances de la filière DEEE avec un niveau de détail qui n'était pas possible auparavant. La Figure 9.5 présente certains résultats obtenus lors de la validation des indicateurs techniques pour l'étude de cas des écrans. Il illustre les résultats obtenus avec les indicateurs « Taux de collecte par élément cible » et « Taux de recyclage par élément cible », en plus de fournir des données sur la qualité et la taille des flux complémentaires et des pertes liées aux limites de recyclage.

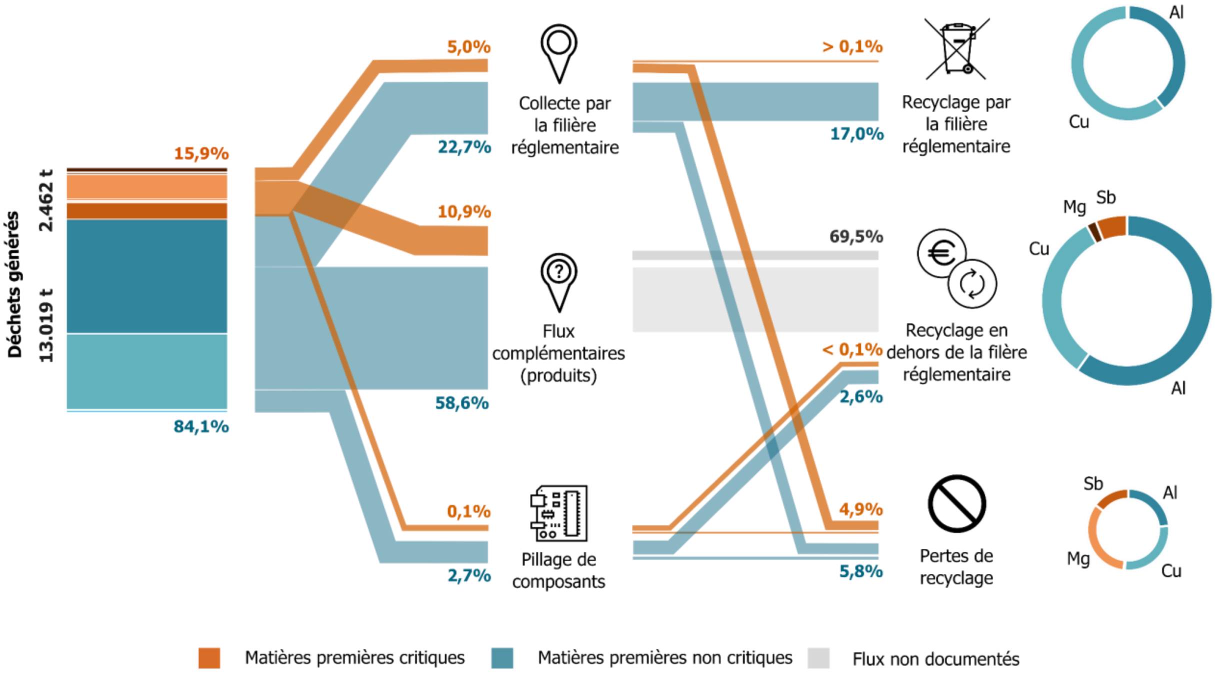


Figure 9.5. Flux des écrans DEEE: matériaux cibles générés, collectés et recyclés (approche basée sur le poids).

En considérant le poids, en 2018, les matières premières non critiques représentent 84 % des éléments cibles générés par les écrans en France, principalement en raison de la forte teneur en aluminium (Al) et en cuivre (Cu) des écrans. En ce qui concerne la collecte, une quantité importante de déchets électroniques est détournée vers des flux de produits complémentaires et le pillage des composants. Compte tenu de l'importance des flux complémentaires (environ 70 % du poids total des éléments cibles générés dans les écrans), elle ne peut être négligée, soulignant l'intérêt de l'approche DEEE générés.

Les résultats de l'étude de cas confirment que le suivi de la performance par élément cible par rapport aux indicateurs de poids global permet d'avoir un aperçu du volume et de la qualité des ressources secondaires potentielles. Il permet également d'identifier les principaux défis à relever pour améliorer la récupération des matériaux vers une économie circulaire.

### 9.5.3. Indicateurs et méthodes d'évaluation des performances environnementales et économiques

La recherche de méthodes pour évaluer la filière des DEEE au-delà de l'approche basée sur le poids a été divisée en trois chapitres : environnemental (**Chapitre 5**), économique (**Chapitre 6**), et critique (**Chapitre 7**). Cette évaluation a suivi les cinq niveaux d'évaluation présentés dans la section précédente.

### Évaluation environnementale

Dans la revue effectuée pour identifier les indicateurs proposés dans la littérature, parmi les études qui proposaient des indicateurs environnementaux, la majorité a utilisé l'ACV. Nous avons donc exploré le potentiel de l'ACV pour fournir des résultats sur les impacts et les avantages de la filière DEEE. Sur la base de l'évaluation de plusieurs études d'ACV pour les DEEE (section 5.3) et de l'étude de cas (section 5.5), nous confirmons que l'ACV peut fournir des informations intéressantes sur les impacts et les bénéfices de la filière DEEE.

Entre autres, nous avons conclu que les impacts totaux du réchauffement climatique et de la toxicité pour l'homme diminuent au fil des ans, avec la réduction du nombre de moniteurs et de téléviseurs à tube cathodique collectés. Toutefois, les scénarios de traitement et les résultats sont influencés par la limitation des données (sections 5.4 et 5.6).

Dans ce contexte, nous recommandons de développer des études d'ACV avec une approche du berceau à la porte (en anglais « cradle-to-gate) par les fournisseurs de traitement des déchets électroniques (qui possèdent des données primaires), par exemple, pour comparer les voies de recyclage dans une perspective environnementale (Grimaud, 2019 ; Grimaud et al., 2017). En raison de la limitation des données et de la complexité de la méthode, nous avons conclu que l'ACV n'est pas adaptée pour calculer des résultats globaux de la performance environnementale de la filière DEEE. La méthode est pertinente du point de vue de la recherche, mais pas pour une application sur le terrain pour un suivi annuel. Ainsi, au lieu de proposer un indicateur basé sur l'évaluation des impacts par tonne de DEEE traités (information difficilement quantifiable actuellement), nous proposons deux indicateurs environnementaux basés sur l'inventaire du traitement des déchets électroniques publié par la base de données ESR (section 5.4.2).

La Figure 9.6 présente les deux indicateurs environnementaux proposés dans le cadre de ce travail, leur niveau d'évaluation et les résultats de l'étude de cas présentés à la section 5.7.

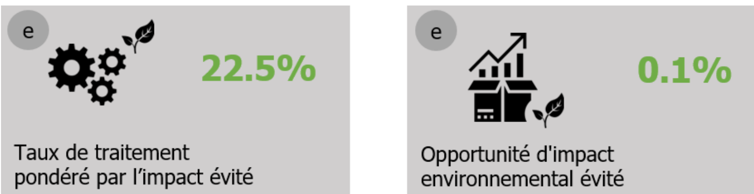


Figure 9.6. Tableau de bord des indicateurs environnementaux.

L'indicateur « Taux de traitement pondéré par l'impact évité » (voir section 5.7.1) utilise des données sur la composition des déchets électroniques générés et collectés par les systèmes officiels, des informations sur les impacts évités par le traitement des DEEE obtenus auprès de la base de la base de données ESR et de l'analyse d'impact du cycle de vie (AICV). Cet indicateur peut soutenir le passage de l'approche actuelle qui consiste à mettre l'accent sur les éléments présents dans les poids plus élevés, afin de stimuler la collecte et le traitement des éléments présentant un plus grand intérêt environnemental. Comme le montre la Figure 9.7, lorsque l'on inclut l'aspect environnemental dans l'évaluation, la part entre les éléments cibles change en raison des différentes performances environnementales des éléments cibles.

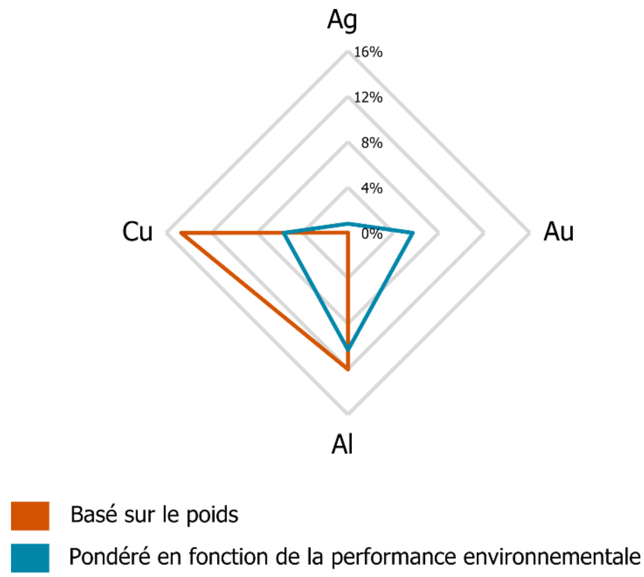


Figure 9.7. Taux de traitement des écrans plats : approches basées sur le poids et sur l'environnement.

Les principales limites de l'indicateur sont liées au périmètre de la base de données. Actuellement, la base contient un nombre limité de produits (p. ex. les tubes cathodiques ne sont pas inclus), pour une gamme limitée de matériaux et d'éléments, et les scénarios de traitement ne sont pas détaillés.

De son côté, l'indicateur « Opportunité d'impact environnemental évité » (section 7.6.2) estime le potentiel des flux complémentaires (flux non captés par la filière réglementaire), pour éviter les impacts environnementaux de la consommation actuelle des éléments ciblés par l'économie française.

### Évaluation économique

Après une analyse plus approfondie du modèle économique de la filière DEEE, ainsi que l'identification de ses coûts et bénéfices du point de vue des éco-organismes (section 6.2), nous proposons deux ensembles d'indicateurs économiques. Les huit indicateurs économiques, leur niveau d'évaluation et les résultats de l'étude de cas (section 6.4) sont présentés dans la Figure 9.8.

Le premier ensemble, appelé « rapport coût-efficacité », regroupe cinq indicateurs (« Eco-participation par EEE mis sur le marché », « Eco-participation par habitant », « Coût total par DEEE collecté », « Coût total par DEEE traité » et « Coût total par habitant ») qui visent à améliorer le suivi et la communication des recettes et dépenses (section 6.3.1). Ces indicateurs peuvent être utilisés pour améliorer la communication des coûts et des recettes et pour sensibiliser davantage les consommateurs aux coûts de la filière des DEEE et à l'importance de l'éco-participations. Dans l'étude de cas, nous n'avons pas pu évaluer une catégorie spécifique de DEEE parce qu'aujourd'hui les éco-organismes ne divulguent pas les données des coûts et dépenses par catégorie. Nous recommandons un reporting par catégorie afin de permettre une meilleure compréhension des particularités par type de déchets électroniques. Magalini et Huisman (2018), ont estimé que les coûts et les revenus peuvent différer considérablement en fonction des catégories.

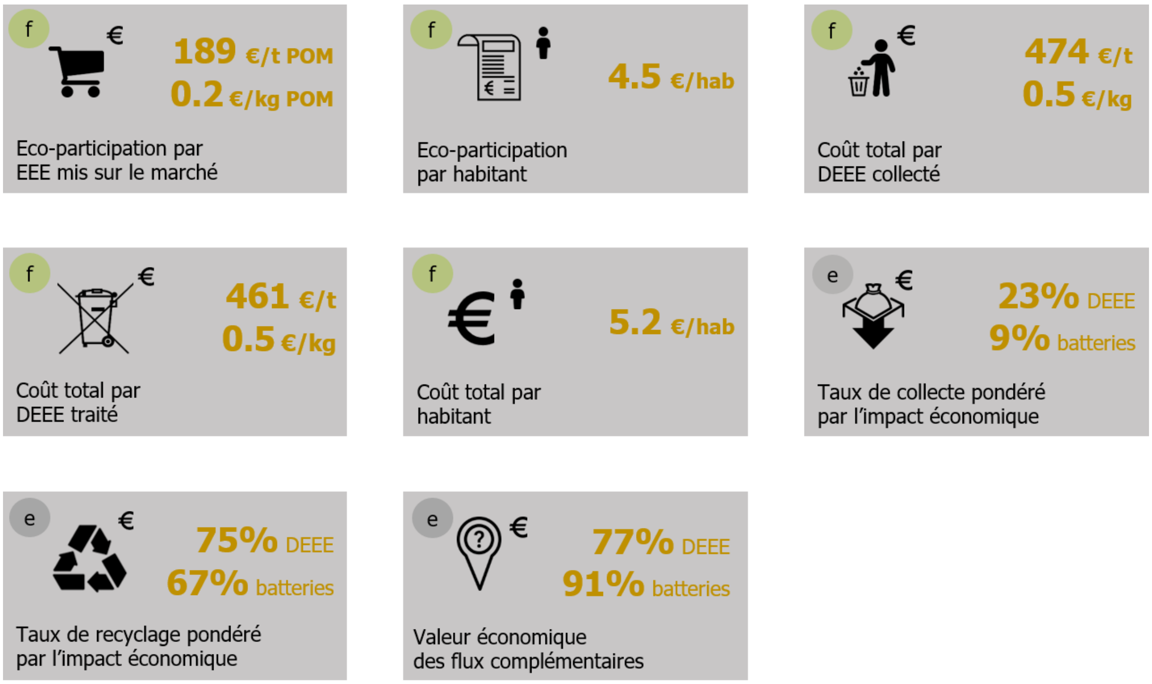


Figure 9.8. Tableau de bord des indicateurs économiques.

Le deuxième ensemble d'indicateurs, nommés « bénéfices économiques » (section 6.3.2), présente une approche alternative aux indicateurs basés sur le poids et regroupe trois indicateurs (« Taux de collecte pondéré par l'impact économique », « Taux de recyclage pondéré par l'impact économique » et « Valeur économique des flux complémentaires »). Cette approche peut être utilisée pour la prise de décision, par exemple, pour prioriser le recyclage des matériaux en tenant compte non seulement de leur poids, mais aussi de leur valeur économique. En ajoutant la valeur économique à l'approche actuelle, l'intérêt de la collecte et du recyclage des éléments trouvés en petites quantités, mais ayant une valeur élevée (p. ex. or et palladium), est soulignée. C'est ce qu'illustre la Figure 9.9.

La valeur intrinsèque des flux complémentaires représente plus de quatre fois la taille, en euros, des éléments cibles recyclés par la filière réglementaire. Même si, potentiellement, une partie des flux non captés est réutilisée ou stockée plus longtemps que prévu dans le modèle de durée de vie (section 4.2.1), il est intéressant d'estimer la valeur économique de ces flux pour souligner leur potentiel et la nécessité d'améliorer la collecte. L'or représente environ 60 % de la valeur intrinsèque des flux complémentaires. Il est surtout présent dans les PCB des ordinateurs portables et des tablettes (56 %) qui ont encore un faible taux de collecte par la filière réglementaire.

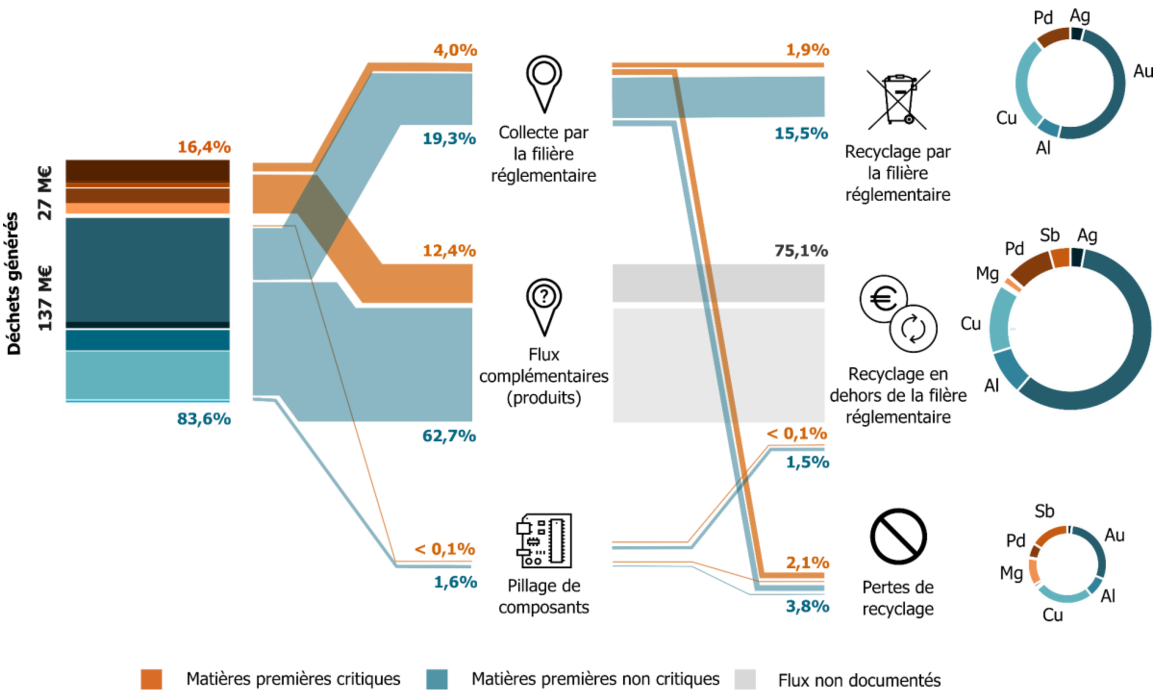


Figure 9.9. Flux des écrans DEEE: matériaux cibles générés, collectés et recyclés (approche économique).

En ce qui concerne les éléments recyclés par la filière réglementaire, on constate une différence entre la part de matériaux critiques et de matériaux non critiques en fonction de la valeur intrinsèque (Figure 9.9) et en fonction du poids (Figure 9.5). Avec l'approche basée sur le poids, les matériaux critiques cibles représentent 0,3 % des éléments recyclés (calculé comme le rapport entre les quantités recyclées et la quantité totale générée), alors que selon une approche économique, ils représentent 1,9 %. Les matériaux critiques recyclés ayant la valeur intrinsèque la plus élevée sont le palladium et le cobalt. En 2018, l'indium, un matériau critique également ciblé par cette étude, a eu des prix de vente plus élevés que le cobalt (tonne/€), mais comme souligné au **Chapitre 4**, son recyclage à l'échelle industrielle est limité.

## Évaluation de la criticité

Étant donné le potentiel des DEEE en tant que gisements de matières premières secondaires, nous avons exploré la combinaison de l'évaluation de la criticité et des mesures pour évaluer l'efficacité de récupération des matières. Cependant, avant de proposer un indicateur, nous nous sommes penchés sur les concepts liés à cette approche (section 7.2), et plus particulièrement sur la méthode d'évaluation de la criticité développée par la Commission européenne (section 7.3).

Sur la base d'études antérieures identifiées dans la revue de la littérature (Mohamed Sultan et al., 2017 ; Nelen et al., 2014 ; Van Eygen et al., 2016), nous proposons trois indicateurs à utiliser par les éco-organismes pour estimer les performances de la filière DEEE dans une perspective de criticité (Figure 9.10).


Figure 9.10. Tableau de bord des indicateurs de criticité.

Les indicateurs « Taux de collecte pondéré par la criticité » et « Taux de recyclage pondéré par la criticité » présentent un résultat complémentaire à la mesure basée sur le poids. Notre approche ne prend en compte que le risque d'approvisionnement selon la méthodologie de la CE, ainsi que le prix du marché des éléments. De plus, puisque cet indicateur vise à capter la capacité de la filière DEEE à fournir des matières secondaires à l'économie, nous incluons leur importance dans le secteur des EEE.

Comme les autres indicateurs proposés dans la présente thèse, ces indicateurs visent à présenter une mesure complémentaire à l'approche actuelle fondée sur le poids. La Figure 9.11 présente une partie des résultats obtenus avec l'approche de criticité suggérée dans cette thèse.

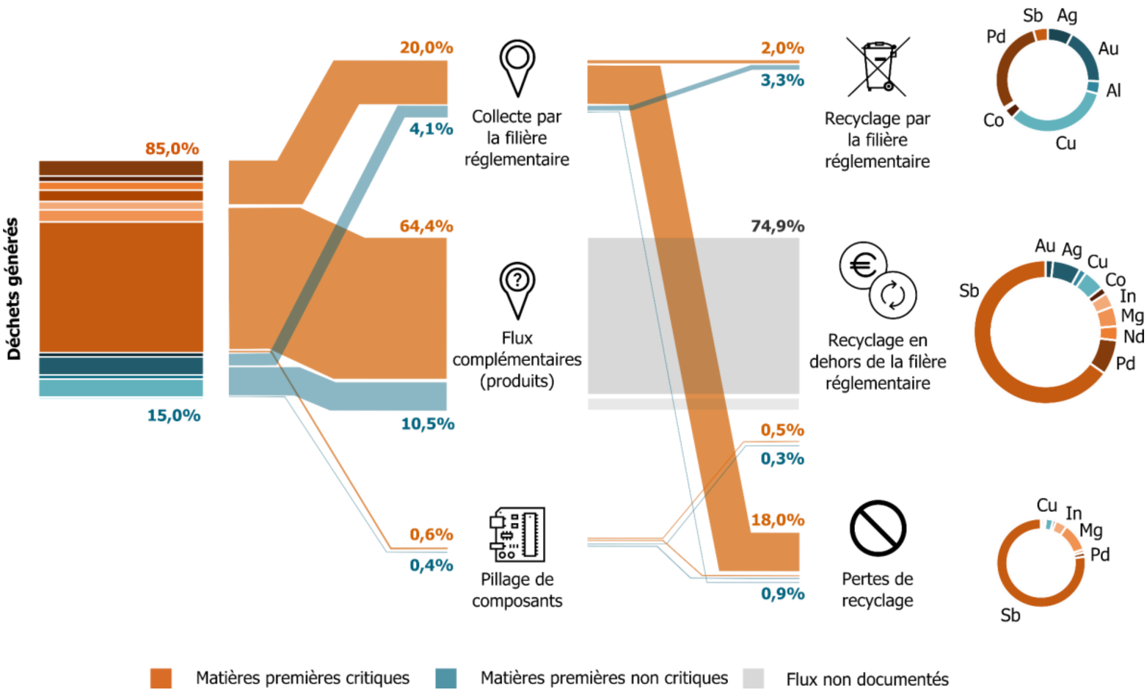


Figure 9.11. Flux des écrans DEEE: matériaux cibles générés, collectés et recyclés (approche basé sur a criticité).

Nous avons identifié, à partir des résultats des approches complémentaires au taux de collecte basé sur le poids présentées dans cette thèse, que lorsque d'autres paramètres sont considérés en plus du poids de l'élément, le résultat global (incluant les matériaux critiques et les matériaux non critiques) diminue. De plus, avec l'approche pondérée par la criticité, des éléments présents en concentrations minimales dans les écrans, comme le néodyme et le cobalt, ont une plus grande importance. Les résultats pondérés en fonction de la criticité sont principalement influencés par l'antimoine (Sb), le magnésium (Mg) et le palladium (Pd). Parmi les éléments recyclés par les flux officiels, les plus significatifs avec l'approche de criticité sont le cuivre (matériaux non critiques) et le palladium (matériau précieux et critique).

Enfin, l'indicateur « Contribution potentielle à la consommation apparente » estime la contribution des flux complémentaires et des pertes de recyclage à la consommation apparente de matières. Nous avons estimé que certains des matériaux critiques ciblés dans notre étude — axée uniquement sur les écrans — pourraient contribuer à plus de 20 % de la demande actuelle de matières premières, ce qui souligne l'importance des déchets électroniques en tant que mines urbaines.

### 9.5.4. Vers une filière plus durable et circulaire

La dernière question de recherche concernant la possibilité de proposer des améliorations en vue d'une filière (D)EEE plus durable et circulaire avec les résultats d'indicateurs multidimensionnels est traitée par la combinaison des résultats présentés aux **Chapitres 4** à **7**. Comme évoqué par Haupt et Hellweg (2019) : « ce qui est mesuré est géré ». Ainsi, un suivi plus détaillé des flux de déchets électroniques améliorera la récupération des éléments cibles dans les années à venir, ainsi que l'amélioration de la gestion actuelle de la collecte et du traitement.

Les résultats individuels de l'étude de cas pour chaque indicateur permettent une vue d'ensemble plus détaillée de la performance de la filière : du niveau macro au niveau microscopique. Par exemple, en quantifiant les flux de déchets électroniques au **Chapitre 4**, nous avons identifié les principales difficultés à améliorer la collecte et le recyclage des éléments cibles. L'évaluation séparée des taux de recyclage et de réutilisation peut stimuler de nouvelles solutions pour augmenter la filière de réutilisation. Du point de vue de l'environnement, au **Chapitre 5**, nous avons identifié, dans un groupe d'éléments cibles, ceux qui pourraient contribuer le plus à étendre les impacts évités par le traitement par la filière réglementaire. L'ensemble d'indicateurs économiques du **Chapitre 6** fournit des informations plus précises sur les dépenses et les recettes de la filière DEEE et une estimation du potentiel économique en euros des flux de déchets électroniques collectés, recyclés et dissipés en flux complémentaires. Ces données peuvent motiver à la fois la collecte, et le développement de technologies de prétraitement plus précises pour réduire les pertes d'éléments présents en petites quantités, mais avec une valeur économique élevée. Les résultats des indicateurs de criticité du **Chapitre 7** permettent d'avoir une vue d'ensemble des taux de collecte et de recyclage en fonction de la criticité et de la capacité de la filière de contribuer en tant que fournisseur de matériaux à la demande de matières premières.

Les résultats de performance de la filière présentés sur une échelle de temps pour certains des paramètres et indicateurs (par exemple, la Figure 4-8 qui présente les données sur les écrans mis sur le marché de 2010 à 2018) permettent d'identifier les changements dans les produits, et par conséquent sur la composition des éléments générés, collectés et recyclés par la filière au fil du temps. Le suivi de ces informations peut soutenir le développement de la filière, ainsi que les investissements et/ou la recherche dans le développement de nouveaux procédés de traitement adaptés aux futurs flux de déchets électroniques.

Bien qu'il puisse être difficile pour les éco-organismes et les décideurs politiques de suivre et d'interpréter en permanence les résultats des vingt-deux indicateurs pour différentes catégories de DEEE, nous ne pensons pas qu'il soit possible de proposer un indicateur unique global (indice) calculé en pondérant et en additionnant tous les indicateurs, ou par type d'indicateur (technique, économique, environnemental et critique). Notre raisonnement repose sur quatre arguments :

- Les indicateurs évaluent différents aspects du traitement des DEEE et ont parfois un périmètre d'évaluation différent, ce qui rend l'agrégation difficile ;
- Nous croyons que des données importantes seraient perdues. Comme indiqué à la section 2.2.2, dans l'UE, nous disposons actuellement de trois indicateurs : le taux de collecte (pour toutes les catégories de DEEE), le taux de recyclage et de préparation à la réutilisation (pour chaque catégorie de DEEE) et le taux de valorisation (pour chaque catégorie). L'une des principales limites de ces indicateurs est qu'ils fournissent un aperçu macroscopique et généraliste des performances ;
- Comme indiqué à la section 3.2, actuellement, les éco-organismes et l'ADEME utilisent un nombre significatif d'indicateurs pour suivre la filière ;
- Enfin, les taux de collecte et de recyclage pondérés par la criticité (indicateurs 20 et 21) sont déjà, à certains égards, une agrégation de différents paramètres : poids, économie et risque d'approvisionnement. Néanmoins, tel que présenté à la section 7.6.1, il est intéressant de suivre cet indicateur en le comparant à des paramètres fondés sur le poids et à des paramètres fondés sur la pondération économique.

Certains des indicateurs suggérés dans ce travail proposent le suivi de différents éléments cibles, ce qui peut également être difficile de mettre en place par les éco-organismes et les décideurs politiques. Une possibilité serait de réduire le champ d'application des indicateurs à quelques éléments cibles sélectionnés par catégorie de DEEE. Nous recommandons que la sélection des éléments cibles se fasse après un premier screening de la composition (par exemple avec les données de la Urban Mine Platform) afin d'identifier les éléments pertinents (du point de vue massique, économique, environnemental et de la criticité) pour chaque catégorie de DEEE en considérant que les déchets électroniques sont très hétérogènes.

## 9.6. Nouvel ensemble d'indicateurs pour évaluer la performance de la filière DEEE

### 9.6.1. Paramètres principaux

Ce travail propose une extension des indicateurs actuels pour évaluer les performances des systèmes DEEE. Notre méthode est basée sur cinq paramètres principaux résumés ci-dessous. Comme l'illustre la

Figure 9.12, le périmètre des indicateurs est variable et peut comprendre la génération, la collecte et le traitement des DEEE par la filière réglementaire, ainsi que les flux parallèles qui détournent les ressources secondaires de la filière réglementaire.

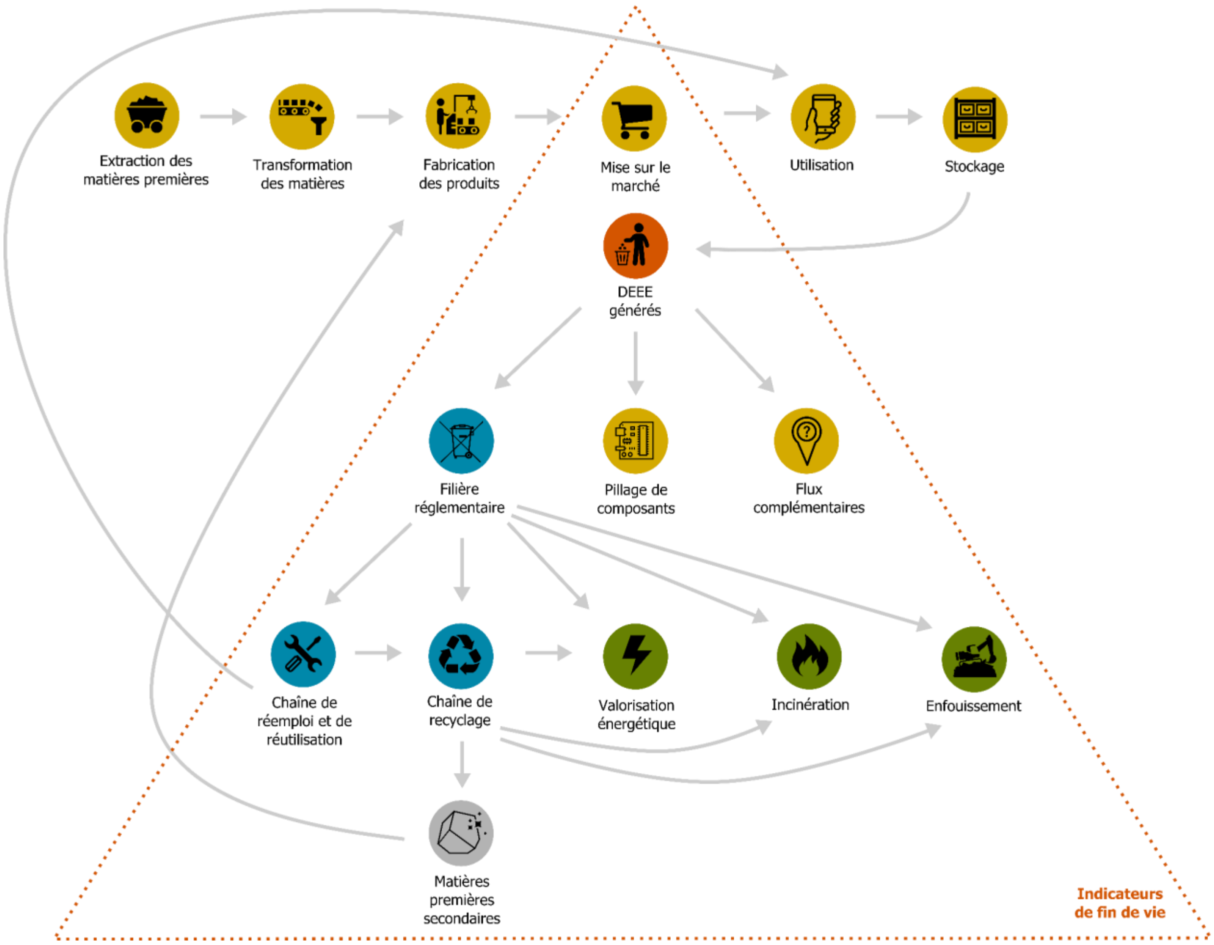


Figure 9.12. Périmètre des indicateurs proposés dans la thèse.

### Suivi des flux de déchets électroniques

Le point de départ de notre approche est la cartographie des flux de déchets électroniques depuis la génération de déchets jusqu'au recyclage des éléments cibles. Cette approche est basée sur l'AFM et comporte ses étapes de base : (1) détermination du modèle du système, dont les procédés et les matériaux, (2) calcul des flux de matériaux et (3) interprétation des résultats (Clarke et al., 2019).

Les déchets électroniques générés sont quantifiés à l'aide de l'outil « WEEE Calculation Tool » (section 4.2.1), qui prend en compte la quantité d'EEE mis sur le marché (POM) au cours des années et les durées de vie correspondantes des produits (Baldé et al., 2017). Par la suite, les déchets électroniques collectés sont quantifiés en fonction des données de la filière réglementaire (section 4.2.2). Les flux complémentaires (incluant le pillage des produits et des composants) peuvent être calculés comme la différence entre ces deux flux (section 4.2.3).

Enfin, le traitement des déchets électroniques est estimé sur la base des données relatives au traitement actuel des déchets électroniques par la filière réglementaire. Le recyclage étant aujourd'hui le traitement le plus réalisé en France, il constitue l'axe majeur de cette étude, mais notre cartographie inclut les autres types de traitement (réutilisation, valorisation énergétique, etc.).

**Composition**

Les éco-organismes et l'ADEME suivent la performance par catégorie de DEEE, mais notre approche inclut non seulement le niveau macroscopique (catégories), mais aussi les niveaux méso (produits), mésoscopique (composants et matériaux) et microscopique (éléments). Par conséquent, dans l'étude de cas (section 4.3), nous avons complété les données de la filière avec des données du projet ProSUM concernant la qualité des déchets électroniques collectés.

**Recyclage**

Comme mentionné précédemment, cette thèse se focalise sur le recyclage (section 4.4). Néanmoins, elle va au-delà de la fin du statut de déchet et inclut le prétraitement et l'efficacité du traitement final. L'information requise pour calculer l'efficacité de recyclage est détaillée à la section 4.6.3 avec l'étude de cas sur les écrans.

**Impacts évités**

L'impact évité est une approche utilisée dans l'ACV dans le contexte de la répartition des impacts environnementaux en présence du recyclage, de la réutilisation et de la régénération énergétique. La collecte, le traitement et le recyclage des DEEE sont donc considérés comme des moyens d'éviter les impacts de la production de nouveaux matériaux (Huisman, 2003 ; Seyring et al., 2015). Fondés sur cette approche, les indicateurs environnementaux proposés dans cette thèse sont basés sur les impacts évités des éléments traités par la filière réglementaire. L'indicateur est basé sur les résultats de l'ACV et sur les inventaires de la base de données ESR.

**Valeur économique (intrinsèque)**

La valeur économique de certains composants et éléments dans les DEEE est l'une des principales motivations pour leur traitement. Nous suggérons donc d'évaluer la valeur des flux de déchets électroniques collectés, recyclés et des flux complémentaires comme une nouvelle approche pour mesurer la performance économique de la filière (voir section 6.3.2 et Figure 6-6). Ce paramètre est également utilisé par les indicateurs de criticité.

**Criticité**

Un matériau est considéré comme critique pour l'UE s'il a une grande importance économique et si son approvisionnement est susceptible d'être perturbé (Blengini et al., 2017). Dans le but de considérer la criticité des matériaux comme un aspect pertinent dans la récupération des matières secondaires, nous suggérons de considérer l'indicateur de risque d'approvisionnement selon la CE et la part des éléments dans le secteur électronique (section 7.5). La vulnérabilité à une rupture d'approvisionnement signifie qu'il existe un risque élevé que l'approvisionnement en matières premières soit probablement insuffisant pour répondre à la demande de l'industrie européenne. En retour, la part du secteur montre, en même temps, la demande du secteur électronique en matériaux, ainsi que la qualité et la taille des matières premières secondaires dans le secteur des déchets électroniques. Cette approche est complémentaire aux mesures environnementales et économiques qui envisagent de fournir d'autres mesures pour évaluer la performance de la collecte et du traitement au-delà du poids total.

### 9.6.2. Analyse de l'ensemble des indicateurs

Dans cette section, nous présentons une analyse des indicateurs résumés au Tableau 9-1 selon les différents critères autres que la classification par type d'indicateur.

Nous avons identifié quatre groupes d'indicateurs en fonction de leur degré de préparation à l'adoption en fonction des données requises :

1. Adoption immédiate, car les paramètres nécessaires au calcul des indicateurs font actuellement l'objet d'un suivi - indicateurs « EEE mis sur le marché par habitant », « Taux de conformité des départements à l'objectif de collecte des DEEE », « Taux de plastiques pouvant contenir des retardateurs de flamme bromés dans les DEEE collectés », « Taux de réutilisation par catégorie de DEEE » et « Taux de recyclage par catégorie de DEEE ».

2. Adoption immédiate par certaines catégories, éléments et/ou pour communiquer un résultat global de la chaîne des DEEE — indicateurs « Eco-participation par EEE mis sur le marché », « Eco-participation par habitant », « Coût total par DEEE collecté », « Coût total par DEEE traité » et « Coût total par habitant ».

3. Adoption à moyen terme, car, outre l'adoption du calcul des déchets générés avec l'outil « WEEE Calculation Tool » il faut un meilleur suivi de la collecte des déchets électroniques par type de produit (UNU-keys) — indicateurs « Taux de collecte basé sur l'approche générée par les DEEE » et « Taux de traitement basé sur l'approche des DEEE générés ».

4. Adoption à moyen terme parce que les données requises sur la composition des produits ne sont pas suivies actuellement - indicateurs « Taux de collecte par élément cible », « Taux de recyclage par élément cible », « Taux de traitement pondéré par l'impact évité », « Opportunité d'impact environnemental évité », « Taux de collecte pondéré par l'impact économique », « Taux de recyclage pondéré par l'impact économique », « Valeur économique des flux complémentaires », « Taux de collecte pondéré par la criticité »,

« Taux de recyclage pondéré par la criticité » et « Contribution potentielle à la consommation apparente ».

En ce qui concerne l'utilisation potentielle des indicateurs, nous avons identifié trois groupes qui regroupent les indicateurs de tous types (techniques, économiques, environnementaux et de criticité), en fonction du niveau d'évaluation :

1. Efficacité macroscopique du système : il présente une vue d'ensemble des performances de la chaîne des DEEE - indicateurs « EEE mis sur le marché par habitant », « Taux de collecte basé sur l'approche générée par les DEEE », « Taux de conformité des départements à l'objectif de collecte des DEEE », « Taux de réutilisation par catégorie de DEEE », « Taux de recyclage par catégorie de DEEE », « Taux de traitement basé sur l'approche des DEEE générés », « Eco-participation par EEE mis sur le marché », « Eco-participation par habitant », « Coût total par DEEE collecté », « Coût total par DEEE traité » et « Coût total par habitant ».

2. Efficacité mésoscopique et microscopique du système : elle présente des informations plus détaillées sur les matériaux, composants et éléments - indicateurs « Taux de collecte par élément cible », « Taux de plastiques pouvant contenir des retardateurs de flamme bromés dans les DEEE collectés », « Taux de recyclage par élément cible », « Taux de traitement pondéré par l'impact évité », « Taux de collecte pondéré par l'impact économique », « Taux de recyclage pondéré par l'impact économique », « Taux de collecte pondéré par la criticité » et « Taux de recyclage pondéré par la criticité ».

3. Avantages du traitement des déchets électroniques : ceux-ci évaluent les avantages potentiels de la filière — indicateurs « Opportunité d'impact environnemental évité », « Valeur économique des flux complémentaires » et « Contribution potentielle à la consommation apparente ».

Enfin, nous avons identifié que certains indicateurs, en plus de fournir des données sur les progrès de la performance de la chaîne des DEEE, peuvent également être utilisés dans la prise de décision et la planification des stratégies futures pour développer les voies de traitement et stimuler la valorisation des éléments cibles. Ces indicateurs représentent un peu plus de 50 % du tableau de bord : indicateurs « EEE mis sur le marché par habitant », « Taux de collecte par élément cible », « Taux de plastiques pouvant contenir des retardateurs de flamme bromés dans les DEEE collectés », « Taux de recyclage par élément cible », « Taux de traitement pondéré par l'impact évité », « Opportunité d'impact environnemental évité », « Eco-participation par EEE mis sur le marché », « Taux de collecte pondéré par l'impact économique », « Taux de recyclage pondéré par l'impact économique », « Valeur économique des flux complémentaires », « Taux de collecte pondéré par la criticité », « Taux de recyclage pondéré par la criticité » et « Contribution potentielle à la consommation apparente ».

### 9.6.3. Disponibilité et robustesse des données

La faisabilité de la mise en œuvre régulière des indicateurs suggérés dans ce travail par les éco-organismes et décideurs politiques dépend de la disponibilité des données nécessaires à leur calcul. Par conséquent, il est nécessaire de définir une stratégie pour résoudre les problèmes de disponibilité des données.

Concernant les données de DEEE générés, nous suggérons que la filière française adopte l'outil « WEEE Calculation Tool ». L'adoption de cet outil par la France et les autres États membres permettra un suivi régulier des taux de collecte en fonction des déchets générés. Étant donné que les paramètres sont dûment détaillés dans l'outil, il serait possible de développer une interface entre cet outil et les bases de données actuelles utilisées par l'ADEME et les éco-organismes pour calculer les déchets générés, sur la base de données actualisées sur les EEE mis sur le marché en France.

En France, les éco-organismes réalisent annuellement des campagnes de caractérisation pour déterminer la quantité de DEEE collectés par catégorie, ainsi que la composition moyenne par flux de déchets. Une évaluation de la collecte par produits (par exemple, UNU-keys), ainsi que des composants pillés, pourrait être incluse dans la caractérisation annuelle. Comme discuté par Baxter et al (2016), la future réglementation devrait inclure une surveillance par échantillonnages et mesures. Cela permettrait de suivre l'évolution du type d'équipements collectés par la filière réglementaire et de quantifier les matériaux potentiellement disponibles pour le recyclage.

En ce qui concerne la composition des (D)EEE, un travail de collaboration entre les producteurs et les recycleurs pour trouver un moyen rentable et efficace de produire et de partager les données sur les produits et les composants permettrait de recueillir des données (Downes et al., 2017). Cependant, ce n'est pas une question qui peut être résolue à court terme. Une solution à court/moyen terme consiste à utiliser les données du projet ProSUM disponibles dans la base de données Urban Mine Platform. Néanmoins, à long terme, une autre solution est nécessaire, car la composition des appareils électroniques change rapidement avec les progrès technologiques.

Il est nécessaire de mettre en place des mécanismes obligatoires pour forcer les fournisseurs de services de traitement des déchets à communiquer aux éco-organismes les données relatives à l'efficacité du traitement des déchets électroniques (opérations de tri et de recyclage final), par exemple en imposant des exigences dans les cahiers des charges pour les DEEE ménagers et professionnels. Pour atteindre la circularité des matériaux, la filière DEEE doit être plus transparente, ce qui inclut l'amélioration du contrôle des flux de DEEE.

Pour déterminer la consommation apparente en France, nous avons utilisé les données sur la consommation de l'UE déterminée dans l'étude de criticité de la CE (Deloitte Sustainability et al., 2017a, 2017b), ainsi qu'une moyenne du pourcentage global des métaux consommés en France selon les statistiques Eurostat (EUROSTAT, 2019). Nous suggérons d'utiliser des données plus précises sur la consommation en France, estimée par la Direction des études statistiques de la Commission Générale du Développement Durable (Calatayud and Mohkam, 2018). Nous n'avons pas eu accès à ces données pour l'étude de cas, mais ces informations sont potentiellement disponibles auprès du Ministère de la Transition Écologique et Solidaire.

Comme souligné dans la section 5.4.2, ESR a développé une base de données d'Inventaire du Cycle de Vie (ICV) pour la fin de vie des déchets électroniques en France. Il s'agissait d'un projet pionnier dans le domaine du traitement de fin de vie, et ESR représentant environ 75 % du marché des éco-organismes, il est représentatif du traitement sur le terrain. Jusqu'à présent, ESR a décidé de ne pas rendre publics les rapports individuels contenant des détails sur les données utilisées et les hypothèses de la modélisation. En outre, les ensembles de données disponibles dans cette base ne couvrent que quelques éléments et matériaux, et seulement pour certains des produits des catégories de DEEE (par exemple, les tubes cathodiques ne sont pas inclus). Nous suggérons d'élargir la base de données ESR actuelle (éventuellement avec des données provenant d'autres éco-organismes), avec plus de transparence sur les scénarios et les choix méthodologiques.

Les indicateurs de « coût-efficacité » proposés à la section 6.3.1 ne peuvent pas être évalués par catégorie, car, aujourd'hui, les éco-organismes ne partagent pas avec l'ADEME les données de recettes et de coûts par catégorie. Nous croyons qu'il est important de communiquer les résultats par catégorie. Cependant, nous comprenons que éco-organismes ne veulent pas divulguer de données sensibles. Ainsi, nous suggérons que l'ADEME publie des résultats moyens des éco-organismes, et non des performances financières.

Les indicateurs de « bénéfices économiques » proposés à la section 6.3.2 requièrent des données sur les prix du marché des matériaux et éléments. Ces données peuvent être obtenues de différentes sources, bien que la plupart d'entre elles ne soient pas gratuites.

Enfin, les deux paramètres liés aux indicateurs de criticité (risque d'approvisionnement et part du secteur des EEE) peuvent être obtenus dans les publications de la CE (plateforme RMIS et Deloitte Sustainability et al., 2017a, 2017b). La CE met à jour les données tous les trois ans (Commission européenne, 2019). Néanmoins, les données se limitent aux matériaux candidats sur lesquels l'évaluation de la criticité a été effectuée.

## 9.7. Commentaire final

Avec les études antérieures publiées dans la littérature (Ardente et al., 2014; Haupt et al., 2016; Huisman, 2003; Nelen et al., 2014; Parajuly et al., 2017; Tansel, 2017; Van Eygen et al., 2016; Wang, 2014), ce travail peut soutenir le développement de différents indicateurs pour évaluer les priorités environnementales, économiques et critiques relatives aux matières secondaires. Les futures politiques pourraient adopter des indicateurs allant au-delà de l'approche fondée sur le poids et de meilleures pratiques de gestion des DEEE afin d'améliorer le suivi et la valorisation des déchets électroniques et des matières secondaires (critiques). En adoptant cette approche, des activités de gestion des DEEE ont la possibilité d'étendre leur champ d'application au-delà de la gestion des déchets et des substances dangereuses, afin de devenir des fournisseurs des matières secondaires de qualité.

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## Appendix 1 | Abbreviations and parameters used in the equations

### Abbreviations

| | |
|---|---|
| ABS | Acrylonitrile Butadiene Styrene |
| ADEME | Agence de l'Environnement et de la Maîtrise de l'Energie |
| Ag | Silver |
| Al | Aluminum |
| Au | Gold |
| BFR | Brominated Flame Retardant |
| CFC | Chlorofluorocarbon |
| Co | Cobalt |
| $CO_2$ | Carbon dioxide |
| Cu | Copper |
| CRM | Critical Raw Material |
| CRT | Cathode-Ray Tube |
| EEE | Electrical and Electronic Equipment |
| EERA | European Energy Research Alliance |
| EC | European Commission |
| EN | European Standards |
| EOL | End-of-Life |
| EPD | Environmental Product Declaration |
| EPR | Extended Producer Responsibility |
| EU | European Union |
| EI | Economic Importance |
| FPD | Flat-panel displays |
| GHG | Greenhouse Gases |
| GIS | Geographic Information System |
| HC | Hydrocarbon |
| HCFC | Hydrochlorofluorocarbon |
| HREE | Heavy Rare Earth Elements |
| HHI | Herfindahl-Hirschman Index |
| HWIP | Hazardous Waste Incineration plant |
| HS | Harmonized System |
| ICT | Information and Communications Technology |
| IGN | Institut Géographique National |
| In | Indium |
| IT | Information Technology |
| IR | Import Reliance |
| JRC | Joint Research Centre |

| | |
|---|---|
| LED | Light-Emitting Diodes |
| LCA | Life Cycle Assessment |
| LCC | Life Cycle Cost |
| LCD | Liquid-Crystal Displays |
| LCI | Life Cycle Inventory |
| LCIA | Life Cycle Impact Assessment |
| Li | Lithium |
| LREE | Light Rare Earth Elements |
| MCDM | Multi-Criteria Decision Making |
| MFA | Material Flow Analysis |
| Mg | Magnesium |
| NHWSF | Non-Hazardous Waste Storage Facility |
| MSW | Municipal Solid Waste |
| MWIP | Municipal Waste Incineration plant |
| Nd | Neodymium |
| Pb | Lead |
| PC | Polycarbonate |
| PCB | Printed Circuit Boards |
| PCR | Product Category Rules |
| Pd | Palladium |
| PGM | Platinum Group Metals |
| PM | Precious Metal |
| PMMA | Poly(methyl methacrylate) |
| POM | Placed on the market |
| PP | Polypropylene |
| PS | Polystyrene |
| PV | Photovoltaic Panels |
| RA | Risk Assessment |
| RQ | Research Question |
| RMI | Raw Materials Initiative |
| Sb | Antimony |
| Sn | Tin |
| SR | Supply Risk |
| S-LCA | Social Life Cycle Assessment |
| TFT | Thin-Film-Transistor |
| UK | United Kingdom |
| WEEE | Waste Electrical and Electronic Equipment |
| WGI | World Governance Index |
| Zn | Zinc |

**Parameters used in the equations**

| | |
|---|---|
| $AC_e$ | apparent consumption of the target element in France |
| $AI_e$ | environmental impact of the target element including the benefits of substitution (avoided impacts) |
| $A_S$ | share of end use of a raw material in a NACE Rev. 2 2-digit level sector |
| $CC_f$ | total costs per WEEE collected by the official schemes per category |
| $CCR_f$ | criticality weighted-based collection rate per WEEE category |
| $CRR_f$ | criticality weighted-based recycling rate per WEEE category |
| $CH_e$ | element content per component and materials in the products |
| $CI_f$ | total costs of the WEEE chain per inhabitant per category |
| $CR_e$ | collection rate of the target element in the WEEE category |
| $CR_d$ | rate of departments complying with the collection target |
| $CR_f$ | collection rate of the WEEE category |
| $CT_f$ | total costs per WEEE treated by the official schemes per category |
| $DC$ | number of French departments with collection target equal or higher than the collection target set by the WEEE Directive |
| $DT$ | total number of departments in France |
| $ECF_f$ | economic value of the complementary flows per WEEE category |
| $ECR_f$ | economic weighted-based collection rate per WEEE category |
| $EI$ | economic importance |
| $EOL_{RIR}$ | end-of-life recycling input rate |
| $ERR_f$ | economic weighted-based recycling rate per WEEE category |
| $EUsourcing$ | actual sourcing of the EU supply |
| $ETR_f$ | environmentally weighted-based treatment rate per WEEE category |
| $FC_f$ | visible fee per EEE POM per category |
| $FI_f$ | visible fee per inhabitant per category in a given year |
| $GS$ | global supply |
| $HHI$ | Herfindahl-Hirschman Index |
| $IR$ | import reliance |
| $IRM_e$ | environmental impact of the raw material production |
| $n$ | number of different UNU-Keys related to the WEEE category |
| $n1$ | number of different UNU-Keys related to the WEEE category |
| $n2$ | number of components and materials present in the different products |
| $n3$ | number of treatment paths for all the products |
| $n4$ | number of target elements considered in the assessment |
| $N_{inh}$ | population of the country or region in a given year |

| | |
|---|---|
| $MP_e$ | market price of the target element |
| $PBR_f$ | rate of plastics that may contain brominated flame retardants in the WEEE category |
| $PC_e$ | potential contribution of target elements generated and not collected by the official schemes for apparent consumption (per target element in the WEEE category) |
| $POM_f$ | EEE placed on the market per inhabitant per WEEE category |
| $PR_e$ | efficiency of the recycling of the target element per treatment scenario |
| $Q_S$ | NACE Rev. 2 2-digit level sector's value added |
| $RR_e$ | recycling rate of the target element in the WEEE category |
| $RR_f$ | recycling rate of the WEEE category |
| $RuR_f$ | reuse rate of the WEEE category |
| $S$ | denotes sector |
| $SG$ | scavenging rate per component |
| $SH_e$ | share of element used within EEE sector |
| $SI_{EI}$ | substitution index related to economic importance |
| $SI_{SR}$ | substitution index related to supply risk |
| $SR$ | supply risk of the element based on EC criticality methodology |
| $ST$ | share of the treatment scenario per product |
| $t$ | trade parameter adjusting WGI |
| $TC$ | total expenses of the official schemes |
| $TR_f$ | treatment rate (recycling, reuse, energy recovery or disposal) of the WEEE category |
| $VF$ | compliance schemes' revenues with the visible fee |
| $WBr$ | weight of plastics that may contain brominated flame retardants |
| $WC$ | weight of e-waste collected by the official schemes |
| $WC_e$ | weight of target element collected |
| $WGI$ | scaled World Governance Index |
| $WP$ | overall weight of plastic fractions in WEEE collected by the official schemes |
| $WR$ | weight of fractions sent to recycling facilities |
| $RR_e$ | recycling rate of the target element in the WEEE category |
| $WRu$ | weight of e-waste reused |
| $WT$ | weight of e-waste that is sent for the different types of treatment |
| $WG$ | weight of e-waste generated |
| $WG_e$ | weight of target element generated |
| $WM$ | amount of EEE placed on the market |
| $WR_e$ | weight of target element recycled |
| $WTO_f$ | environmental impact avoidance opportunity with WEEE treatment |

## Appendix 2 | Lists of equations, figures and tables

### Equations

#### Chapter 4



#### Chapter 5



#### Chapter 6



#### Chapter 7

## Figures

### Chapter 1



### Chapter 2



### Chapter 3



### Chapter 4

**Chapter 5**



**Chapter 6**

**Chapter 7**



**Chapter 8**



**Chapter 9**

**Tables**

**Chapter 2**



**Chapter 3**



**Chapter 4**



**Chapter 5**

**Chapter 6**



**Chapter 7**



**Chapter 8**

**Chapter 9**

## Appendix 3 | Publications

### Papers published in peer-reviewed journals

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

Horta Arduin, R., Mathieux, F., Huisman, J., Andrea, G., Charbuillet, C., Wagner, M., Baldé, C.P., Perry, N., 2020. Novel indicators to better monitor the collection and recovery of (critical) raw materials in WEEE : Focus on screens. Resour. Conserv. Recycl. 157. https://doi.org/10.1016/j.resconrec.2020.104772

### Conference papers

Horta Arduin, R., Charbuillet, C., Berthoud, F., Perry, N., 2017. Life cycle assessment of end-of-life scenarios: tablet case study. Proceedings Sardinia 2017. Sixteenth International Waste Management and Landfill Symposium. 10 p.

Horta Arduin, R., Martínez Leal, J., Grimaud, G., Charbuillet, C., Pompidou, S., Laratte, B., Perry, N., 2017. Scope Definition on End-of-Life Chain Performance Assessment: Recycling Rate and French E-Waste Chain Case Study. Proceedings of EcoDesign 2017 International Symposium. 8 p.

Horta Arduin, R., Charbuillet, C., Berthoud, F., Perry, N., 2018. LCA of recycling chains: influence of transport modelling. Proceedings of VI Congresso Brasileiro sobre Gestão do Ciclo de Vida. 471–477. ISBN 978-85-7013-146-1

Horta Arduin, R., Grimaud, G., Charbuillet, C., Laratte, B., Perry, N., 2019. Plastics in WEEE screens: difficulties and opportunities to improve the recycling rate. Proceedings of 16th International Conference on Environmental Science and Technology - CEST2019.

Horta Arduin, R., Charbuillet, C., Berthoud, F., Perry, N., 2019. Indicateurs de la filière DEEE. In : Presses des Mines - Les enjeux d'écoconception associés à l'économie circulaire : Etat des besoins terrain et des solutions développées - EcoSD Annual Workshop 2018. 7 p.

**De la gestion des dechets a l'approvisionnement de matieres secondaires : developpement d'indicateurs pour la gestion des DEEE - focus sur la filiere française**

**Resume :** Les Déchets d'Equipements Electriques et Electroniques (DEEE) sont parmi les principaux flux de mines urbaines en raison de leur composition et de leur volume croissant. Actuellement, dans l'Union Européenne (UE), la performance de la filière DEEE est évaluée principalement au moyen d'indicateurs techniques qui visent à garantir la conformité aux objectifs de collecte et de valorisation fixés par la Directive DEEE. La Directive DEEE et la réglementation française fixent des taux de collecte et de traitement plus élevés pour les années à venir. Par conséquent, pour garantir une augmentation de la quantité et de la qualité des DEEE collectés, réutilisés et recyclés, il est nécessaire d'améliorer la connaissance et le contrôle des flux de DEEE. L'objectif de cette thèse est d'établir un groupe d'indicateurs couvrant les aspects multidimensionnels liés à la collecte et au traitement des DEEE. Ces indicateurs visent à améliorer la visibilité sur les progrès réalisés par la filière réglementaire dans une économie circulaire. Différentes priorités techniques, environnementales, économiques et de criticité liées à la récupération des matières secondaires des DEEE sont évaluées. Les indicateurs sont présentés et validés avec un cas d'étude sur les écrans en tenant compte des données et des particularités de la filière française. L'approche multidimensionnelle présentée dans cette étude peut soutenir les politiques futures et les meilleures pratiques en matière de gestion des DEEE afin d'améliorer le suivi et la valorisation des DEEE et des matières secondaires (critiques). En adoptant cette approche, des activités de gestion des DEEE ont le potentiel d'étendre leur champ d'application au-delà de la gestion des déchets et des substances dangereuses, afin de devenir des fournisseurs des matières secondaires de qualité.

**Mots clés :** Déchets d'équipements électriques et électroniques; Indicateurs ; Filière; Recyclage ; Matières Premières Secondaires ; Matières Premières Critiques.

**From waste management to supplier of secondary raw materials: development of indicators to support WEEE chain management - focus on the French system**

**Abstract :** Waste Electrical and Electronic Equipment (WEEE) is among the key urban mining stream due to its composition and rising volume. Currently, in the European Union, WEEE chain performance is mainly assessed by technical indicators that aim to ensure system compliance with collection and recovery targets set by the WEEE Directive. The WEEE Directive and French regulation target higher collection and treatment rates in the coming years. Therefore, to ensure an increase in quantity and quality of e-waste collected, reused and recycled, it is necessary to improve our knowledge and control of the WEEE flows. The goal of this thesis is to establish a robust set of indicators covering multidimensional aspects related to the collection and treatment of WEEE. These indicators intend to improve the visibility on the progress of the WEEE official schemes in a circular economy. Different technical, environmental, economic and criticality priorities related to the recovery of raw materials from e-waste are assessed. The indicators are presented and validated with a case study focused on waste screens, considering data and particularities of the e-waste chain in France. The multidimensional approach presented in this study can support future policies and best practices in WEEE management in order to improve e-waste tracking and the recovery of (critical) raw materials. In so doing, more targeted WEEE management activities have the potential to extend the scope from waste and hazardous substances management to enhancing the supply of quality secondary raw materials.

**Keywords** : Waste Electrical and Electronic Equipment; Indicators; E-Waste Schemes; Recycling; Secondary Raw Materials; Critical Raw Materials.

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