Science of the Total Environment 908 (2024) 168295

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

# Environmental and socio-economic effects of construction and demolition waste recycling in the European Union

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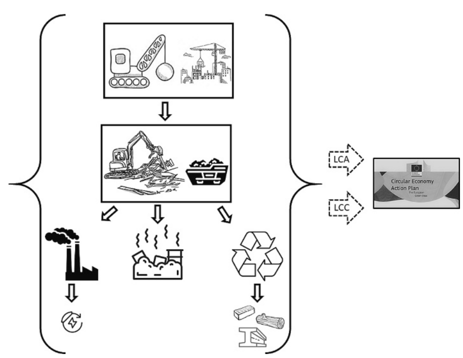
## HIGHLIGHTS

- Construction and demolition sector is crucial to lead the material circularity in the EU
- The environmental impacts and costs of 12 CDW material fractions are assessed
- The best environmental performances are achieved by the most expensive pathways
- The maximum potential for recycling would reduce EU emissions of 33 Mt $CO_2$-eq
- Concrete and bricks have the highest potential in terms of environmental improvements

## GRAPHICAL ABSTRACT




## ARTICLE INFO

Editor: Daniel CW Tsang

*Keywords:*
Infrastructure waste
Circular economy
Recycling
Environmental assessment
Socio-economic assessment
Secondary materials

## ABSTRACT

The recovery rate of construction and demolition waste (CDW) in the European Union (EU) is at 89 % and thus high relative to other waste streams. However, the relatively high figure can be misleading because it typically does not correspond to high-value material recovery but rather "poor" levels of circularity. From a life-cycle perspective, we assess the environmental impacts and costs of 12 CDW material fractions relying on alternative pathways and treatment technologies. The results indicate important trade-offs in the transition towards the circular economy. Indeed, recycling of concrete, bricks, gypsum, and ceramics and tiles represent the best environmental performance but also the most expensive pathway. However, when shifting from landfill to recycling the total societal costs in the EU are reduced mainly due to the lower external costs. Overall, recycling CDW in the EU with advanced technologies would save about 264 kg $CO_2$-eq $t^{-1}$ with a cost of 25 EUR $t^{-1}$. The maximum potential for recycling under current technology in the EU would lead to an annual total reduction of about 33 Mt. of $CO_2$-eq using 2020 as reference year. The fractions with the highest potential for improving current waste management practices in terms of environmental improvements are concrete and bricks. The economic and non-economic barriers for realising this potential at EU level are discussed in relation to the European Green Deal and the EU's circular economy objectives.

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https://doi.org/10.1016/j.scitotenv.2023.168295
Received 31 August 2023; Received in revised form 20 October 2023; Accepted 1 November 2023
Available online 4 November 2023

## 1. Introduction

Globally, the building and construction sector represents the single most important source of human natural resource consumption. Additionally, the construction industry contributes roughly 39 % of energy-related global carbon dioxide emissions (United Nations, 2021) and is responsible for 35 % of global waste disposed in landfills with related environmental consequences (OECD, 2019). Therefore, the construction and demolition sector is integral to improve the sustainability and material circularity of societies globally and in the European Union (EU).

At EU level, to foster material recycling and greater circularity in the sector, a series of legislative acts and non-legislative guidance documents have been published. Notably, the Waste Framework Directive (European Commission, 2008) establishes a target by which a minimum of 70 % (by weight) of non-hazardous construction and demolition waste (CDW), excluding naturally occurring material[1], need to be prepared for re-use, recycled and other materially recovered by 2020. The Directive further sets out requirements for Member States related to waste prevention as well as preparing CDW for re-use and the recycling of CDW. As an example for a non-legislative initiative, the "EU Construction and Demolition Waste Protocol and Guidelines" propose improvements in waste identification, source separation and collection, logistics, processing and quality waste management (European Commission, 2016).

In 2020, the recovery rate of CDW in the EU is relatively high compared to other waste streams, standing at 89 % at EU level (Williams et al., 2020) even though huge variations can be observed between Member States (OECD, 2019). However, this relatively high recovery rate can be misleading because it typically does not correspond to high-value material recovery from CDW. In general, demolition material recovered is repurposed as filler material in road construction or as backfilling material. These represent the most prevalent CDW recovery routes. In a nutshell, these recovered materials do not achieve the necessary technical properties to fulfil the functions for which the original material was designed for. This type of recovery ultimately translates into significantly lower market values of the secondary material entering the market relative to the original material. Thus, the result is a "poor" level of circularity and substitution/displacement of primary material demand, with foregone energy, material and carbon savings.

CDW represents a complex mix of materials that poses challenges to recycling and bringing secondary materials stemming from CDW into the market. On average in the EU, the individual material fractions composing CDW are concrete (24 %), bricks (5 %), ceramics and tiles (1.2 %), metals (4.3 %), plastic (0.2 %), glass (0.2 %), gypsum (1.4 %), wood (2.3 %), insulation (0.3 %), and paper and cardboard (0.2 %), with a remaining high percentage of mixed (59.2 %) and hazardous (1.8 %) waste (Damgaard et al., 2022).

Many of the CDW individual fractions are not available for recycling because of poor demolition and collection practices or, if potentially available, may simply not be recycled in the local market due to economic constraints or market failures. Although recycling technology exists for the majority of CDW material fractions, potential economic and non-economic barriers to recycling CDW in the EU include the following: (i) perceived high cost of recycling relative to other treatment options such as incineration and landfilling, (ii) the lack of local buyers, (iii) regulatory impediments (e.g., chemical composition, safety requirements), and (iv) competition with low-cost products stemming from primary materials that not always internalise their externalities – all these factors contribute to the lack of well-functioning markets for some CDW fractions (European Environment Agency, 2022). While disposal in landfills is often the cheapest option, another common hurdles to recycling CDW in the EU is the lack of confidence in the quality of recycled materials (European Commission, 2016).

These factors lead to limited high-quality recycling of CDW in the EU despite its potential to facilitate reaching the European Green Deal and the EU's circular economy objectives. Among other factors, the potential is particularly high since non-metallic minerals, a key input for the building and construction sector, accounts for 54 % of domestic material consumption in the EU in 2021 and the resulting CDW accounts for 39 % of all waste generate in the EU (Eurostat, 2023). This means that increasing the material efficiency and circularity of the entire life cycle of CDW could significantly contribute to the overall circularity of the EU, while also improving EU's competitiveness both at the macroeconomic and firm levels (Flachenecker, 2018). Greater circularity of materials contained in CDW could also support the EU's strategic autonomy agenda, in particular related to metals, including critical and strategic raw materials, and energy-intensive materials such as cement.

The recent "Circular Builders"-project[2] has shown that improving the circularity of building and construction materials has also a tremendous potential for reducing climate impacts, and supporting innovation and job creation in the EU. To this end, better data and insights in management and recycling options of CDW, via economic and environmental assessments, are necessary to better support policy-making (Damgaard et al., 2022).

Studies assessing the environmental and economic impacts of CDW management have provided key information about the environmental and economic performances associated with the management of CDW. However, most of the studies focused on single fractions of CDW (Chandel and Goyal, 2022; Gebremariam et al., 2021; Faraca et al., 2019a; Faraca et al., 2019b; Ibáñez-Forés et al., 2011; Liikanen et al., 2019; Mir et al., 2022; Rodrigo-Bravo et al., 2022) or were limited to specific Member States or even regions within the EU (Iodice et al., 2021; Borghi et al., 2018; Faraca et al., 2019b; Pantini et al., 2019; Pedreño-Rojas et al., 2020; Stabile et al., 2021; Yılmaz et al., 2022; Suárez Silgado et al., 2018; Coelho and de Brito, 2013; Blengini, 2009; Fraj and Idir, 2017; Di Maria et al., 2018). Moreover, only one study (Iodice et al., 2021) accompanied the LCA with a thorough assessment of financial and external costs, though limited to concrete recycling into aggregates and to a specific geographic region, as highlighted by a recent review (Bayram and Greiff, 2023). The same authors further showed that most of these studies were not performed thorough sensitivity and uncertainty analyses, thus not providing information about the relevance of their input parameters and robustness of their results. Moreover, the full potential of CDW in terms of generated waste flows, both via data recorded and reported by/to authorities (e.g., Eurostat and national statistical offices) as well as via material flow analyses has not been captured yet. We conclude that no study so far has provided a complete overview of the environmental and socio-economic impacts of the management of CDW individual fractions in the EU. Furthermore, no study so far has assessed the recycling potential of each CDW individual fraction towards achieving European Green Deal and the EU's circular economy objectives.

To close the gap, this study assesses the environmental and socio-economic effects of CDW management in the EU, with a focus on recycling of individual material fractions composing CDW. We build upon a previous study where a three-fold exercise was performed (Damgaard et al., 2022): (i) systematic literature review of CDW generation and composition across the EU, (ii) Material Flow Accounting of CDW generation in the EU, with focus on the available potential for each single material fraction, and (iii) a systematic bibliometric analysis to identify current and potential waste management technologies for each material fraction. The data obtained in the previous study is complemented with additional data presented for the first time herein. The Life Cycle

[1] Excluding naturally occurring material defined in category 17 05 04 in the list of waste (i.e., soil and stones other than the ones containing hazardous substances).

[2] https://www.gate21.dk/circular-builders/.

Assessment (LCA) and Life Cycle Costing (LCC) in this study provide new insights for policy. Detailed sensitivity and uncertainty analyses are performed to highlight the importance of process parameters and the variability of the results obtained.

The overarching objectives of this study are to: (i) identify the most effective options, from a life cycle perspective, for the management of CDW in the EU, (ii) quantify the potential environmental impacts and costs resulting from recycling CDW in comparison with business-as-usual management technologies, and (iii) support policymaking in relation to CDW management, in view of possible revisions of the EU Waste Framework Directive. The study's novel findings therefore provide a quantitative basis to inform future circular economy and waste policies in the EU.

## 2. Methods

### 2.1. Life cycle assessment

#### 2.1.1. Functional unit, scope, and system boundaries

The goal of the study is to evaluate the environmental and socio-economic effects of managing individual fractions of CDW in the EU with a focus on recycling and its potential. The Functional Unit (FU) is the management of one tonne of an individual fraction of CDW, which is represented by the input-waste to a set of different management pathways ('LCA scenarios' presented in the next chapter).

CDW is generated along the whole life cycle of construction and demolition activities. It should be noted that the FU corresponds to the reference flow of one tonne including non-targeted material that may end up with the selected fraction due to demolition and collection activities. Based on this FU, each comparison addresses alternative waste management pathways / techniques to handle the same fraction of a CDW stream but rely on technologies that do not necessarily produce the same end products. For example, in the case of recycling and incineration pathways for high calorific value CDW, the latter produces energy and the former produces products such as particle board.

The scope of the investigation of CDW composition in EU27 was primarily based on a thorough literature review, which included more than 90 reports and articles (Damgaard et al., 2022). According to this literature review, the generation of CDW in the EU27 in 2020 amounted to about 397 Mt. when including building and infrastructure waste but excluding soil (Table 1). The available datasets for CDW composition for each country were assessed and processed to obtain a single, representative CDW composition for each analysed country. Following, a Material Flow Analysis of CDW in the EU27 (for more details see Damgaard et al., 2022) led to the final 12 relevant material fractions herein used and their average contribution to the CDW composition for EU 27 (Table 1). We observe that the 12 fractions analysed cover about 85 % of CDW composition in EU27 resulting in about 336 Mt. of CDW (building waste + infrastructure waste). Other fractions were not included here due to different reasons such as their negligible quantity or because they resulted in mixed inert waste.

Potential environmental impacts of the management of the CDW fractions herein studied were evaluated based on LCA, while costs were assessed via Environmental LCC (financial assessment). Societal life cycle costs (SLCC), composed of the sum of internal and external costs (monetised environmental emissions), were also assessed. The assessment was carried out with the software EASETECH (Clavreul et al., 2014) and a consequential approach was applied (Ekvall, 2002).

The system boundaries of the study start at demolition and collection. Further sorting and pre-treatment technologies were included, as well as any of the technologies transforming each fraction of CDW into recyclates, co-products, energy and emissions. Consecutive (cascading) use of the recyclates was not taken into consideration in this study (but implications are discussed for the specific case of wood waste). The assessment considered three waste management treatments for each fraction: recycling (REC), landfill (LAN) and incineration (INC). The system boundary of each LCA scenario (Fig. 1) included all the operations involved in the management of the waste through the specific technology, i.e., a) transport of the input-waste from centralised sorting facilities or collection centres to treatment facilities; b) recycling, landfilling or incineration of the CDW fraction (depending on the scenario); c) further recycling of any non-targeted material fractions separated/recovered during recycling (e.g., metals, paper/cardboard and rubber) or of materials recovered from treatment of bottom ash from incineration (i.e., metals, when included in the input-waste); d) handling of separated non-recyclable material fractions, residues and losses from recycling and residues from energy recovery processes. Depending on the CDW fraction and process, non-recyclable fractions, residues and losses from recycling were assumed to be incinerated with energy recovery or landfilled. The input-waste was assumed to carry no environmental burden from the respective upstream life cycle, following the common "zero-burden" assumption applied in waste management LCA (Ekvall et al., 2007).

**Table 1**

Average contribution of relevant material fraction to the CDW composition for EU27 expressed as % of the total CDW generation in 2020 for buildings and buildings & infrastructure (using 2020 as reference year).

| Material fraction in CDW | Share (% of CDW)[1] | Annual flow buildings[1] (Mt) | Annual flow buildings & infrastructure[2] (Mt) |
|---|---|---|---|
| Concrete | 56.2 | 74.1 | 223.4 |
| Bricks | 6.50 | 8.6 | 25.8 |
| Ceramics & Tiles | 5.56 | 7.3 | 22.1 |
| Steel | 4.89 | 6.1 | 18.6 |
| Glass | 4.04 | 5.3 | 16.0 |
| Wood | 2.91 | 3.8 | 11.5 |
| Aluminium | 1.76 | 2.3 | 6.9 |
| Expanded polystyrene | 0.79 | 0.9 | 2.8 |
| Polyvinyl chloride | 0.79 | 0.9 | 2.8 |
| Gypsum | 0.57 | 0.7 | 2.2 |
| Stone wool | 0.35 | 0.4 | 1.2 |
| Glass wool | 0.35 | 0.4 | 1.2 |
| Others[3] | 15.6 | 20.4 | 61.6 |
| Total | 100 | 131.9 | 397.5 |
| Total (without Others) | 84.4 | 111.5 | 335.9 |

[1] Based on the Material Flow Accounting presented in Damgaard et al. (2022).
[2] Assuming that infrastructure waste has the same composition as building waste.
[3] Cardboard, paper, copper, electronics, other construction minerals, sand, paint and glue,

#### 2.1.2. Waste management scenarios

The waste management scenarios assessed for each CDW fraction rely on alternative pathways and technologies for treatment of individual CDW material fractions. The scenarios are shown in Table 2. Landfill is always considered as a pathway while incineration only for those fractions having a positive calorific value. Reuse was not included in the scope of the study.

It can be argued that the recycling pathways for wood, steel, aluminium PVC, EPS, gypsum as well as all recycling pathways producing recycled aggregates (this applies to concrete, ceramics & tiles, bricks, glass) are business-as-usual recycling technologies. This does not mean that the recycling technology is normally applied, but that it is commercially available at full-scale. We call this group of pathways as 'business-as-usual recycling' (BAU-R). Conversely, recycling of concrete, bricks, ceramics & tiles to produce cement is a technology with lower level of maturity. It can be argued that also closed loop flat glass recycling is not business-as-usual, because of the challenges in material separation and collection. We call this second group of pathways as 'improved recycling' (IMP-R).

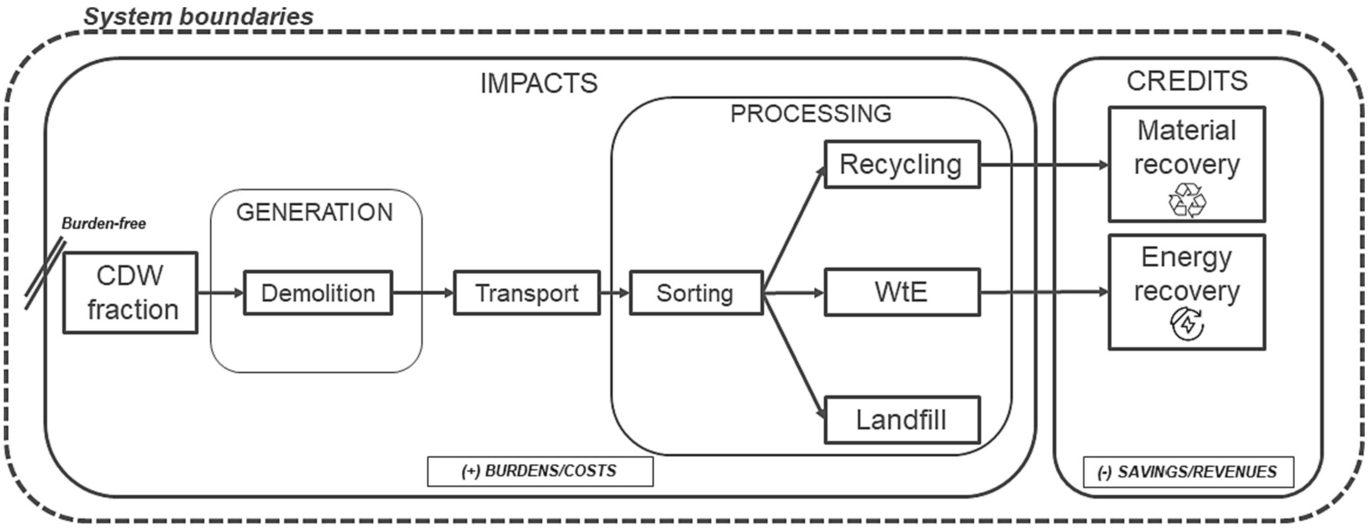


**Fig. 1.** System boundaries for the three classes of waste management treatments. Impacts (left side) include burdens (LCA) and costs (LCC) coming from the waste management of each fraction whereas credits (right side) include savings (LCA) and revenues (LCC) coming from the material and energy recovery.

**Table 2**
Overview of the waste management scenarios assessed for each CDW fraction. Products obtained from recycling processes and products substituted are presented as well as the substitution ratio. Finally the identification code for each scenario is presented. The identification code refers to Fig. 2 and Fig. 3 and is composed of a first set of letters referring to the treatment (e.g., REC = recycling; LAN = landfilling; INC = incineration) and a second set of letters referring to either the eventual material produced in case of recycling (e.g., RA = recycled aggregates; CEM = cement; PBD = particle board; STE = steel; ALU = aluminium).

| Fraction | Technology | Product | Product substituted | Substitution ratio | Code |
|---|---|---|---|---|---|
| Concrete (CON) | Recycling | Recycled aggregates | Sand/Gravel | (0.85:1) | REC-RA |
| | Recycling | Recycled aggregates | Sand/Gravel | (0.85:1) | REC-CEM |
| | | Cementitious material | Cement | (0.71:1) | |
| | Landfill | | | | LAN |
| Wood (WOD) | Recycling | Particle board | Particle board | (1:1) | REC-PBD |
| | Landfill | | | | LAN |
| | Incineration | Electricity/Heat | Electricity/Heat | (1:1) | INC |
| Steel (STE) | Recycling | Iron scrap | Iron ingot | (0.75:1) | REC-STE |
| | Landfill | | | | LAN |
| Aluminium (ALU) | Recycling | Aluminium scrap | Aluminium ingot | (0.85:1) | REC-ALU |
| | Landfill | | | | LAN |
| Plastic PVC (PVC) | Recycling | Polyvinylchloride | Polyvinylchloride | (0.69:1) | REC-PVC |
| | Landfill | | | | LAN |
| | Incineration | Electricity/Heat | Electricity/Heat | (1:1) | INC |
| Plastic EPS (EPS) | Recycling | Polystyrene | Polystyrene | (0.69:1) | REC-EPS |
| | Landfill | | | | LAN |
| | Incineration | Electricity/Heat | Electricity/Heat | (1:1) | INC |
| Gypsum (GYP) | Recycling | Plasterboard | Plasterboard | (0.88:1) | REC-GYP |
| | Landfill | | | | LAN |
| Ceramics & tiles (CT) | Recycling | Recycled aggregates | Sand/Gravel | (0.83:1) | REC-RA |
| | Recycling | Cementitious material | Cement | (0.71:1) | REC-CEM |
| | Landfill | | | | LAN |
| Glass wool (GSW) | Recycling | Glass wool fibres | Virgin rock | (0.83:1) | REC-GLW |
| | Landfill | | | | LAN |
| Stone wool (STW) | Recycling | Stone wool fibres | Virgin rock | (0.83:1) | REC-STW |
| | Landfill | | | | LAN |
| Bricks (BRK) | Recycling | Recycled aggregates | Sand/Gravel | (0.83:1) | REC-RA |
| | Recycling | Cementitious material | Cement | (0.71:1) | REC-CEM |
| | Recycling | Alkali activated blocks | Concrete | (0.65:1) | REC-CON |
| | Landfill | | | | LAN |
| Glass (GLA) | Recycling | Recycled aggregates | Sand/Gravel | (0.83:1) | REC-RA |
| | Recycling | Flat glass | Flat glass | (1:1) | REC-GLA |
| | Landfill | | | | LAN |

### *2.1.3. Inventory data*

To describe the system we used data on: i) CDW composition, ii) CDW flows, iii) energy, electricity, material, fuels and resource provision. For more information about data used for each scenario listed in Table 2 (see *Supplementary Information*, Tables S1, S2, S3). Complementary data for modelling waste treatment technologies were taken from Ecoinvent centre 3.7.1 (Ecoinvent, 2022). Transport distances from demolition site to centralised sorting facilities (Fig. 1) were assumed to be 50 km (Zhang et al., 2020). While shorter or longer distances may occur with different treatment options over the 27 Member States, the same distance was assumed for all scenarios to highlight differences in the performance of individual management technologies. However, we tested this assumption in a sensitivity analysis. Moreover, we distinguished the transportation of each CDW fraction considering the weight and volume transported. This distinction is especially relevant for the different intrinsic features of CDW fractions. Following Lu et al. (2021) the utilization rate (%) was estimated starting from the bulk density of the fraction and the weight and volume of the cargo and finally applied to the transport computation.

For all scenarios, a reference study is used to describe the related process (Table 2). All the CDW fractions are initially demolished (Fig. 1). Two demolition options based on the fractions and respective management technologies were considered, namely traditional and selective demolition. Primary data on these two different demolition options and the substitution rates used for each recycling scenario are provided in the *Supplementary Information* (Tables S1 and S3, respectively).

### 2.1.4. Life cycle impact assessment

The following 16 environmental impact categories that are included in the currently recommended EU methods for Environmental Footprint (European Commission, 2021) were considered in this study (as implemented in the software EASETECH v3.4.0): Climate Change; Ozone Depletion; Human Toxicity cancer; Human Toxicity non-cancer; Particulate Matter; Ionising Radiation; Photochemical Ozone Formation; Acidification; Eutrophication terrestrial; Eutrophication freshwater; Eutrophication marine; Resource Use minerals and metals; Resource Use fossils; Water use; Land use; Ecotoxicity freshwater. For space reasons, only results for Climate Change are presented in the main text of the paper. Results for the remaining impact categories are reported in the *Supplementary Results*.

In the results presented for Climate Change, expressed in kg $CO_2$-eq per FU, positive impact contributions represent burdens to the environment, while negative impact contributions represent savings to the environment. The total net impact of the management of the waste at the level of individual scenarios is calculated as the difference between the burdens of the management pathway and the savings from the substituted products and co-products arising from that pathway. The "total" impact is thus a 'net saving' when negative or a 'net burden' when positive.

## 2.2. Socio-economic assessment

The following socio-economic impact categories were considered in this study: Environmental Life Cycle Cost (ELCC), Societal Life cycle Cost (SLCC), and Total Employment. The ELCC accounts for internal costs (budget costs and transfers) and reflects a traditional financial assessment. Budget costs are costs incurred by the different actors involved in the management chain of the waste (collectors, operators, transporters, etc.), while transfers refer to money redistributed among stakeholders (taxes, subsidies, value added tax - VAT, and fees). The SLCC accounts for internal (covered in ELCC) and external costs associated with environmental emissions, striving to take perspective of the entire costs borne by society. The external costs are not covered by current market prices (i.e., not internalised in the current products and/or services price paid by consumers). To price externalities we here used the shadow prices of environmental emissions by De Bruyn et al. (2018) as suggested by the official EC guidelines for impact assessment (European Commission, 2023a). Yet, the $CO_2$ price was corrected to 100 EUR/t based on a recent EC Commission updates and recommendation (van Essen et al., 2019). Notice that only the externalities associated with environmental emissions to soil/water/air are included in De Bruyn et al. (2018) while other external costs (e.g., convenience for sorting, gender or other inequality issues, other nuisance or disamenities) are here not accounted for. For this reason, recent studies suggest a different name for this impact category (full environmental life cycle costing). The ELCC and SLCC share the same object, scope, functional unit, and system boundaries of the LCA and were facilitated with the software EASETECH v3.4.4 (Clavreul et al., 2014), following state-of-the-art methodology as suggested in recent works (Martinez-Sanchez et al., 2015). Data used for the socio-economic assessment are reported in the *Supplementary Information* (Table S4). For brevity, only the results for the Environmental and Societal costs are presented in the main text of the paper. Results for employment, alongside a further disaggregation of the costs into OPEX, CAPEX, taxes, and externalities are reported in the *Supplementary Results*. In the results presented here for ELCC and SLCC, expressed in EUR per FU, positive contributions reflect financial costs, while negative contributions reflect revenues. The "total" cost of the management of the waste at the scenario level is calculated as the difference between the sum of the costs associated to the management pathway and the revenues obtained from selling any products and co-products arising from that pathway. A negative "total" indicates a net income for the scenario.

## 2.3. Overall potential for CDW recycling in the EU

To capture the recycling potential for CDW in the EU as a whole, results for each impact category analysed were also calculated per tonne of CDW. For the purpose of this calculation, the Functional Unit (FU) is not anymore one tonne of an individual material fraction (e.g., 1 t of wood) but becomes the management of one tonne of CDW in the EU as a whole (i.e., total CDW). To this end, we applied the individual material shares summarised in Table 1 expressing an average CDW composition for EU27.

Two scenarios were considered to analyse the potential of the CDW management at EU level resulting from the effect of an improvement of recycling in the EU. First, a baseline (BL) scenario was calculated representing the CDW management status for each fraction according to the waste management pathways for individual material fractions of CDW in the EU27. This is estimated based on the available techno-scientific literature and reported in the *Supplementary Information* (Table S5). Second, in order to calculate the associated impacts at EU27 level, the material flows reported in Table 1 are multiplied by the treatment shares reported for each material fraction in Table S5, as well as by the impacts calculated in Section 3.1 (for climate change) and 3.2 (for environmental costs) per tonne of managed waste material fraction. In the BL scenario it is assumed that recycling is only business-as-usual (mainly production of recycled aggregates). A Maximum Potential (MP) scenario capturing the total potential of recycling when considering technical losses at sorting and recycling is then defined (see *Supplementary Information*, Table S5). It means that the MP scenario represents the maximum recycling potential where: i) landfill (of mineral waste) and incineration (of waste with calorific value) are set to a minimum respecting the minimum technical loss from sorting and recycling; ii) improved recycling (IMP-R) replaces business-as-usual recycling (BAU-R).

The latter means that for the fractions, such as metals, plastics, gypsum and mineral wool that have only one recycling pathway available, the amount of waste sent to landfill or incineration in the BL scenario is shifted to recycling in the MP scenario. Finally, a Marginal Abatement Cost Curve (MACC) chart is used to show the effect of shifting from a BL to a MP scenario.

## 2.4. Uncertainty and sensitivity analysis

The uncertainty was propagated analytically by combining parameter and data quality uncertainty. The total uncertainty of a parameter (i.e., of a single data point that is input to the model) is obtained considering both the uncertainty related to the intrinsic variation of the value (e.g., the electricity recovery efficiency at incinerators in EU has a certain range of variation around a mean or likely value) and an additional uncertainty related to the quality of the data itself. Parameter uncertainty was addressed using analytical propagation following the approach suggested in Bisinella et al. (2018). Conservatively, we used a uniform distribution (because the type of distribution was unknown for most data; except for incineration electricity recovery for which a triangular is normally considered better representative) and the range assigned to the parameters is either based on literature, when available, or assumed to be +/− 20 % following previous studies (Bisinella et al., 2018; Damgaard et al., 2022). While we are aware that the latter is an assumption, no parametric ranges are available for most data. The additional uncertainty on data quality is quantified by means of the Pedigree Matrix using the approach suggested by Ciroth et al., 2016). For the Pedigree Matrix calculation, parameters are grouped in clusters and valued according to five indicators based on the scope of the study: reliability, completeness, temporal correlation, geographical correlation, and further technological correlation, each with a score of 1 to 5. A score of 1 means that the data is of high quality with regard to that particular indicator (e.g. 'data from the area under study' for the

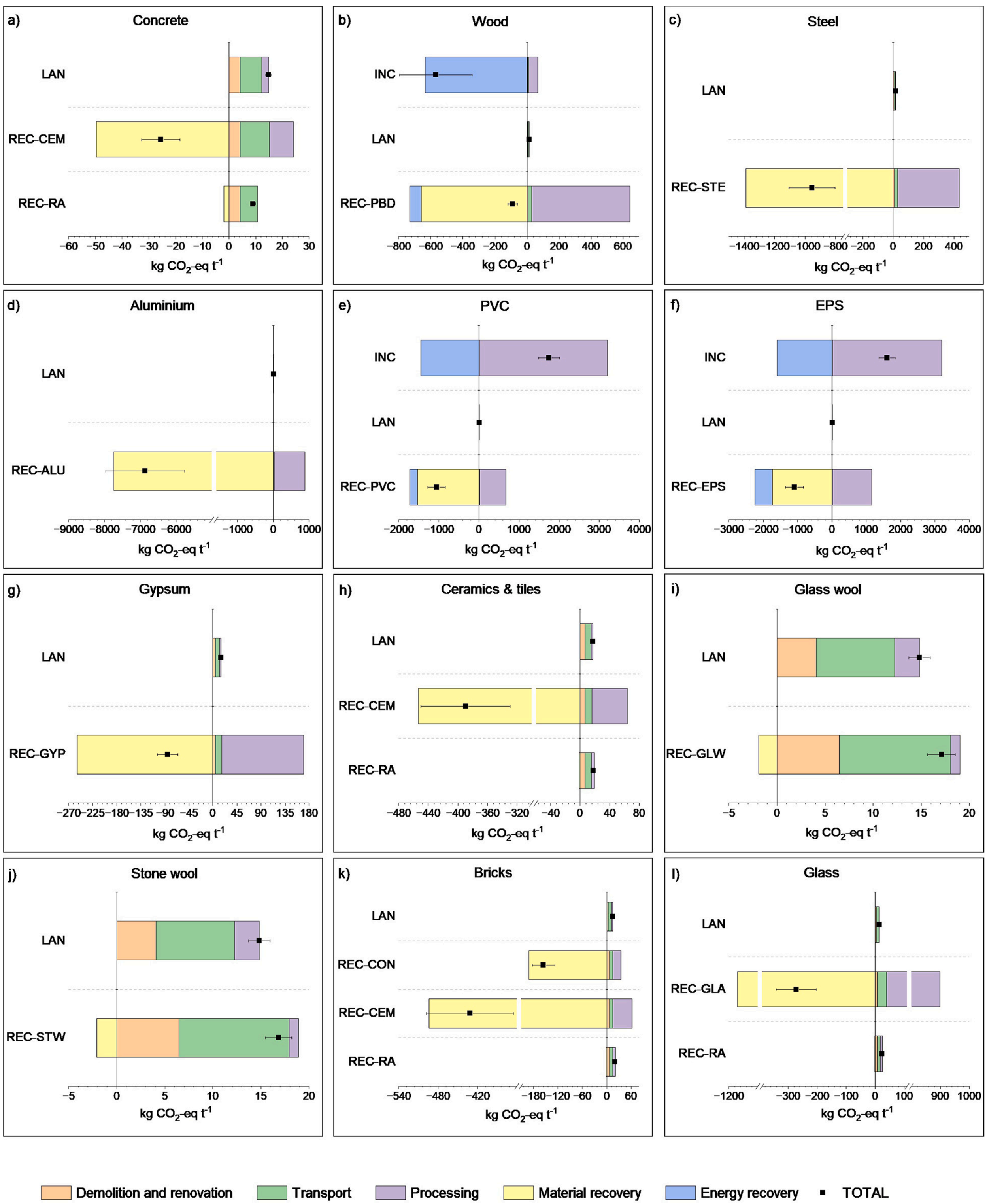


**Fig. 2.** Climate Change results for the management of each individual waste material fraction of CDW: a) concrete waste; b) wood waste; c) steel waste; d) aluminium waste; e) PVC waste; f) EPS waste; g) gypsum waste; h) ceramic & tiles waste; i) glass wool waste; j) stone wool waste; k) bricks waste; l) glass waste. Results are expressed as kg $CO_2$-eq per tonne of waste material fraction.

indicator geographical correlation); a score of 5 means the data quality for that indicator is low (e.g. 'non-qualified estimate' for the indicator reliability). Each combination of indicator and score gives an uncertainty factor.

## 3. Results

### 3.1. Climate change

On Climate Change, recycling performs better than landfilling and incineration for most of the CDW fractions (Fig. 2). Exceptions are wood and stone/glass wool. For wood (Fig. 2b), incineration performs better than recycling owing to the high energy recovery and low GHG emissions because of the carbon neutrality assumption for C-biogenic in wood. For stone wool (Fig. 2i) and glass wool (Fig. 2j), recycling scenarios offer limited GHG savings as the recycling scenarios only achieve limited material savings. We in general observe a significant difference between recycling to aggregates or to cement, concrete or other closed-loop recycling. In particular, we find that, except for concrete production, recycling to produce recycled aggregates is not sufficiently environmentally beneficial when compared with landfilling. This means that the credits obtained from virgin material substitution are less than the burdens of collection, sorting, transport and recycling operations. For instance, recycling of concrete waste to cement (Fig. 2a) records a total net GHG saving of 26 ± 7 kg $CO_2$-eq $t^{-1}$ that is substantially higher than the net burden achieved through recycling concrete to aggregates (9 kg $CO_2$-eq $t^{-1}$). Another example is the closed-loop recycling to produce flat glass from glass waste that achieves a total net saving of 272 ± 69 kg $CO_2$-eq $t^{-1}$ (Fig. 2l). This is substantially higher than the net GHG burden achieved through recycling glass to aggregates (23 kg $CO_2$-eq $t^{-1}$). The same relationships are present for ceramics & tiles (Fig. 2h) and bricks (Fig. 2k) where recycling to aggregates does not result in substantial GHG savings relative to landfilling or incineration.

The highest GHG savings are achieved by recycling metals where aluminium and steel contribute with 6862 ± 1102 kg $CO_2$-eq $t^{-1}$ (Fig. 2d) and 954 ± 151 kg $CO_2$-eq $t^{-1}$ (Fig. 2c) saved, respectively. This outcome is expected, due to the carbon-intensive production process of these materials and the consequent substantial energy savings in recycling them. Aluminium and steel are already routinely separated from CDW and recycled to a large extent owing to their relatively high market value. Recycling of EPS (Fig. 2f) and PVC (Fig. 2e) save 1088 ± 256 and 1058 ± 216 kg $CO_2$-eq $t^{-1}$, respectively resulting in a significant reduction of GHG emissions relative to landfill (15 kg $CO_2$-eq $t^{-1}$) and incineration (1605 ± 240 kg CO2-eq t-1 and 1746 ± 258 kg CO2-eq t-1, respectively). Recycling gypsum to plasterboard (saving 85 ± 19 kg CO2-eq t-1) performs better than landfilling (Fig. 2g). Recycling bricks to cement or concrete also generates important net GHG savings (431 ± 66 kg CO2-eq t-1 and 155 ± 27 kg CO2-eq t-1, respectively) relative to landfilling (15 kg CO2-eq t-1). It should be noted that low TRL is associated with bricks/ceramics recycling to cement/concrete and therefore these scenarios result to be uncertain.

As for the contributions to the impact, in recycling scenarios producing recycled aggregates the contribution of the processing is minor, and in many cases lower than transport (see concrete, ceramics & tiles, bricks and glass), which becomes the most important parameter in the management scenario. This trend is reversed in the more advanced recycling scenarios. Indeed, the sensitivity analysis reveals that doubling the distance to treatment or disposal does not affect the ranking of the scenarios, across all the material fractions investigated, however the pathways producing recycled aggregates (REC-RA) are more sensitive to the distance. This is due to a lower relevance of other parameters such as material substitution and processing for this specific recycling pathway. Notice that energy substitution also is important, both when incineration is applied and also in selected recycling pathways (for wood, EPS and PVC) due to the fact that sorting and recycling losses are incinerated. However, it should be noticed that notwithstanding the energy recovery savings, the overall GHG balance for direct incineration of plastics is always a net burden on Climate Change (GHG emissions at stack are higher than GHG savings). The GHG contribution of demolition is negligible compared to the others. Concerning the substitutability factors, the sensitivity analysis reveals that most of the CDW fractions are extremely sensitive to these parameters. However, pathways leading to recycled aggregates (REC-RA), result to be less sensitive to the substitution factor. This is due to a lower relevance of this parameter for these specific recycling pathways, as the saving contribution from substitution of natural aggregate is indeed very limited, making processing and transport relatively more important.

### 3.2. Environmental life cycle costs

We find that recycling of concrete (Fig. 3a), ceramics & tiles (Fig. 3h) and bricks (Fig. 3k) to cement is definitely more expensive than landfilling (notice that an EU average landfill tax of 19 EUR $t^{-1}$ is included). Also, the cost of closed-loop recycling of gypsum to plasterboard and glass to flat glass is comparable to or more expensive than landfilling (Fig. 2g and l, respectively). Recycling of steel (Fig. 2c) and aluminium (Fig. 2d) stand out as lower cost and clearly favourable to landfilling as expected. For plastics, the recycling of PVC (net income of 431 ± 150 EUR $t^{-1}$) appears to be less expensive than landfilling (45 EUR $t^{-1}$) but more expensive than incineration (net income of 839 ± 245 EUR $t^{-1}$) when using the higher revenues obtained from energy recovery (Fig. 2e). Recycling of EPS (net income of 674 ± 135 EUR $t^{-1}$) is preferred to landfilling (45 EUR $t^{-1}$) but still more expensive than incineration (net income of 956 ± 255 EUR $t^{-1}$). It should be noted that, even when considering recycling scenarios, inevitably a part of the revenues comes from energy recovery of the recycling residues incinerated (Fig. 2f). Recycling of stone wool (Fig. 2j) and glass wool (Fig. 2i) is more expensive than landfilling because of relatively low revenues. Across all scenarios investigated, the most important contribution to the costs is the recycling process itself. The tax for landfilling is the most important contribution to the cost for this treatment. Revenues come from sale of materials and energy, the latter when CDW fractions are diverted to incineration.

Findings reveal that a trade-off between benefits for climate change (Fig. 2) and economic (Fig. 3) impacts currently exists at the EU level. For example, recycling of materials such as concrete, bricks, ceramics & tiles, gypsum where the best environmental performance is achieved by the worst economic performances in terms of environmental costs.

### 3.3. Potential of CDW recycling in the EU

Fig. 4a reveals that overall, recycling CDW would have the potential to save about 264 ± 51 kg $CO_2$-eq $t^{-1}$ with advanced technologies (IMP-R) and 181 ± 28 kg $CO_2$-eq $t^{-1}$ with business-as-usual (BAU-R). We observe a difference of about 83 kg $CO_2$-eq $t^{-1}$ CDW between the total savings in the IMP-R and BAU-R scenarios. It should be noted that for steel, aluminium, EPS, PVC, stone wool and glass wool only one recycling pathway is available and therefore BAU-R and IMP-R result in the same outcome for these fractions. However, for the other fractions, Fig. 4 shows that the type of recycling pathway plays a key role in the potential emissions savings. In particular, significant differences between the IMP-R and BAU-R and landfilling are observed for the GHG savings contribution from bricks (−28.0, +1.31 and +0.96 kg $CO_2$-eq $t^{-1}$ CDW, respectively), concrete (−14.4, +5.0 and +8.3 kg $CO_2$-eq per $t^{-1}$ CDW, respectively) and ceramic & tiles (−21.7, +1.0 and +0.96 kg $CO_2$-eq $t^{-1}$ CDW, respectively). It should be noted that when shifting from landfilling to the IMP-R or BAU-R, an additional impact saved from the landfill avoidance should be taken into account.

Looking at environmental costs, Fig. 4b also shows that in the EU, the implementation of the IMP-R would cost 25 ± 7 EUR per tonne of CDW, which is higher than the cost associated with the BAU-R (4.6 ± 1.1 EUR per tonne of CDW). We find that the BAU-R has the potential to cut

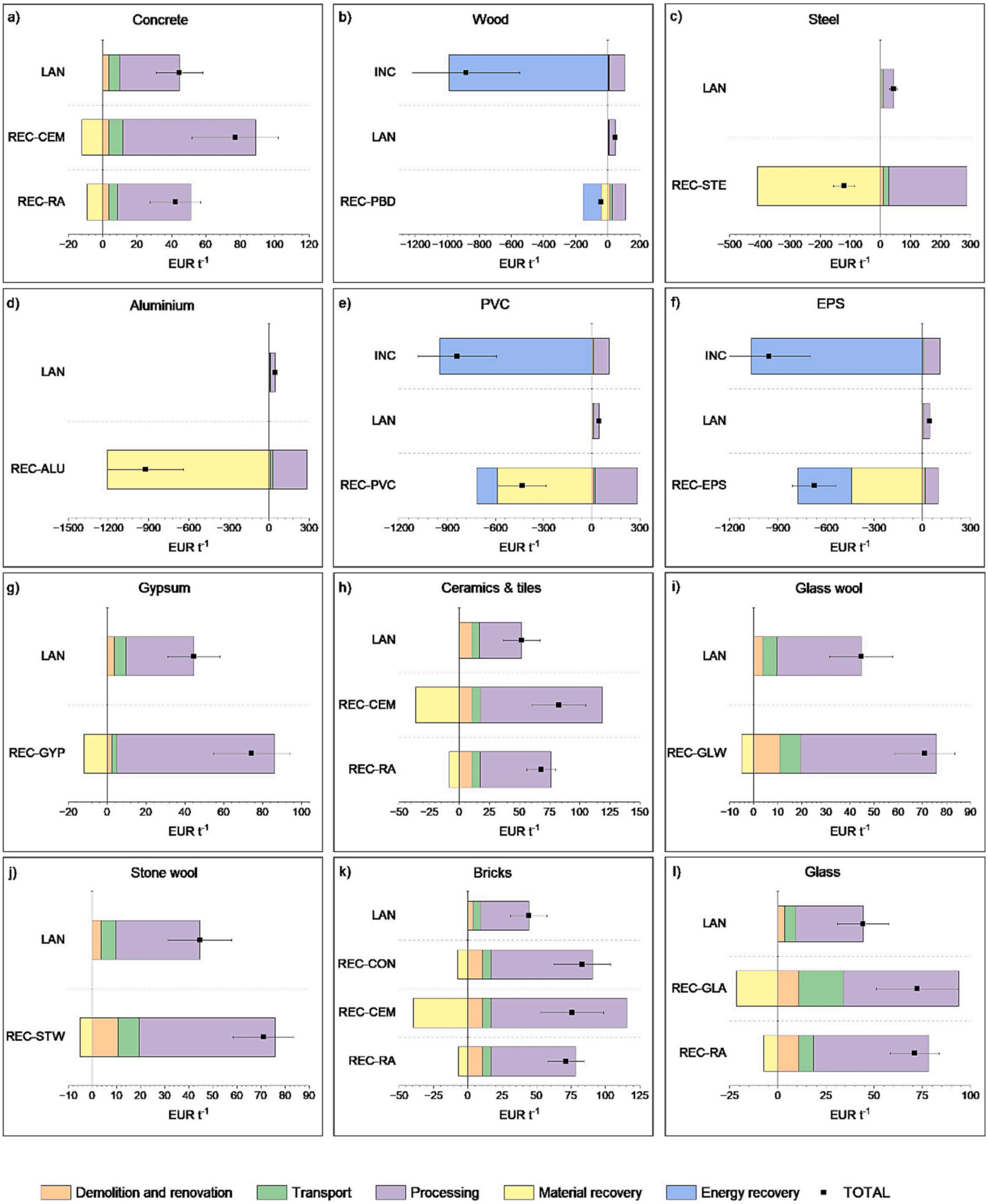


**Fig. 3.** Results for the Environmental Life Cycle Costs category for each individual waste material fractions of CDW: a) concrete waste; b) wood waste; c) steel waste; d) aluminium waste; e) PVC waste; f) EPS waste; g) gypsum waste; h) ceramics & tiles waste; i) glass wool waste; j) stone wool waste; k) bricks waste; l) glass waste. Results are expressed as EUR per tonne of waste material fraction.

environmental costs to about a fifth compared to the IMP-R. Hence, the type of recycling pathways significantly affects the environmental costs. It should be noted that when shifting from landfilling to the IMP-R or BAU-R, an additional impact saved (both environmental and economic) from the landfill avoidance should be taken into account.

Results show that a significant difference between the IMP-R and BAU-R scenarios exists for concrete (43 versus 24 EUR $t^{-1}$ CDW, respectively) while the other fractions show smaller differences. The environmental costs of landfill shown in Fig. 4b include the application of the landfill tax considered in this study (19 EUR $t^{-1}$ CDW). Finally, concerning societal costs, Fig. 4c reveals a total potential of about 14 to 26 EUR saved per tonne of fraction of CDW in the EU for the IMP-R and

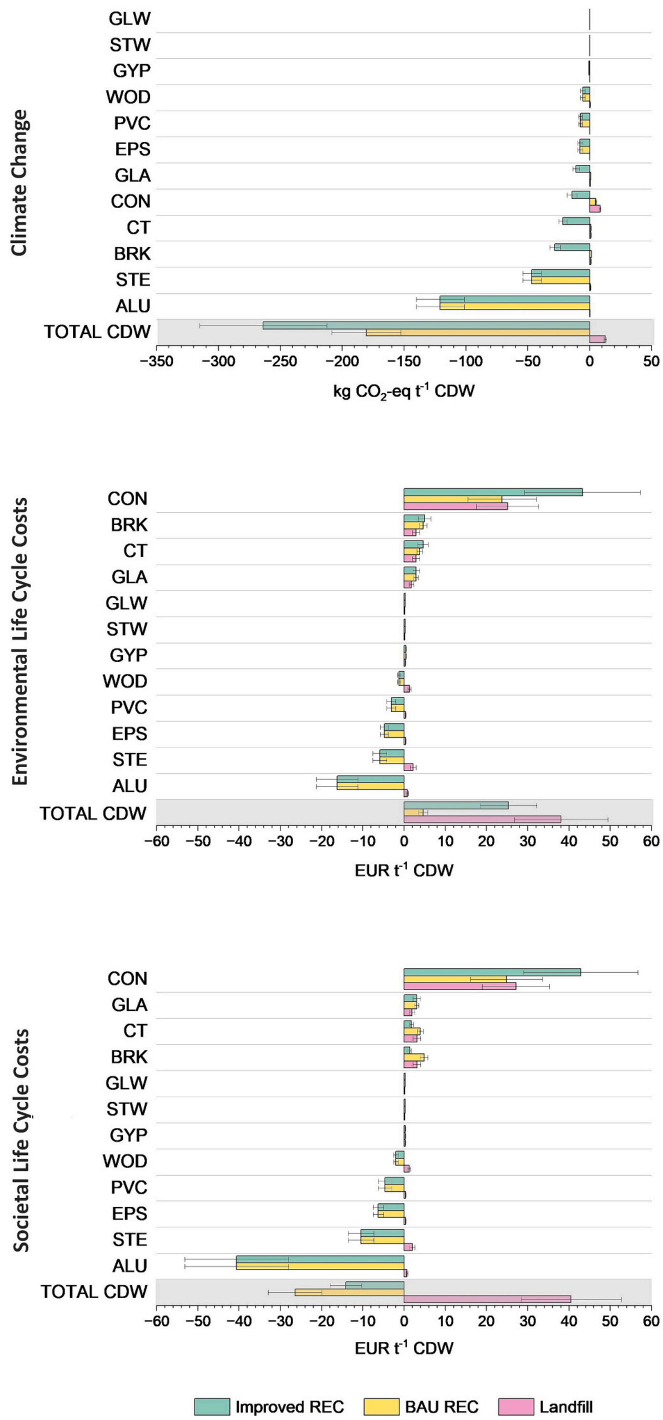


**Fig. 4.** Results regarding potential impacts on a) Climate Change, b) Environmental Life Cycle Costs and c) Societal Life Cycle Costs for concrete waste, wood waste, steel waste, aluminium waste, PVC waste, EPS waste, gypsum waste, ceramics & tiles waste, glass wool waste, stone wool waste, bricks waste, glass waste. Results expressed as kg $CO_2$-eq per one tonne of CDW. Share of waste fractions are based on Table 1.

BAU-R, respectively. Because of the lower environmental costs, the BAU-R has the potential to reduce societal costs more than the IMP-R, although the IMP-R has the potential to reduce more external costs (see *Supplementary Results*). The high societal costs of landfill shown in Fig. 4c are mainly due to the absence of revenues and the application of the landfill tax, here accounted as an internal cost. Similarly to the other impacts, it should be noted that when shifting from landfilling to the IMP-R or BAU-R, an additional societal cost saved from the landfill avoidance should be taken into account.

Fig. 5 shows the Marginal abatement cost curve (MACC) chart, presenting the costs or savings expected from different actions in relation to the cost of GHG abatement per unit. In this case the action is represented by the implementation of the MP scenario intended as shift from the BL scenario (see *Supplementary Information*, Table S5). Boxes above the axis represent costs. For instance, for concrete, a cost of 0.98 EUR per kg of $CO_2$-eq reduced is identified. Below the axes, each box represents savings. For instance, for aluminium, a saving of 0.14 EUR per kg of $CO_2$-eq reduced is identified. The width of the boxes indicates the potential volume of reduction, expressed as Mt. of $CO_2$-eq. The width of the MACC strictly relies on the total amount of each CDW fraction (see Table 1 - annual flow for buildings & infrastructure). For instance, Fig. 5 reveals that the MP scenario, with respect to the BL one, could save 6.6 Mt. of $CO_2$-eq in 2020, for concrete. Bricks and ceramics & tiles also show substantial savings of emissions in the MP scenario (9.9 Mt. of $CO_2$-eq and 7.6 Mt. of $CO_2$-eq in 2020, respectively) with a very low positive cost (0.02 EUR per kg of $CO_2$-eq and 0.04 EUR per kg of $CO_2$-eq, respectively). Concerning wood, a slight increase of emissions (0.1 Mt. of $CO_2$-eq) is obtained with a negative cost (about 6 EUR per kg of $CO_2$-eq). This is mainly due to specific assumption made for wood which are discussed in the next caption. We may deduce that recycling PVC, EPS, steel and aluminium offset a small volume of emissions at a negative cost in comparison to the other fractions. Oppositely, concrete recycling will reduce a large volume of emissions at a positive cost (higher that other fractions). Overall, according to the total CDW generation shown in Table 1, with respect to the BL scenario, the MP scenario would lead to a total reduction of about 33 Mt. of $CO_2$-eq in 2020. The lowest cost options with greatest benefit for GHG mitigation among the recycling options is recycling PVC and Aluminium. The cost of recycling CDW as GHG mitigation cost may be compared to the traded cost of carbon in the EU. The price of tradeable permits for 1 t of carbon under the EU Emissions Trading System (EUTS) is approximately EUR 92 (0.092 EUR per kg $CO_2$) at the time of this writing and the value used in our societal costing is 100 EUR per tonne (0.1 EUR per kg $CO_2$) as recommended by European Commission (European Commission, 2021).

## 4. Discussion

### *4.1. The impact of CDW management*

This study has shown that recycling CDW, in particular advanced technologies (producing cement, concrete or other closed-loop recycled) perform better than landfilling, incineration, and business-as-usual recycling for almost all of the individual CDW fractions in terms of climate change mitigation impact (Fig. 2). As for the other environmental impact categories, they generally follow a similar trend compared to climate change mitigation with respect to the ranking of the management scenarios (*Supplementary Results*). The main channel for these results is secondary materials resulting from recycling substituting high-value primary materials and thereby generating associated environmental savings. This is therefore an important indication that on average advanced recycling technologies applied to CDW are preferable from a climate/environmental perspective over incineration or landfilling. It reiterates the relevance of the ongoing discussion about the need of a clear definition of what is intended for quality of recycling and a framework for operationalising it (Roosen et al., 2023; Tonini et al., 2022).

At the same time for wood waste, incineration may be competitive with (or preferred to) recycling because of the GHG savings from energy recovery. This occurs due to two assumptions: (i) C-neutrality (biogenic-$CO_2$ released from wood is assigned a characterisation factor of zero on climate change), and (ii) we do not consider subsequent life cycles (cascading uses), but only the first life cycle. The first assumption follows current recommendations and the current mainstream approach

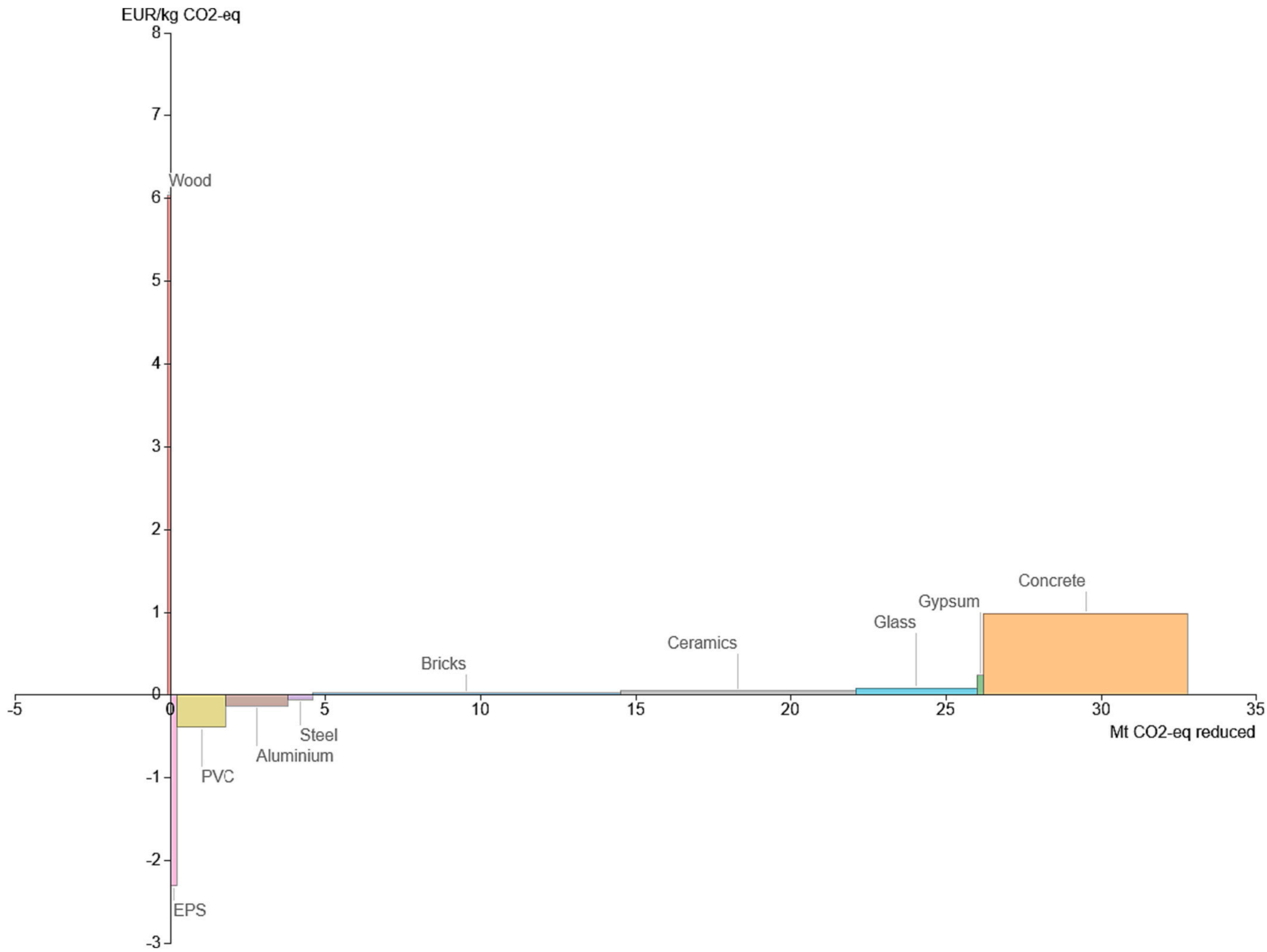


**Fig. 5.** Marginal abatement cost curve (MACC) showing the cumulative reduction potential in terms of Mt. of $CO_2$-eq (x axis) and the cost-effectiveness (EUR per kg of $CO_2$-eq) of implementing the MP scenario intended as shift from the BL scenario. That is, it represents the effect of shifting from the current situation to the maximum potential for CDW recycling in the EU. The curve shape is created by ordering the lowest cost to the left, to highest cost on the right.
*values for stone wool and glass wool are negligible.

(Brandão et al., 2013; Leinonen, 2022), even though the issue is debated among scholars (Peng et al., 2023). Adding the contribution related to indirect land use change (iLUC) on top of our results (e.g., 0.05–0.32 kg $CO_2$ $kg^{-1}$ wood; Schmidt and Brandão, 2013; Faraca et al., 2019a), to account for the fact that wood carbon is not a short-carbon-cycle, would not be sufficient to change the ranking between incineration and landfilling. As for the second assumption, recent studies have shown that when accounting for the second and third life cycles, then indeed wood recycling becomes favourable over direct incineration (Damgaard et al., 2022; Faraca et al., 2019b). This result for wood does, however, not apply to plastics, where fossil $CO_2$ emissions from incineration are not counterbalanced by the energy recovery savings (EPS and PVC; Fig. 2).

In terms of costs and in particular for advanced pathways, recycling is generally more expensive than landfilling and incineration. When for the same fraction the IMP-R is compared with the BAU-R, the former is always more expensive. Environmental costs for recycling stem mainly from the recycling process itself, sorting and selective demolition (when applicable). The exception of metals (Fig. 3) can be explained through the high revenues from secondary materials generated such as aluminium scrap and iron scrap. Instead, a competition between the BAU-R and landfill or incineration is observed. Even, for some BAU-R pathways, comparable costs with landfill are observed (concrete, ceramics and tiles, bricks, and glass).

This suggests an important trade-off between environmental and economic considerations in view of the transition towards a circular economy in the EU. This is an important indication that on average advanced recycling technologies applied to CDW are preferable from a climate/environmental perspective over incineration or landfilling, which is in line with the European Green Deal, the EU's circular economy objectives and the waste hierarchy set out by the Waste Framework Directive. At the same time, the relative prices of recycling CDW compared to incineration or landfilling at the level of economic actors might currently be too high for the general uptake of the existing advanced recycling technologies across the EU. From a societal perspective, however, the move towards the maximum potential for recycling under current technology in the EU would lead to an annual total reduction of about 33 Mt. of $CO_2$-eq using 2020 as reference year and result in positive net costs of around EUR 6.4 billion.

The finding that the environmental life-cycle costs and societal life-cycle costs differ in their result may be an indication for market failures for breach the assumptions underlying the First Fundamental Welfare Theorem that states that the competitive equilibrium where supply equals demand, maximizes social efficiency. Since the Second Welfare Theorem states that in these situations market interventions can theoretically remedy the given economic allocation, policy intervention may be justified.

### 4.2. Barriers to take off at the EU level

For the majority of CDW material fractions, technologies for recycling exist today. The fact that recycling in several Member States is not widespread is due to several barriers, which can be divided into four main categories: regulatory, technical, economic, and awareness (Damgaard et al., 2022).

- *Regulatory barriers*: Every Member State has its own regulations and legal frameworks that influence the potential for recycling. For instance, while it is ascertained that in many cases, selective demolition provides substantial environmental benefits compared to traditional demolition due to the potential for material recycling (Pantini and Rigamonti, 2020); however, no strict requirements for selective demolition of CDW are currently in place across the EU. Indeed, the 2018 amendment of Article 11 in the Waste Framework Directive encouraged Member States to take measures to promote selective demolition. The recently adopted EU Environmental Taxonomy includes selective demolition as one of the criteria for an activity to be considered environmentally sustainable (European Commission, 2023b). Consequently, in the future, a more widespread use of selective demolition could be foreseen, even if the detailed implementation is left to Member States.
- *Technical barriers*: From a technical point of view, a barrier may be represented by the complex composition of CDW which is often contaminated with hazardous substances, making the process of separate collection, transportation and treatment challenging. Another barrier is identified in the buildings not being designed for deconstruction, which refers to the design of a building with the intent to manage its end-of-life more efficiently. The process is intended to ensure the easy disassembly of buildings in order to reduce or even prevent waste generation and maximise the recovery of high value secondary building components and materials for reuse and recycling.
- *Economic barriers*: Williams et al. (2020) concludes that economic barriers, mainly cost of recycling of CDW, prevent widespread uptake of recycling. Our results indicate that IMP-R pathways such as processes substituting CDW fractions with cement, concrete or closed-loop recycling, have the potential to maximise recovery while minimising GHG impacts (Fig. 2). But, they are still relatively more expensive compared to landfilling and incineration (Fig. 3) to take hold in the market and the technologies are still not at full-scale maturity (low TRL). This is especially true for relevant fractions in the EU CDW composition such as concrete and bricks (Table 1).

  The presence of sufficient levels of landfill and incineration taxes to tilt the relative prices represent an important lever to promote CDW recycling (European Commission, 2017). There is a range of landfill and incineration taxes applied to CDW among Member States. It should be noted that in the Member State with the highest landfill taxes such as the Netherlands, the level of recovery/recycling of CDW is also the highest (Luciano et al., 2022). In our analysis, an average EU landfill tax is used for all scenarios (19 EUR $t^{-1}$ waste). According to our results, this value is presumably still too low and does not provide significant incentives for recycling. Most of the IMP-R scenarios, especially those generating higher environmental impact savings, are still more expensive than landfilling (Fig. 4). As for incineration, we applied an average EU incineration tax (29 EUR $t^{-1}$ waste; Albizzati et al., 2023). PVC and wood waste recycling still are more expensive than incineration (Fig. 4). It is clear that, notably for PVC (and plastics broadly), the current incineration tax is significantly too low to incentivise recycling over incineration. This is even more relevant with the upcoming development of chemical recycling options which are expected to contribute to significantly improve plastic circularity (Lase et al., 2023).

  Taxes on landfill and incineration of CDW represent an advance towards internalisation of negative environmental externalities, thus raising the prices of those treatments and incentivising recycling. Although a quantitative analysis of their impact at the EU level is difficult due to their heterogeneity across Member States (and sometimes even regions within Member States), this analysis suggests that taxes on landfill and incineration are sensible instruments that can make recycling economically competitive with landfill and incineration.
- *Awareness barriers:* The lack of coordination between actors and know-how belongs to the awareness barriers. The lack of knowledge of the techniques of recycling and their potential benefits is currently an obstacle in some Member States as recycling information can be complex and often confusing, which may subsequently reduce participation in any waste recycling schemes (Oke and Kruijsen, 2016).

### 4.3. Potential contribution of CDW towards the European green Deal and the EU's circular economy objectives

The 2020 Circular Economy Action Plan is the strategic document of the European Commission to implement the European Green Deal in terms of its circular economy objectives (European Commission, 2020a). The plan considers construction and buildings, and with it CDW, as a key value chain due to the potential to produce secondary materials from its single fractions and its associated potential to improve environmental quality. Further, the 2020 building renovation wave for Europe theorises that "applying circularity principles to building renovation will reduce materials-related greenhouse gas emissions for buildings" (European Commission, 2020b). Directive 2008/98/EC and following amendment of this Directive in 2018, set a minimum of 70 % as target to recycle or recover of CDW by 2020. The Directive also sets out in Article 11(6) a review clause under which "[…] the Commission shall consider the setting of preparing for re-use and recycling targets for construction and demolition waste and its material-specific fractions […]". Additionally, the proposed criteria for a substantial contribution to the circular economy objective under the EU Environmental Taxonomy sets out ambitious levels of CDW preparing for re-use or recycling, without taking into account backfilling (European Commission, 2023b).

The potential of CDW to contribute to the achievement of these ambitions is particularly high since non-metallic minerals, a key input for the building and construction sector, accounts for 54 % of domestic material consumption in the EU in 2021 and the resulting CDW accounts for ca. 36 % of all waste generate in the EU (Eurostat, 2023). This means that increasing the material efficiency and circularity of the entire life cycle of CDW could significantly contribute to the overall circularity of the EU. It is also an area with currently limited EU harmonised rules compared to other waste streams, often relying on non-legislative and non-binding guidance documents. This may also partly reflect limited EU-level competences and unanimity requirements across Member States when it comes to taxation, including on incineration or landfilling.

Another EU policy objective to which improved CDW management could contribute to is strategic autonomy. This is particularly relevant for metals, including critical and strategic raw materials, and energy-intensive materials – with the aim to reduce energy use and imports – such as cement.

However, as documented by the literature, while the recovery of CDW is high, the actual substitution of primary material and thus the circularity of CDW via IMP-R remains still low (Williams et al., 2020) and barriers obstruct alternative pathways. Except for metals, most of the remaining materials that are recovered after demolition end up for use as road sub-base or environmental landscaping/filling replacing natural aggregates, here identified as the BAU-R pathways.

This study provides a detailed assessment of the potential for CDW recycling in the EU, thus supporting policy in pursuing an improved management of CDW in view of the aforementioned objectives. The results presented can be used as a basis to identify key material fractions and recycling pathways, quantifying their environmental impacts and costs in a systematic manner. In particular, the diverging findings between environmental and societal life-cycle costs reveal the need for policy to consider intervening to achieve the goals set out at EU level. It further outlines the barriers that currently obstruct progress towards scaling up recycling of CDW across the EU, which can be used as a starting point for policy to consider instruments to address them. As such, this study could serve as a basis for EU-level policy discussions on how to effectively support and guide the sector towards greater circularity.

## 5. Conclusions

This study provides a detailed techno-economic data and environmental assessment of CDW management in the EU with the aim of identifying the CDW fractions in which improved management via recycling may achieve the largest potential environmental benefits, with a focus on GHG emission savings. Overall, the analysis has provided three crucial results for the management of individual CDW material fractions: (i) environmental performances for 16 environmental impact categories, (ii) economic feasibility in terms of environmental and societal life-cycle costs, and (iii) information about GHG abatement costs when moving to recycling from current situation. With this evidence, we conclude that moving to recycling under currently existing technology in the EU would provide GHG emission savings and economic benefits from a societal perspective.

For the majority of CDW material fractions, recycling options do exist today. The fact that the advanced recycling technologies in several Member States are not widespread is due to several economic and non-economic barriers, which are herein discussed. In particular, this paper has highlighted as taxes on landfill and incineration may represent an important lever to promote CDW recycling in the EU. Our analysis has showed that such tools can make recycling economically competitive with other waste management technologies thus balancing socio-economic and environmental benefits.

The main limitation of the study lies in the data elaboration and assumptions used to determine the average CDW composition in the EU. Indeed, the 12 fractions analysed through a systematic literature review, have covered about 85 % of CDW composition in the EU with a remaining 15 % of mixed CDW. In this context, further research is needed primarily to achieve better estimates of material flows in the EU, thus revealing the full potential of CDW management in the EU. Although re-use and preparing for re-use operations were not included in the study scope, they could also represent a viable solution to improve the environmental performance of CDW management that should be investigated in the future. In this analysis, the common "zero-burden" assumption was used. Hence, none of the upstream burdens into the waste-management are considered. By focusing only on CDW management the analysis does not capture the potential problems of CDW generation. While the same transport distances are assumed within the scenarios analysed, they may vary significantly across Member States. Future detailed country-based studies are expected to overcome this limitation by using specific regional parameters for transport.

Our analysis and findings aim to support policymaking in relation to CDW management. Particularly, this study could serve as a basis for EU-level policy discussions on how to effectively support and guide the sector towards greater circularity in the future with a view of achieving the European Green Deal and the EU's circular economy objectives.

## CRediT authorship contribution statement

**Dario Caro:** Conceptualization; Methodology; Data curation; Formal analysis; Writing.
**Concetta Lodato:** Data curation; Formal analysis; Supervision.
**Anders Damsgaard:** Data curation; Formal analysis; Supervision.
**Jorge Cristobal:** Data curation; Formal analysis; Supervision; Writing.
**Gillian Foster:** Formal analysis; Validation; Supervision.
**Florian Flachenecker:** Validation; Supervision; Writing.
**Davide Tonini:** Conceptualization; Methodology; Validation; Supervision; Writing.

## Acknowledgments Disclaimer

The results and conclusions are those of the authors and do not necessarily represent the official position of the European Commission, Member States of the European Union, or any organisation the authors are associated with.

## Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

## Data availability

Data will be made available on request.

## Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2023.168295.

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