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Techno-economic and environmental assessment of construction and demolition waste management in the European Union

Status quo and prospective potential

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Contents

Abstract

Construction and demolition waste (CDW) accounts for almost 40% of all waste generated in the EU. The European Commission is taking important binding and non-binding legislative actions to ensure CDW is managed in an environmentally sound manner and contributes to the circular economy. This report reviews, analyses and reconciles data on CDW generation, composition and management at EU level. It also performs an environmental and techno-economic assessment of the most important management technologies through Life Cycle Assessment and Costing for individual material fractions. Results show that, subject to the uptake of best available technologies, recycling and preparing for reuse are preferred over incineration and landfilling for most of the individual material fractions of CDW because of the associated environmental benefits. However, this shift comes with increased costs (while indicating positive societal gains when internalising externalities) for most material fractions, except for soils and dredging spoils, for which uncertainties are significant, and for metals which are already today profitably reused and recycled. The study further estimates the potential for recycling and preparing for reuse for each individual material fraction of CDW, indicating that, excluding excavated soils and dredging spoils due to their significant uncertainty, 83% of CDW can potentially be sent for preparing for reuse and recycling (of which potentially 16% for preparing for reuse). Taking as the baseline the *status quo* of CDW management in the EU in 2020 for each material fraction, and excluding excavated soils and dredging spoils, this would lead to an additional 33 Mt CO_2 equivalent (CO_2 eq.) savings annually (more than for example the combined annual CO_2 eq. emissions from Estonia, Latvia and Luxembourg) at a net cost of EUR 6.3 billion when assuming recycling only (up to 34 Mt CO_2 eq. savings at a net saving of approximately EUR 2.9 billion when including excavated soils and dredging spoils). Under stylised assumptions and when considering the maximum preparing for reuse and recycling scenario, also excluding excavated soils and dredging spoils, a total reduction of about 48 Mt CO_2 eq. with a net saving of approximately EUR 7.3 billion could potentially be achieved (up to 51.5 Mt CO_2 eq. savings at a net saving of approximately EUR 19.5 billion when including excavated soils and dredging spoils). Thus, preparing for reuse should be promoted along with recycling to maximise potential environmental and economic benefits.

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Executive summary

The objective of this study is to provide a techno-economic and environmental assessment of CDW management in the EU, focusing on individual material fractions and including soil that is usually excluded from the scope of most analyses. The report: i) provides a detailed review of the data available on CDW generation, composition and treatment at EU level; ii) provides an overview of CDW management technologies and processes for the different material fractions to identify key technologies and compile inventory data; iii) provides the environmental impacts and costs of managing individual material fractions of CDW via different technologies and processes; and iv) explores the potential for improved CDW management at the EU level considering possible higher preparing for reuse and recycling rates. Finally, the economic and non-economic barriers are discussed as well as the limitations of the study and the feedback (survey results) from stakeholders.

Policy context

The Waste Framework Directive (WFD; Directive 2008/98/EC; European Commission, 2008), later amended in 2018 (European Commission, 2018a), regulates the management of CDW in the EU. In Article 11(2)(b) of Directive 2008/98/EC it is highlighted that by 2020 the preparing for reuse, recycling and other material recovery, including backfilling operations using waste to substitute other materials, of non-hazardous CDW excluding naturally occurring material (i.e. soil and stones) shall be increased to a minimum of 70% by weight. For CDW, the current average recovery rate in the EU is about 89%, which is higher than the 70% by weight goal. However, it should be noted that current recovery uses mainly low-quality recycled aggregates for backfilling material or road construction sub-bases in the best-case scenario. Metals are already often reused or recycled for the same function. On the other hand, despite the potentially high market value of some CDW fractions such as bricks, ceramics, wood and polyvinylchloride (PVC), they are typically not reused or recycled for the same application or function for which they were originally produced. Instead, they are mostly recovered to become aggregates or incinerated or landfilled. For this reason, the European Commission has accompanied the WFD with guidance documents, for example the 'EU Construction and Demolition Waste Management Protocol' (European Commission, 2016; non-binding guidelines on how to properly treat CDW), 'Guidelines for audits before demolition of building' (European Commission, 2018b) and 'Circular Economy – Principles for Building Design' (European Commission, 2020a). In general, the European Commission aims to promote circular economy approaches in the construction and buildings value chain, in line with the 2020 Circular Economy Action Plan (European Commission, 2020b). Sustainable and circular use of excavated soil from construction and demolition waste, which is in line with the EU soil strategy for 2030 (European Commission, 2021a), is also an objective.

Main findings

Using Life Cycle Assessment and Costing we find that material-specific preparing for reuse and advanced recycling processes create significantly higher greenhouse gas (GHG) savings and better environmental performances than incineration and landfill, and are also better than recycling processes only producing recycled aggregates for road construction or backfilling. This is true for almost all material fractions, except in a few cases that are duly explained in this document.

We find that at the EU level the preparing for reuse and recycling rate potential could be in the range of 27-100% across the individual material fractions of CDW investigated, averaging 83% for CDW as a whole (excluding excavated soils and dredging spoils) (these values should be interpreted as the proportion of waste generated that can be 'sent for preparing for reuse and recycling', i.e. without considering the losses within the recycling or reuse process; the value drops to 79% when considering losses). Note that this figure is calculated excluding the mixed inert waste fraction (representing ca. 14% of the total CDW) and soil and dredging spoils, as they are excluded from the 70% recovery rate target of the WFD. When defining the recovery of the mixed fraction as recycling in the equation, the total potential preparing for reuse and recycling rate of CDW would rise to as much as 97%[1]. As for the potential for preparing for reuse alone, we estimated that this could vary between 0% and 50% depending on the material fraction (excluding

[1] Our estimate of 83% is thus not directly comparable with the current EU recovery rate of CDW of 89%, as estimated by Eurostat, because i) Eurostat's 89% is a recovery rate (not a recycling rate) and ii) Eurostat's 89% includes mixed inert waste (which we did not include in the calculation leading to our 83%). If a comparison should be made, our value of 97% should be used instead, which is derived assuming that all of the 'mixed inert waste' fraction can be recycled or recovered to some form of recycled aggregates.

excavated soils and dredging spoils), averaging 16% of the CDW in total (to be interpreted as 'sent for preparing for reuse'; the value drops to 14% when considering losses). These values should be considered as preliminary estimates, based on existing literature and specific case studies, especially in the case of preparing for reuse, and excluding excavated soil and dredging spoils.

Based on these figures, two scenarios are developed and analysed. First, a scenario for a maximum recycling rate, and, second, one for a maximum preparing for reuse and recycling rate. The first scenario is done by reducing landfilling and incineration to the minimum and assuming implementation of the best performing recycling processes following a best available technologies approach, although the reuse rate is kept at the same levels as today. The second scenario follows the exact same assumptions as the first one but prioritises preparing for reuse whenever applicable and to the maximum extent technically possible. The results of the assessment show that, relative to the baseline (*status quo* management of CDW in the EU; using 2020 waste generation figures) and excluding soil and dredging spoils, a total annual reduction of ca. 33 Mt CO_2 eq. at a net cost of approximately EUR 6.3 billion would be achieved with the maximum recycling potential scenario (up to 34 Mt CO_2 eq. savings at a net saving of approximately EUR 2.9 billion when including excavated soils and dredging spoils). In the same line, with the maximum preparing for reuse and recycling scenario a total reduction of ca. 48 Mt CO_2 eq. with a net cost saving of approximately EUR 7.3 billion would be achieved (up to 51.5 Mt CO_2 eq. savings at a net saving of approximately EUR 19.5 billion when including excavated soils and dredging spoils). The cost savings due to a maximum preparing for reuse scenario are due to the theoretical savings gained by reusing instead of processing the waste via incineration, landfilling and recycling (as is the case in the baseline year 2020) and should be considered as a theoretical tendency rather than an accurate cost estimate. Remarkably, using a Marginal Cost Abatement Curve, the study shows that all material fractions contained in CDW truly have non-negligible potential contributions to GHG reductions and environmental savings at below the current CO_2 price, except for wood, gypsum and concrete waste.

Key conclusions

Current recycling and/or recovery of CDW is mainly based on the production and/or use of recycled aggregates mostly for road bases or backfilling. However, more advanced processes and technologies exist that can increase the value of the products recovered and the environmental performance of the recycling itself. These span from preparing for reuse of selected material fractions (e.g. bricks, aluminium, steel, wood, concrete) to advanced processes that recover cement and high-value recycled aggregates for structural uses from concrete waste. While such processes and technologies are available or may be available in the short term, they are generally more costly than landfilling and incineration, with the exception of metals for which reuse or recycling are the most profitable route. The costs are higher, even when an average landfill tax on inert waste disposal is in place. The reasons are the increased costs for processing and selective demolition. However, when accounting for external costs (i.e. monetised environmental emissions), recycling pathways bring overall societal savings. From a purely financial perspective, significantly higher landfill taxes (above the levels in the current study of EUR 19 t^{-1}), as applied in selected Member States (up to or more than EUR 100 t^{-1}, e.g. in the Netherlands), have proven to discourage disposal and favour recycling or recovery operations. Still, this does not ensure that recovery is steered towards high-quality material recycling for use in high-value markets and applications. To this aim, other complementary measures and/or instruments may be necessary. The findings of this study indicate that significant environmental benefits can be obtained from improved reuse and recycling of individual material fractions of CDW. We show that not only concrete, but basically all waste material fractions have a relevant contribution to GHG emission reductions and environmental savings when recovered rather than disposed of or incinerated.

Related and future JRC work

The information contained in this report is supported and complemented by the analyses presented in the following additional studies: Damgaard et al. (2022), Caro et al. (2024), Cristóbal et al. (forthcoming), and Pristerà et al. (forthcoming). Also, the JRC plans to perform a subsequent study to identify policy measures to improve CDW management.

Quick guide

This report is organised in 10 sections as follows. Section 1 presents the introduction and the policy background of the report and the objectives of the study. Section 2 focuses on the CDW characterisation including the quantification and the composition of the different fractions at EU level. It includes a theoretical material fraction generation potential estimated via Material Flow Analysis (MFA), which is estimated without the mixed inert fraction. Section 3 summarises the main technologies used to manage the individual material fractions of CDW. Section 4 details the management of each individual CDW material fraction, including a final summary of the current management pathways in the EU. Section 5 introduces the methodology and results of the Life Cycle Assessment and Costing performed for the different waste management options of the different CDW fractions. Section 6 assesses two scenarios to unravel the potential benefits of increasing the recycling and preparing for reuse of the different CDW fractions. Section 7 analyses the economic and non-economic market barriers for CDW preparing for reuse and recycling. Sections 8 and 9 present the limitations of the study and the stakeholder consultation results, respectively. Finally, Section 10 derives the conclusions of the study.

1. Introduction and policy background

All waste generation has social, economic and environmental impacts associated with it, as well as a loss of valuable materials and resources within the economy. Construction and demolition waste (CDW) accounts for almost 40% of all waste generated in the EU. This makes it an important target for the EU to ensure its management in an environmentally sound way and to ensure it contributes to the circular economy. Thus, in the last two decades, the European Commission has taken clear steps towards the efficient use of resources, including waste prevention, mainly through different communications, legislation (including Directives to be transposed at national level), and guidelines.

Management of CDW in the EU is regulated by the EU Waste Framework Directive (WFD; Directive 2008/98/EC; European Commission, 2008), later amended in 2018 (European Commission, 2018a). The target set in Article 11(2)(b) of Directive 2008/98/EC reads: "by 2020, the preparing for reuse, recycling and other material recovery, including backfilling operations using waste to substitute other materials, of non-hazardous construction and demolition waste excluding naturally occurring material defined in category 17 05 04 in the list of waste shall be increased to a minimum of 70% by weight." The main changes in the management/reporting of CDW introduced by the amended WFD as of 2018 consists of: i) a revised definition of backfilling, to further clarify the distinction between backfilling and other recovery operations, notably recycling, and ii) an increased frequency of reporting to the Commission via Eurostat (2-year instead of 3-year).

As for the revised definition of backfilling, the amended WFD reads "backfilling means any recovery operation where suitable non-hazardous waste is used for purposes of reclamation in excavated areas or for engineering purposes in landscaping. Waste used for backfilling must substitute non-waste materials, be suitable for the aforementioned purposes, and be limited to the amount strictly necessary to achieve those purposes". This is stricter than the earlier definition in Commission Decision 2011/753/EU (European Commission, 2011), which reads "backfilling means a recovery operation where suitable waste is used for reclamation purposes in excavated areas or for engineering purposes in landscaping and where the waste is a substitute for non-waste materials".

The rules for the calculation of the recovery rate[2] and for the frequency of the related reporting to the Commission via Eurostat are detailed in Commission Decision 2011/753/EU, later amended in Commission Implementing Decision 2019/1004/EU (European Commission, 2019). Eurostat's guidance document for reporting of CDW recovery according to the abovementioned rules is available to Member States, and is regularly updated (European Commission, 2022a). CDW reporting has a frequency of 2 years, and must follow the format detailed in 2019/1004/EU (European Commission, 2019) and must include the data collected within 1 calendar year. As for the measurement point, for reporting the amount of CDW material recovered (the numerator of the recovery rate), Article 2(2) of Commission Decision 2011/753/EU sets out that "the weight of the waste prepared for reuse, recycled or materially recovered shall be determined by calculating the input waste used in the preparation for the final recycling or other final material recovery processes. A preparatory operation prior to the submission of the waste to a recovery or disposal operation is not a final recycling or other final material recovery operation. Where waste is collected separately or the output of a sorting plant is sent to recycling or other material recovery processes without significant losses, that waste may be considered the weight of the waste, which is prepared for reuse has undergone other material recovery." In other words, the amount of CDW reported as recovered reflects the amount of waste that enters the final material recovery process. As for the generated CDW (the denominator of the recovery rate), Member States have two options to calculate and report it, i.e. a general method provided by Eurostat in the guidance document (European Commission, 2022a) or their own national data, if deemed to be more accurate.

The current recovery rate of the EU for the year 2020 varies from 63% (Finland) to 99% (Luxembourg) with an average of 89% for the EU[3]. As illustrated later in this document, such recovery often reflects recovery pathways that mainly produce low-quality recycled aggregates (RA) from the mineral fraction of CDW for use as backfilling material or road construction sub-bases in the best-case scenario. For example, a relatively low share of such RA is currently used for structural concrete applications (8.2%) (Pacheco et al., 2023), while cement (a carbon-intensive material) is not recovered at all. Other material fractions of potentially high market value such

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[2] The recovery rate for CDW and the calculation methods are defined in Annex III to 2011/753/EC and is equal to 'CDW material recovered divided by CDW material generated'. The recovery includes preparing for reuse, recycling and other material recovery operations including backfilling operations.

[3] These figures may be found on the Eurostat website under the Circular Economy Indicators (CEI WM_040) and are also reported in Section 4 of this document.

as bricks, ceramics, wood and PVC are typically not reused or recycled for the same application/function for which they originally were produced, but rather recovered as aggregates in the best scenario (e.g. bricks and ceramics) or incinerated or landfilled (PVC and wood). These recovery pathways, notably recycling into aggregates, while certainly diverting the waste from landfills, result in products with low market value and often low quality (process denominated as downcycling), overall incurring low environmental benefits and circularity.

To foster better management practices and the circular economy in the construction and demolition sector, the European Commission has accompanied the revised WFD with guidance documents such as the 'EU Construction and Demolition Waste Management Protocol' (European Commission, 2016; non-binding guidelines on how to properly handle this waste stream), 'Guidelines for audits before demolition of building' (European Commission, 2018b) and 'Circular Economy – Principles for Building Design' (European Commission, 2020a). More recently, in the context of the construction and buildings key value chain under the Circular Economy Action Plan (European Commission, 2020b), the Commission has pointed out the following:

- It would and subsequently did adopt a proposal for a revised Construction Product Regulation on 30 March 2022, taking this opportunity to improve the sustainability performance of construction products, introducing recycled content requirements for certain construction products.
- Promote circular economy approaches in the construction industry ecosystem (European Commission, 2022b) and the development of digital logbooks for buildings.
- Use Level(s), which is the European framework for sustainable buildings, to integrate Life Cycle Assessment in public procurement and the EU sustainable finance framework (the latter was achieved by the adoption of Annex 2 to the Commission Delegated Regulation supplementing Regulation (EU) 2020/852 on the EU Environmental Taxonomy (European Parliament and the Council, 2020)).
- Consider a revision of EU waste legislation, focusing on preparing for reuse and recycling objectives for construction and demolition waste and its material-specific fractions (Article 11(6) of the WFD).
- Promote soil-related initiatives, aiming to increase safe, sustainable, and circular use of excavated soils. This last point is in line with the EU soil strategy for 2030 (European Commission, 2021a) which promotes the waste hierarchy introduced in the WFD and states that excavated soils should be reused in the same or another location (most excavated soils are clean, fertile and healthy), and if not possible they should be prioritised for recycling or some other form of recovery rather than landfilling.

The European Commission has also published a Transition Pathway for Construction (European Commission, 2023c), which aims to offer a bottom-up and co-created understanding of the scale, cost, and conditions for resilience, competitiveness, and the green and digital transition of the construction ecosystem, including actions related to CDW.

1.1 Objectives of the study

The overall aim of this report is to compile data and provide a techno-economic and environmental assessment of the CDW management options, focusing on individual material fractions (including the excavated soils that are usually excluded from the scope of most analyses). This report intends to support further policymaking on proposals for CDW objectives for preparing for reuse and recycling and its material-specific fractions in accordance with the WFD. The specific objectives of this study are as follows:

- Review, analyse and reconcile data on CDW generation, composition and treatment at EU level to be used in subsequent analysis.
- Review the literature on CDW management technologies for the different fractions in order to identify key technologies and compile technical data for further modelling.
- Establish life cycle inventories for selected CDW management technologies from the reviewed literature.
- Conduct a Life Cycle Assessment (LCA) and Environmental and Societal Life Cycle Costing (ELCC and SLCC) for the selected CDW management options for the studied individual CDW fractions.
- Explore the potential for improved CDW management at EU level through two scenarios in which preparing for reuse and recycling rates are significantly improved.

Note that this report is supported by the following complementary studies and associated publications:

- Damgaard et al. (2022): providing a detailed review of the data reported across Member States for CDW generation and management as well as a material flow analyses of CDW from buildings in the EU for 2020 and 2050.
- Caro et al. (2024): providing a detailed environmental (via LCA) and socio-economic assessment (via ELCC and SLCC) of CDW management.
- Cristóbal et al. (forthcoming): providing a detailed environmental (via LCA) and socio-economic assessment (via ELCC and SLCC) of excavated soil and dredging spoil management.
- Pristerà et al. (forthcoming): providing a detailed analysis of selective demolition and design for deconstruction measures as means of achieving a reduction of CDW and a prioritisation of preparing for reuse and high-quality recycling options.

2. CDW characterisation: generation and composition

Based on a literature review and on the previous study by Damgaard et al. (2022), this section aims at characterising and quantifying the generation and composition of CDW for the EU. Thus, this section tries to reconcile data for CDW, setting the basis for further analysis performed in subsequent sections.

2.1 Relevant material fractions for CDW in the European Waste Code (EWC)

CDW is composed of different waste fractions and materials that are registered under specific codes according to the two main coding systems applied within the EU: the List of Waste (LoW) and the European Waste Code Statistics (EWC-Stat). The former is the waste classification in the EU for administrative purposes and is structured in 20 chapters, mainly according to the source of waste (i.e. the economic sector or process of origin). The latter is a substance-oriented aggregation of the waste types defined in the LoW. It is possible to unambiguously convert the waste types classified according to the LoW into the EWC-Stat waste categories that is the main method reported in the Commission Decision 2011/753/EU (European Commission, 2011) setting out the rules and calculation methods for the compliance monitoring (European Commission, 2022a). Table 1 shows the relation of LoW and EWC-Stat codes for the CDW material fractions considered in this report.

Table 1. Description of the CDW fractions considered in this study with a correlation with the List of Waste (LoW) and the European Waste Code Statistics (EWC-stat) codes.

CDW fractions	Considered in scope	LoW code	EWC-stat
Mineral waste	Mineral waste	17 01	W12.1
Concrete	Concrete	17 01 01	W12.11
Bricks	Bricks	17 01 02	W12.11
Tiles and ceramic	Tiles and ceramic	17 01 03	W12.11
Other materials from road demolition	EXCLUDED		
Mixed/other mineral/inert waste	Mixed/other mineral/inert waste	17 01 07	W12.11
Asphalt waste	Bituminous mixtures containing coal tar	17 03 02(1)	W12.12(1)
Plastic	Plastic	17 02 03 / 19 12 04(2)	W07.42
Metal	Metal	17 04	W06
Mixed metals, incl. cables	Mixed metals	17 04 07, 17 04 11	W06.32, W06.26
Ferrous	Ferrous	17 04 05 / 19 12 02(2)	W06.11
Non-ferrous	Non-ferrous	17 04 01, 17 04 02, 17 04 03, 17 04 04, 17 04 06 / 19 12 03(2)	W06.24, W06.23, W06.25, W06.26
Glass	Glass	17 02 02 / 19 12 05(2)	W07.12
Wood	Wood	17 02 01 / 19 12 07(2)	W07.53
Gypsum	Gypsum	17 08 02	W12.11

Insulation	Insulation	17 06 04	W12.13
Paper and cardboard	Paper and cardboard	NA[3] 19 12 01[2]	W07.23[2]
Mixed waste, generic	Mixed waste, generic	17 09 04 / 19 12 09[2]	W12.13 / W12.81[2]
Mix of non-hazardous, non-inert wastes			
Mix of inert and non-hazardous, non-inert wastes			
Others			
Soil	Soils		
Unpolluted	Soils and stones	17 05 04[1]	W12.61[1]
Polluted	Soils and stones containing dangerous substances	17 05 03*[1]	W12.61[1]
Dredging spoil	Dredging spoils		
Unpolluted	Dredging spoils	17 05 06[1]	W12.71[1]
Polluted	Dredging spoils containing dangerous substances	17 05 05*[1]	W12.71[1]
Track ballast	Track ballast		
Unpolluted	Track ballast	17 05 08[1]	W12.11[1]
Polluted	Track ballast containing dangerous substances	17 05 07*[1]	W12.11[1]
Hazardous waste (excl. hazardous soil, dredging spoil, track ballast)	Hazardous waste (excluding hazardous soil and dredging spoil)	17 01 06*, 17 02 04*, 17 03 01*, 17 03 03*, 17 04 09*, 17 04 10*, 17 06 01*, 17 06 03*, 17 06 05*, 17 08 01*, 17 09 01*, 17 09 02*, 17 09 03*	W12.11, W12.12, W10.22, W12.13, W12.21, W07.73

(1) In green, the codes that were excluded in Damgaard et al. (2022) but included in this study.
(2) In red, the LoW entries that according to the Commission Decision 2011/753/EC (Annex III) shall also be included in the calculation of CDW recovery targets but that in this study have not been considered since it was not possible to know whether they were generated from the treatment of waste coming from construction and demolition activities.
(3) There is no LoW code for Paper and Cardboard within the Construction and demolition waste category (Chapter 17) of the European Waste Catalogue (EWC, Commission Decision 2014/955/EC).

Note: Any waste marked with an asterisk (*) in the list of waste shall be considered as hazardous waste. It is important to highlight that reporting obligations are only for non-hazardous wastes excluding soils and dredging spoils (i.e. naturally occurring materials).

Source: Adapted from Damgaard et al. (2022).

2.2 CDW characterisation based on a literature review of reported data

According to the literature review by Damgaard et al. (2022), which includes more than 90 reports and articles (comprising Eurostat sources, techno-scientific literature, and country-specific data obtained via stakeholders and environment agencies), the generation of CDW in the EU in 2018 amounted to ca. 848 Mt when including soil, track ballast, dredging spoils, and asphalt (Table 2). When excluding soil, track ballast, dredging spoils, and asphalt the quantity of CDW generated amounted to ca. 276 Mt. The authors provide a detailed breakdown of the CDW material fraction composition at the EU level (Table 3) as well as at Member State level (for the latter, the reader is referred to the original document by Damgaard et al., 2022).

The main messages from the literature review are the following:

- A high variation in the amount of reported CDW generation exists across Member States, from 0.02 t/capita in Bulgaria to 3.72 t/capita in Malta (excluding soil, track ballast, dredging spoils and asphalt).
- The largest material fraction on average in CDW, when excluding excavated soil waste, is the mineral fraction (77%). It consists mainly of concrete and bricks, followed by metal (4.3%), wood (2.3%), and gypsum (1.4%). However, great variation exists across Member States, notably for wood (reaching 18-21% in Sweden and Finland).
- A high variation in the amount of reported soil waste generation exists across Member States, from almost zero in Malta to significant amounts in Finland and Luxembourg, where soil waste is the largest fraction of CDW. This is likely related to the way soil from excavation is classified at Member State level (by-product versus waste) and thus reported.
- The level of detail reported for the material fraction composition varies greatly across Member States. For some countries (e.g. Denmark, Germany, Luxembourg, Netherlands), data are available for almost all material fractions, including a breakdown into the different components of the mineral fraction (concrete, bricks, tiles and ceramic), which might indicate a good level of source separation. Other countries, e.g. Belgium, Bulgaria, Cyprus and Finland, only report mineral waste data in aggregated form, possibly reflecting poor practices within source segregation and selective demolition. In addition, these countries also report data for plastic, metal, glass, wood and hazardous waste. This further supports the argument that the reported CDW data do not necessarily represent the actual material composition, but rather reflect CDW management practices.
- Many Member States report a high share of 'mixed CDW' fraction. This fraction ranges from 5% (Germany) to 100% (Poland), possibly suggesting poor practices within source segregation and selective demolition. Note that data for the "mixed CDW" fraction are not available in Eurostat datasets as Eurostat does not include it.
- The data for insulation are likely to refer only to mineral insulation as polymer-based insulating material would be likely sorted and classified as plastic waste. However, this remains unclear in the reporting.
- CDW data for Italy were adjusted, as the categorisation of the main Italian CDW fraction as LoW code 17 09 04 (indicating a mix of waste from construction and demolition) was understood as a misclassification; as this main fraction is, to a large extent, used to produce RA. A more accurate classification that better reflects the actual material composition of Italian CDW was suggested to be LoW code 17 01 07. The revised CDW composition for Italy is used for further assessment.

Table 2. Current CDW generation in 29 European countries, as well as total CDW generation for all 27 Member States of the EU and for EU + Norway. CDW data are presented as both including (incl.) and excluding (excl.) soil, track ballast, dredging spoils and asphalt, with a detailed focus on soil waste generation.

	Total CDW generation incl. soil, track ballast, dredging spoils and asphalt [t]	Total CDW generation excl. soil, track ballast, dredging spoils and asphalt [t]	Total soil waste generation [t]	Per capita CDW generation excl. soil, track ballast, dredging spoils and asphalt [t/capita]	Per capita CDW generation excl. track ballast, dredging spoils and asphalt, incl. soil [t/capita]	Per capita soil waste generation [t/capita]

Austria	48 961 689	11 521 240	37 440 449	1.29	5.50	4.21
Belgium	26 791 280	22 960 461	2 973 938	1.99	2.25	0.26
Bulgaria	384 408	161 090	74 535	0.02	0.03	0.01
Croatia	1 239 094	646 163	582 492	0.16	0.30	0.14
Cyprus	1 048 713	333 468	715 245	0.38	1.18	0.81
Czechia	14 422 791	4 262 791	9 442 000	0.40	1.28	0.88
Denmark	14 162 000	3 818 000	9 139 000	0.66	2.23	1.57
Estonia	2 917 272	1 260 097	1 599 472	0.95	2.15	1.20
Finland	23 676 196	1 871 918	21 789 333	0.34	4.28	3.94
France	252 951 500	53 151 500	175 110 000	0.79	3.39	2.60
Germany	201 345 300	72 215 800	108 582 300	0.87	2.17	1.31
Greece	3 244 848	1 440 182	1 730 862	0.13	0.30	0.16
Hungary	7 399 179	3 520 557	3 808 105	0.36	0.75	0.39
Ireland	2 857 434	733 745	2 123 689	0.15	0.58	0.43
Italy	56 681 821	43 045 079	13 600 000	0.72	0.95	0.23
Latvia	390 530	385 959	4 571	0.20	0.20	0
Lithuania	934 554	890 240	44 297	0.32	0.33	0.02
Luxembourg	5 121 118	432 067	4 481 481	0.69	7.85	7.16
Malta	1 975 105	1 915 040	65	3.72	3.72	0
Netherlands	101 562 751	24 317 000	7 766 598	1.40	1.84	0.45
Poland	15 322 360	4 523 831	10 071 815	0.12	0.38	0.27
Portugal	2 035 326	1 696 938	338 234	0.16	0.20	0.03
Romania	1 584 229	965 633	618 596	0.05	0.08	0.03
Slovakia	3 322 470	859 643	1 542 577	0.16	0.44	0.28
Slovenia	4 934 998	1 085 440	3 022 189	0.52	1.96	1.44
Spain	39 539 766	14 807 048	24 729 360	0.31	0.84	0.52
Sweden	12 959 008	3 627 928	8 885 143	0.35	1.21	0.86

Norway	4 650 868	1 750 394	2 702 226	0.33	0.83	0.50
UK	145 116 633	71 788 216	62 009 410	1.07	2.00	0.93
Total EU	847 765 740	276 448 858	450 216 346	0.78	1.86	1.08
Total EU + Norway	852 416 608	278 199 252	452 918 572	0.76	1.82	1.06

Source: Damgaard et al. (2022).

Table 3. Current average CDW composition (expressed as % of the total per capita CDW amounts) for EU and EU+Norway. CDW data are presented both excluding and including soil, track ballast, dredging spoils, and asphalt.

CDW	Total CDW generation excl. soil, track ballast, dredging spoils and asphalt		Total CDW generation incl. soil, track ballast, dredging spoils and asphalt	
	EU	EU+Norway	EU	EU+Norway
Mineral waste	77.0%	76.6%	27.5%	27.4%
Concrete	*24.0%*	*23.9%*	*8.6%*	*8.5%*
Bricks	*5.0%*	*4.9%*	*1.8%*	*1.8%*
Tiles and ceramics	*1.2%*	*1.2%*	*0.4%*	*0.4%*
Mixed/other mineral/inert waste	*46.9%*	*46.6%*	*16.8%*	*16.7%*
Plastic	0.2%	0.2%	0.1%	0.1%
Metal	4.3%	4.3%	1.5%	1.6%
Mixed metals	*0.5%*	*0.5%*	*0.2%*	*0.2%*
Ferrous	*3.4%*	*3.4%*	*1.2%*	*1.2%*
Non-ferrous	*0.4%*	*0.4%*	*0.1%*	*0.1%*
Glass	0.2%	0.2%	0.1%	0.1%
Wood	2.3%	2.5%	0.8%	0.9%
Gypsum	1.4%	1.5%	0.5%	0.5%
Insulation	0.3%	0.3%	0.1%	0.1%
Paper and cardboard	0.2%	0.3%	0.1%	0.1%
Mixed waste, generic	12.3%	12.0%	4.4%	4.3%
Hazardous waste (total, excluding hazardous soil and dredging spoil)	1.8%	2.0%	0.6%	0.7%
Soil (hazardous and non-hazardous)	-	-	54%	54.2%

Dredging spoil (hazardous and non-hazardous)	-	-	9.2%	9.1%
Track ballast and asphalt	-	-	1.0%	1.0%
TOTAL	100%	100%	100%	100%

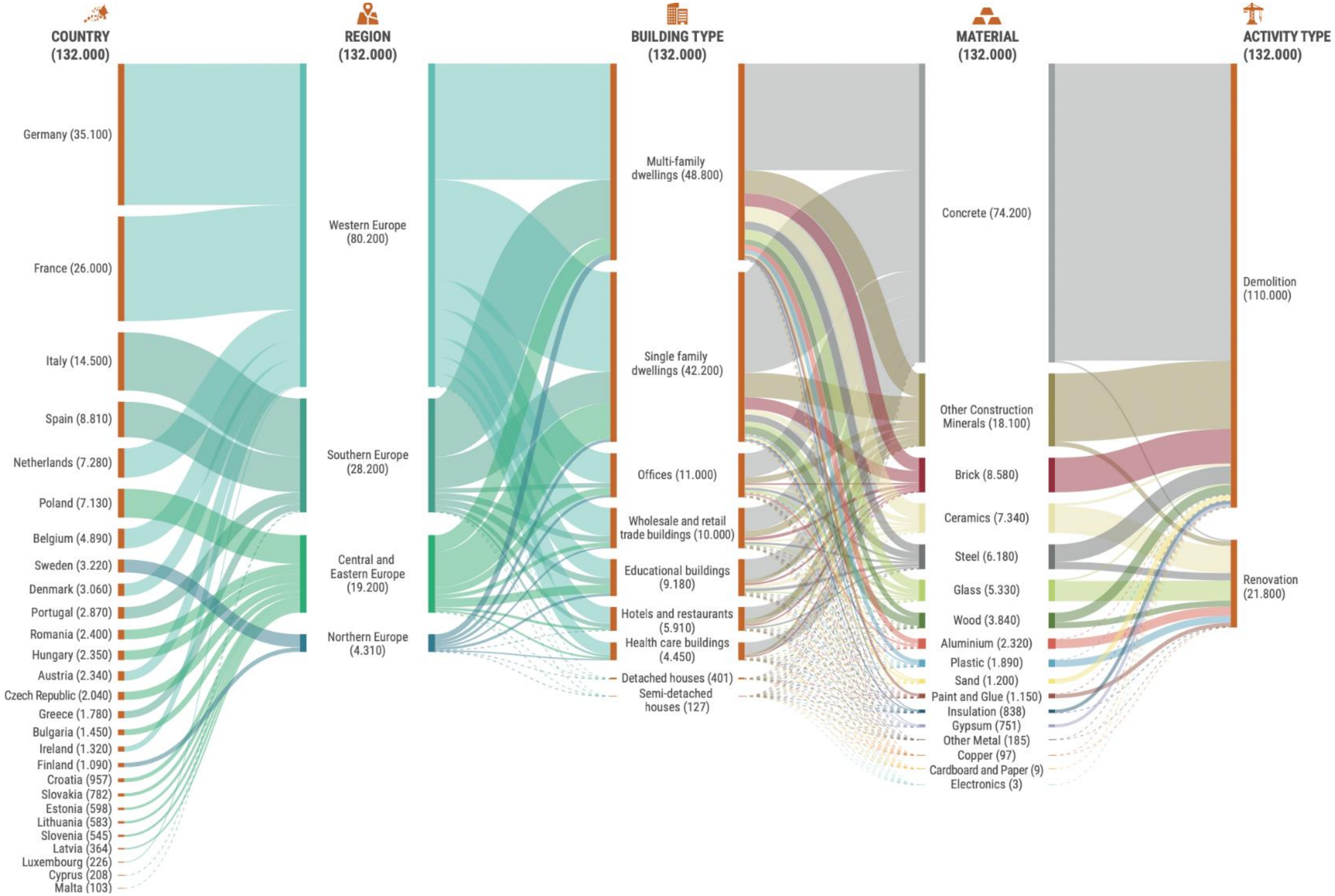
Source: Adapted from Damgaard et al. (2022).

2.3 Theoretical material fraction generation potential via Material Flow Analysis (MFA) modelling

The data reported to EU or national authorities (e.g. Table 2), from which the EU average is reported in Table 3, is derived and is highly affected by the demolition, separation and management practices applied. This can be easily seen as almost 50% of the material is reported as 'mixed'. To estimate the theoretical potential of each individual material fraction of CDW (i.e. how much is theoretically present in CDW regardless of the demolition and separation/management techniques, and that could potentially be recovered), Damgaard et al. (2022) performed MFA modelling. The MFA modelling was established by dividing the EU in four macro-regions (northern, southern, eastern and western). The material stock and flows in the EU in 2020 were calculated for four Member States as representatives of the four regions. Material stock refers to the mass of materials contained in the building stock. The flows refer to both the inflow and outflow of materials from and to the building stock. Inflow describes the materials used for construction and renovation, while outflow refers to waste originating from demolition at end of life and renovation. For this, several input datasets were required. Once the data on the building stock composition in 2020 in the EU was established, using the stock quantities as a baseline, the flows originating from the building stock were subsequently calculated using construction, demolition and renovation rates. Finally, the stock and flow quantities were multiplied with the material intensities (materials in the different building stocks) that represent the four regions, building types, and construction years. A similar approach was applied for 2050, taking however into account projected increases in renovation waves and in related use of specific materials, such as insulation.

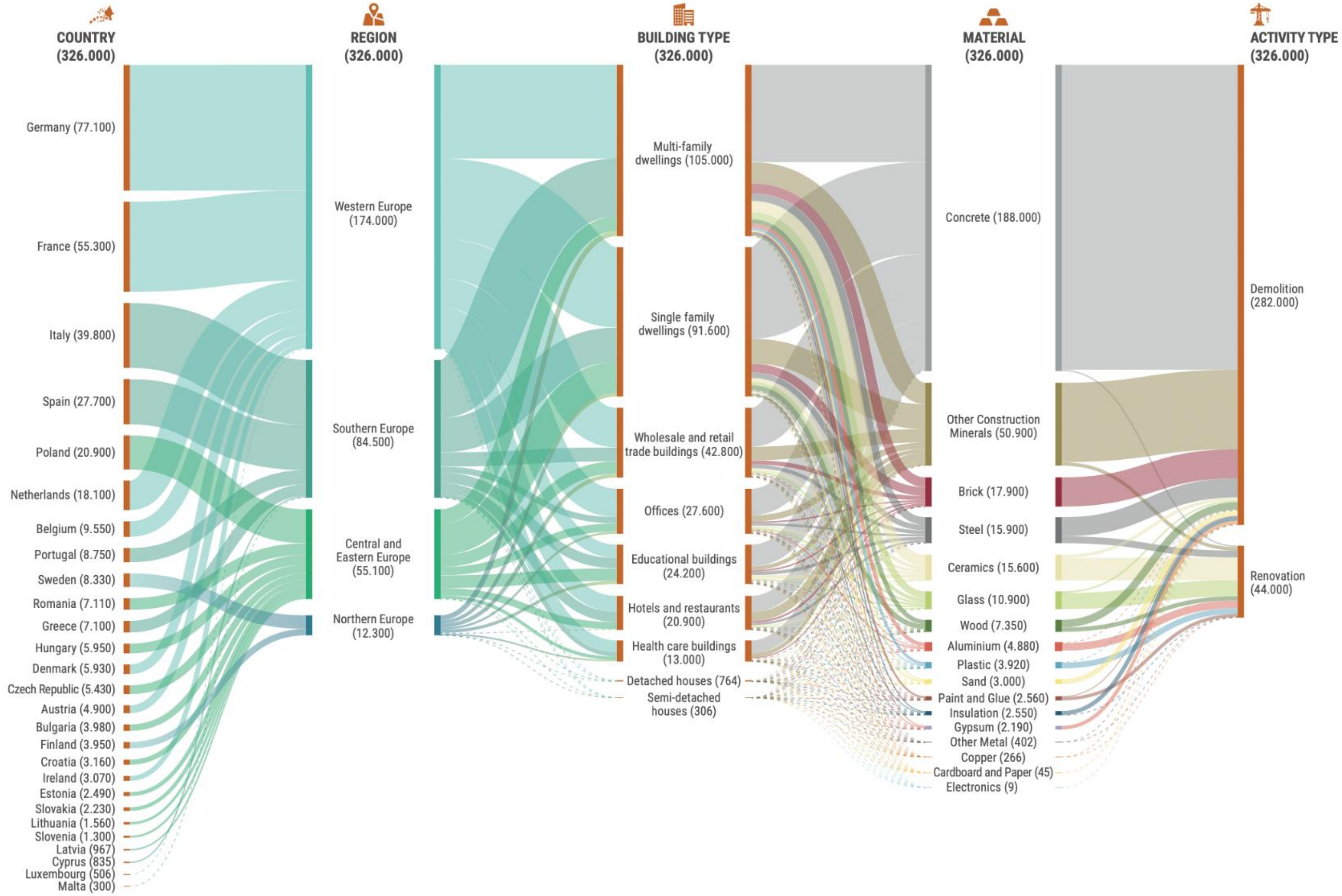
The benefit of using this MFA modelling approach over the reported CDW data is the added detail the modelling can offer regarding the individual material fractions and the origin of materials (building types and building ages). This modelling approach also allows separating and excluding infrastructure waste from building CDW. While modelling assumptions and background data used are thoroughly detailed in Damgaard et al. (2022), the MFAs for year 2020 and 2050 are illustrated in Figure 1 and Figure 2 respectively, and the CDW composition for both points in time in Table 4.

Figure 1. MFA showing the origin of building-related material flows from countries, regions, building types and demolition and renovation activities in 2020 in the EU (infrastructure waste not included) in kt. Note that the total inflow of material to construction and renovation was estimated to be ca. 1 004 000 kt. Dashed lines in the Sankey diagram represent flows under 210 kt.



Source: Damgaard et al. (2022).

Figure 2. MFA showing origin of building-related material flows from countries, regions, building types and demolition and renovation activities in 2050 in the EU (infrastructure waste not included) in kt. Note that the total inflow of material to construction and renovation was estimated to be ca. 1 594 000 kt. Dashed lines in the Sankey diagram represent flows under 600 kt.



Source: Damgaard et al. (2022).

The main messages from the MFA modelling are the following:

- In 2020, demolition is responsible for 83% of material flows (ca. 110 Mt), while renovation is responsible for 17% (ca. 22 Mt; (Figure 1)). Projecting waste generation to 2050, demolition will be responsible for 87% of material flows (ca. 282 Mt), while renovation for 13% (ca. 44 Mt; Figure 2).
- In 2020, the total outflow from demolition and renovation represents only about 13% of the inflow to construction and renovation (1004 Mt; from Damgaard et al. (2022)). In 2050, the total outflow from demolition and renovation represents about 20% of the inflow to construction and renovation (1 594 Mt; from Damgaard et al. (2022))[4]. From a circular economy perspective, this means that even if all material outflows were to be prepared for reuse and recycled, only a share of the primary material needed for construction and renovation could be substituted.
- The material composition of waste from demolition and renovation differs significantly. For instance, renovation is responsible for a larger share of the ceramics, glass and insulation material (e.g. replacement of kitchen, bathroom, toilets and energy-related interventions; see Figure 1- Figure 2), whereas demolition is mainly about concrete.
- The material fractions most abundant in the outflow are concrete, other construction minerals, and bricks (Table 4). These materials are mainly used in the foundation and structure of buildings, and typically have a high (embodied carbon) density. From a circular economy (material efficiency) and climate change mitigation perspective, this is an argument in favour of the transformation (reuse) of buildings as opposed to new construction. Building transformation maintains the structural elements of a building, which allows for the retention of 80-90% of the materials on-site, while only substituting the outfitting, finishes and mechanical and electrical installations, which represent a relatively small fraction of the total material mass.
- It is projected that by 2050 the relative fractions of concrete and insulation will increase, while the relative fractions of wood, plastic, ceramics and glass will decrease slightly. Since concrete is such a dominant fraction (from a weight perspective), the percentage change in the other fractions is relatively small (Table 4).
- Overall, concrete represents by far the largest material fraction in 2020 and 2050, with 56.2% and 57.6% respectively. Other construction minerals increase from 13.7% to 15.6%, while bricks decrease from 6.5% to 5.5%. Steel remains roughly the same. Wood decreases from ca. 3% to 2.3%, while insulation increases from 0.6% to 0.8% (Table 4).
- Plastics are responsible for 1.4% and 1.2% of the total outflow of CDW in 2020 and 2050, respectively (Table 4).

Table 4. Theoretical CDW composition from building demolition and renovation activities estimated via MFA. Note that this composition only refers to CDW from buildings and excludes infrastructure waste and soil waste. As such, it cannot be directly compared to the fractional composition reported for CDW as a whole in Table 3.

Material fraction	Year 2020		Year 2050	
	kt	%	kt	%
Aluminium	2 323	1.8%	4 883	1.5%
Brick	8 583	6.5%	17 937	5.5%
Cardboard and Paper	9	0.01%	45	0.01%
Ceramics	7 340	5.6%	15 632	4.8%
Concrete	74 169	56.2%	187 891	57.6%

[4] Damgaard et al., 2022 (see Figure 22 for 2020 and Figure 27 for 2050 and related additional material).

Copper	97	0.07%	266	0.08%
Electronics	3	0.002%	9	0.003%
Glass	5 328	4.0%	10 895	3.3%
Gypsum	750	0.6%	2 187	0.7%
Insulation(1)	839	0.6%	2 547	0.8%
Other Construction Minerals	1 809	13.7%	50 85	15.6%
Other Metal	184	0.1%	402	0.1%
Paint and Glue	1 148	0.9%	2 562	0.8%
Plastic	1 889	1.4%	3 921	1.2%
Sand	1 195	0.9%	3 004	0.9%
Steel	6 174	4.7%	15 893	4.9%
Wood	3 835	2.9%	7 351	2.3%
Total	131 956	100%	326 275	100%

(1) Insulation materials in the MFA modelling include wall, floor, and roof insulation of various material compositions: inorganic (e.g. glass wool or stone wool), organic (e.g. cellulose insulation) or polymer based (e.g. EPS and PUR).

Source: (Damgaard et al., 2022; elaborated from Table F15 and F20 of the supplementary information provided along with the main report).

2.4 Focus on infrastructure waste (besides buildings)

According to Damgaard et al. (2022), infrastructure waste is waste from infrastructure activities besides buildings, i.e. construction, maintenance, renovation and demolition of roads, bridges, tunnels, and other infrastructures. Thus, infrastructure CDW includes the different asphalt waste fractions (i.e. bituminous mixtures not containing coal tar – LoW code 17 03 02 that mainly consist of aggregates, a binder (such as bitumen[5] binder) and additives (e.g. rejuvenators, anti-stripping agents). According to Arm et al (2014), the asphalt waste fraction includes three major types of products, i.e. asphalt-based paints (that represent a very small minority of the use of asphalt and are not subject to recycling activities), roofing asphalts and paving asphalts that are a mixture of mineral aggregate, bituminous binder (up to 7%) and filler (Arm et al., 2014).

Along with the asphalt stream, national CDW data from Austria (BIO by Deloitte, 2015a), Germany (BIO by Deloitte, 2015b), and Sweden (BIO by Deloitte, 2015c) confirm track ballast as infrastructure waste from the construction/disassembling of railroad tracks. Finally, concrete is a CDW stream belonging to both infrastructure and building waste and therefore it is reported as a whole, making it difficult to identify the quantity originating from infrastructure only.

A mini-literature review is herein performed based on the systematic approach proposed by Denyer & Tranfield (2009) to assess the qualitative-quantitative characterisation and management practices as inclusive as possible, and to produce a general picture of the current situation avoiding bias. Specifically, based on literature information in ScienceDirect[6], we herein refer to 'qualitative characterisation' when there is a correlation between CDW and infrastructure activities, and refer to 'quantitative characterisation' and 'management practice' when information on quantity and end-of-life (EoL) management practices is available, respectively. A synthesis of the method of analysis is provided in Figure 3.

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[5] Bitumen is the product of the non-destructive distillation of crude oil in petroleum refining.
[6] https://www.sciencedirect.com

Figure 3. Materials and methods for the literature review on infrastructure CDW.

Database search (31 May 2023)
ScienceDirect
(Title, abstract or author-specified keywords)
Keywords selection
construction and demolition waste
AND
management OR
recovery OR
infrastructure OR
road OR
building

→

Publications after screening and data cleaning (title, abstract, text)
Deletion of duplicates
Validation of the coherence and usefulness

↓

Results are extracted from publications
Qualitative characterisation
Quantitative characterisation
Management practices

Source: Own elaboration.

As shown in Figure 3, the following keywords were entered to conduct the search: “construction and demolition waste”; “management”; “recovery”; “infrastructure”; “road”; “building”. Keywords were inputted on May 31, 2023. The choice of keywords is based on the aim of the research, and the results of the bibliographic review are shown in Table 5.

Table 5. Numbers of publications according to the material collection process.

		KEYWORD 1+2	
KEYWORD 1	KEYWORD 2	N° of publications	N° of publications after screening and data cleaning
Construction and demolition waste	Management	325	112
	Recovery	73	
	Infrastructure	54	
	Road	91	
	Building	236	

Source: Own elaboration.

Given the base keyword “construction and demolition waste” and each of the five measurement evaluation keywords mentioned above, many publications are detected (column “N° of publications”). From this pool of articles, a subset of 112 studies is identified from the ScienceDirect database after screening and data cleaning processes in the form of deletion of duplicates and validation of the coherence and usefulness for the purpose of the mini review. At the end of the sorting process 13 relevant papers are characterised in which a correlation between infrastructure waste and concrete, asphalt or track ballast is detected (see Table 6).

Table 6. Qualitative characterisation of the detected publications.

Reference	Qualitative characterisation		
	Concrete	Asphalt	Track ballast
Arm et al., 2017		X	X
Lederer et al., 2020	X	X	
Menegaki & Damigos, 2018	X	X	
Mhatre et al., 2021	X	X	X
Youcai & Sheng, 2017	X	X	

El-Haggar, 2007	X	X	
Zhang et al., 2022		X	
De Melo et al., 2011		X	
Fatta et al., 2003		X	
Gálvez-Martos & Istrate, 2020	X	X	
Sáez et al., 2014	X	X	
Cristiano et al., 2021	X	X	X
Roque et al., 2016	X		X

Source: Own elaboration.

Eight of the selected sources apply to the EU context (either to country-specific or general frameworks). Concerning the qualitative characterisation, asphalt is recognised as infrastructure waste either alone (De Melo et al., 2011; Fatta et al., 2003; C. Zhang et al., 2022) or together with concrete (El-Haggar, 2007; Gálvez-Martos & Istrate, 2020; Lederer et al., 2020; Menegaki & Damigos, 2018; Sáez et al., 2014; Youcai & Sheng, 2017) or track ballast (Maria Arm et al., 2017; Cristiano et al., 2021; Mhatre et al., 2021; Roque et al., 2016).

No detailed information is available in the 13 studies analysed to perform a systematic characterisation of infrastructure waste in terms of quantity generated. These studies however confirm concrete, asphalt and track ballast as the most relevant CDW flows related to infrastructure. Using the Eurostat database, it is not possible to determine the amounts of infrastructure waste of all the Member States, as this is reported as part of the mineral waste fraction (i.e. W121 Mineral construction and demolition wastes) and not as individual category (i.e. LoW code 17 03 02 for bituminous mixtures, and 17 05 08 for track ballast).

Additional literature insights (see the first column of Table 7) report infrastructure (asphalt) waste data for 7 of the 23 analysed countries. Due to the lack of data, two secondary raw materials streams deriving from infrastructure CDW, the reclaimed asphalt pavement and asphalt recycled aggregates, are considered as possible proxies and compared with the asphalt waste fraction to obtain a more complete picture. The reclaimed asphalt is generated due to maintenance, reconstruction, resurfacing, or to obtain access to buried utilities and is covered by harmonised European standards, while the asphalt recycled aggregates are reprocessed granular materials previously used in construction of infrastructure (UEPG, 2023). Specific data for reclaimed asphalt from the European Asphalt Pavement Association (EAPA) are found (EAPA, 2023), covering 12 of the 23 countries analysed in the current study (i.e. Austria, Belgium, Croatia, Czechia, Denmark, France, Germany, Hungary, Ireland, Slovakia, Slovenia, Spain). These data are reported in the second column of Table 7 while data from the Aggregates Europe – UEPG are shown in the third column (UEPG, 2023) and apply to 16 of the 23 countries analysed (i.e. Austria, Belgium, Bulgaria, Cyprus, Denmark, Finland, France, Germany, Greece, Malta, Netherlands, Poland, Portugal, Slovakia, Spain, Sweden). The significant differences between the asphalt waste fraction and the two secondary raw material flows (e.g. asphalt recycled aggregates is up to 7 times larger than the asphalt waste fraction in France) is related to the fact that both reclaimed asphalt and asphalt recycled aggregates are often not reported as waste, especially if prepared for reused/recycled on-site, as also highlighted in Arm et al. (2017). However, the lack of a suitable level of detail of data does not make it possible to perform a systematic comparison and thereby draw any overall conclusion. Therefore, room for improvement exists.

The annual reporting on material recovery from CDW according to Commission Decision 2011/753/EU (European Commission, 2011) and Commission Implementing Decision (EU) 2019/1004 (European Commission, 2019) could represent a starting point from which to extrapolate the individual fractions of the Member States, with a particular focus on infrastructure waste asphalt (LoW code 17 03 02) and track ballast (LoW code 17 05 08) (see Table 3). In this respect, though the asphalt and track ballast average compositions do not exceed the 1% of the overall CDW amount, only five of the 27 Member States (i.e. Czechia, Denmark, France, Germany and Luxembourg) are represented within that percentage making it less significant and very uncertain. It would be desirable to standardise the data collection methods at the European level so that all Member States can contribute with the same level of detail (e.g. indicating the individual LoW codes rather than EWC-stat category).

Table 7. Amounts and comparison between asphalt waste fraction in the European countries and asphalt secondary raw materials (reclaimed asphalt and asphalt recycled aggregates).

Country	(1) Asphalt waste fraction (Source: Damgaard et al., 2022; BIO by Deloitte, 2015a; BIO by Deloitte, 2015d) [t year^{-1}]	(2) Reclaimed asphalt (Source: EAPA, 2021) [t year^{-1}]	**Δ%** between (1) and (2)	(3) Asphalt recycled aggregates (Source: UEPG, 2021) [t year^{-1}]	**Δ%** between (1) and (3)
Austria	2 402 000	900 000	167%	4 000 000	-67%
Belgium	-	1 506 000		22 000 000	
Bulgaria	-	-		1 000 000	
Croatia	-	390 000		-	
Cyprus	-	-		400 000	
Czechia	508 000	2 500 000	-392%	-	
Denmark	1 169 000	1 172 000	-0.26%	3 000 000	-157%
Finland	-	-		5 200 000	
France	9 300 000	6 042 000	54%	67 900 000	-630%
Germany	15 416 500	11 600 000	33%	76 000 000	-393%
Greece	-	-		600 000	
Hungary	-	160 000		-	
Ireland	-	500 000		-	
Italy	636 902	-		-	
Luxembourg	203 599	-		-	
Malta	-	-		300 000	
Netherlands	-	-		24 300 000	
Poland	-	-		7 000 000	
Portugal	-	-		200 000	
Slovakia	-	70 946		600 000	
Slovenia	-	79 000		-	
Spain	-	2 495 000		3 500 000	
Sweden	-	-		5 800 000	

Source: Own elaboration.

2.4.1 Management practices for infrastructure waste

Concerning the asphalt waste fraction, mainly asphalt pavement, recycling processes include operations such as milling, crushing, sieving (screening), blending (Ali & Rojali, 2023; EAPA, 2023) and clearing with magnetic separators (El-Haggar, 2007). Recycling asphalt to produce new asphalt[7] in either stationary or mobile plants is well-established, using different techniques to include the reclaimed asphalt in hot mix, warm/half warm mix or cold mix. There are often restrictions on the quantity of reclaimed asphalt to be used in the mix according to the standard EN 13108-8 (EAPA, 2023). Besides, it is possible to recycle the reclaimed asphalt through hot in-place recycling processes performed by specialised machines where the road surface is heated, milled and the asphalt mix is recycled in place (Arm et al., 2014). It is also possible to recycle asphalt as unbound aggregates through a crushing process. Specifically, reclaimed asphalt could be used as road base with other crushed and screened aggregates, new paving material, and asphalt products by mixing it with new asphalt binders. Finally, reclaimed asphalt can be sometimes mixed with other recovered materials for backfilling purposes or used in

[7] According to Arm et al. (2017), it is not considered preparing for reuse since formally it does not fulfil the WFD definition.

unspecified backfilling as it is (Arm et al., 2017). For some countries, such as Germany, available data from stakeholders (Kreislaufwirtschaft Bau, 2023) show that for bituminous mixtures (LoW code 17 03 02), in 2020, 92.9 % were recycled, 3 % were backfilled and 4.1 % were disposed of in landfills. Roofing asphalt can also be recycled into paving asphalt by existing techniques (Barry et al., 2014), acknowledging that even if the quantity of this material is small compared to paving asphalt, the bitumen content is not negligible (Arm et al., 2017).

For the track ballast, preparing for reuse again as track ballast is possible through a cleaning process using rail-mounted machines, where a residual fine-grained fraction that cannot be used as track ballast is removed (Arm et al., 2017). Track ballast when resulting from demolition or excavation of rail tracks is commonly recycled as aggregate in asphalt production or unbound applications. The recycling operation can include crushing in order to produce a certain desired particle size. The substitutability factor for both options (preparing for reuse and recycling) is usually one, i.e. the quality of the recycled material is equivalent to that of the primary material. Finally, track ballast can be recovered through backfilling substituting other aggregates/soil in backfilling operations (in this case the substitution factor will depend on the filling materials' properties as compared to track ballast) (Arm et al., 2014). Finally, for concrete, since as mentioned before this fraction is a CDW stream belonging to both infrastructure and building activities, the main management options are reported in Section 3.1.

3. Technologies and processes for CDW management

Based on a literature review, this section summarises the main technologies (with different Technology Readiness Levels; TRL) used to manage the individual material fractions of CDW (see Table 8). Note that management options for contaminated fractions or containing hazardous substances are different from the processes used for non-contaminated wastes. Since the quantity of contaminated or hazardous waste is limited (around 1-2% of the total generated), and the management options are case-specific depending on the specific contamination levels and substances, herein only management options for non-contaminated and non-hazardous wastes are reported. A reference to the demolition process, either conventional demolition (CD) or selective demolition (SD), that foster the different management technologies is done. Besides, this section makes a reference to the enabling measure known as 'Design for Deconstruction (DfD)[8]', referring to a group of measures taken at the design phase of buildings and construction products that contribute to increasing circularity within the sector, by promoting reuse and recycling at the end of life of a building.

By way of a literature review, information was collected pertaining to different building materials and their (potential) handling in three contexts: conventional demolition (status quo), selective demolition and DfD, with the goal of describing current practice and highlighting areas for improvement. Further info on this can be accessed in a separate publication (Pristerà et al., forthcoming) that goes deeper into the analysis considering four building types differentiated on the basis of their structural material (concrete, masonry, steel and timber; this classification has its roots in the Eurocodes; European Commission, 2023b).

Table 8: Summary of management options reported in the literature for the different CDW material fractions. CD: Conventional demolition; DfD: Design for Deconstruction; SD: Selective demolition.

Waste fraction	Enabling process/ measure	Management option	Main output	Potential material substituted	Reference	TRL
Concrete	SD, DfD	Preparing for reuse	Concrete material	Concrete	(Marsh et al., 2022)	9
	CD, SD	Recycling	Cementitious material Recycled aggregates	Cement Sand/Gravel	(Gebremariam et al., 2020; C. Zhang et al., 2020)	7-8
	CD	Recycling(1)	Recycled aggregates	Sand/Gravel	(C. Zhang et al., 2020)	9
	CD	Landfill	-	-	(Data by Ecoinvent)	9
Wood	DfD, SD	Preparing for reuse	Structural timber/wood	Wood	(Whittaker et al., 2021)	9
	SD, CD	Recycling	Particle board	Particle board	(Faraca, Tonini, et al., 2019)	9
	SD, CD	Incineration	Electricity & heat	Electricity & heat	Multiple refs(2)	9
	CD	Landfill	-	-	(Data by Ecoinvent)	9

[8] Design for adaptability, design for disassembly, design for longevity and durability and reversible building design, henceforth grouped under the term design for deconstruction (DfD), are all methods by which this goal can be achieved.

Steel	DfD, SD	Preparing for reuse	Steel	Steel	(Coelho et al., 2020)	9
	SD, CD	Recycling	Iron ingot	Iron ingot	(Rigamonti et al., 2009)	9
	CD	Landfill	-	-	(Data by Ecoinvent)	9
Aluminium	DfD, SD	Preparing for reuse	Aluminium	Aluminium	(Diyamandoglu & Fortuna, 2015)	9
	SD, CD	Recycling	Aluminium ingot	Aluminium ingot	(Rigamonti et al., 2009)	9
	CD	Landfill	-	-	(Data by Ecoinvent)	9
Plastic PVC	SD, CD	Recycling (mechanical)	Polyvinylchloride	Polyvinylchloride	(Faraca, Martinez-Sanchez, et al., 2019)	9
	SD, CD	Recycling (chemical)	Polymer Base chemicals	Polymer Base chemicals	(Lase et al., 2023)	4-9
	SD, CD	Incineration	Electricity & heat	Electricity & heat	Multiple refs(2)	9
	SD, CD	Landfill	-	-	(Data by Ecoinvent)	9
Plastic EPS	SD, CD	Recycling (mechanical)	Polystyrene	Polystyrene	(European Commission, 2023a)	9
	SD, CD	Recycling (chemical)	Polymer Base chemicals	Polymer Base chemicals	(Lase et al., 2023)	4-9
	SD, CD	Incineration	Electricity & heat	Electricity & heat	Multiple refs(2)	9
	SD, CD	Landfill	-	-	(Data by Ecoinvent)	9
Gypsum	SD, CD	Recycling	Plasterboard	Plasterboard	(Pantini et al., 2019)	9
	SD, CD	Recycling (retarder in cement)	Recycled gypsum	Natural gypsum	(Pantini et al., 2019)	9
	SD, CD	Recycling (in sewage sludge treatment)	Recycled gypsum	Natural gypsum	(Pantini et al., 2019)	9
	SD, CD	Recycling (agriculture)	Recycled gypsum	Lime	(Pantini et al., 2019)	9
	SD, CD	Landfill	-	-	(Data by Ecoinvent)	9

Ceramic & Tiles	SD, DfD	Preparing for reuse	Ceramic material	Ceramic material	(Whittaker et al., 2021)	9
	SD	Recycling	Cementitious material	Cement	(Fořt & Černý, 2020)	6-7
	SD, CD	Recycling(1)	Recycled aggregates	Sand/Gravel	(Fořt & Černý, 2020)	9
	CD	Landfill	-	-	(Data by Ecoinvent)	9
Glass wool	SD	Recycling	Glass wool fibres	Natural fibres	(Hendriks & Janssen, 2001) (Väntsi & Kärki, 2014)	8
	SD, CD	Recycling	Recycled aggregates	Sand/Gravel	(Väntsi & Kärki, 2014)	6-7
	CD	Landfill	-	-	(Data by Ecoinvent)	9
Stone wool	SD	Recycling	Stone wool fibres	Natural fibres	(Hendriks & Janssen, 2001) (Väntsi & Kärki, 2014)	8
	SD, CD	Recycling	Recycled aggregates	Sand/Gravel	(Väntsi & Kärki, 2014)	6-7
	CD	Landfill	-	-	(Data by Ecoinvent)	9
Bricks	SD, DfD	Preparing for reuse	Brick	Brick	(REBRICK, 2013)	7-9
	SD	Recycling	Cementitious material	Cement	(Fořt & Černý, 2020)	6-7
	CD	Recycling(1)	Recycled aggregates	Sand/Gravel	(Fořt & Černý, 2020)	9
	SD	Recycling	Alkali activated blocks	Concrete	(Fořt & Černý, 2020)	6-7
	CD	Landfill	-	-	(Data by Ecoinvent)	9
Glass	SD, DfD	Preparing for reuse	Glass	Glass	(Pristerà et al., forthcoming)	9
	SD, DfD	Recycling	Flat glass	Flat glass	(Rigamonti et al., 2009)	9
	CD	Recycling	Other glass products (container glass)	Container glass	(Hestin et al., 2016)	9

	CD	Recycling	Recycled aggregates	Sand/Gravel	(Mohajerani et al., 2017)	9
	CD	Landfill	-	-	(Data by Ecoinvent)	9
Excavated soil	-	Preparing for reuse	Soil	Sand/Gravel	(Kataguiri et al., 2019)	9
	-	Recycling	Individual components (sand, clay)	Sand/Gravel Clay	(Huang et al., 2022)	9
	-	Recycling - stabilisation	Stabilised soil	Concrete or Sand/Gravel	(Firoozi et al., 2017)	9
	-	Recovery - backfilling	Soil	Sand/Gravel	(Guyer, 2012) (Haas et al., 2020)	9
	-	Landfilling	-	-	(Hale et al., 2021).	9
Dredging spoil	-	Preparing for reuse (use on aquatic habitat)	Dredged sediments	Depending on the use: Sand/Gravel Fertiliser	(Maryland Department of the Environment, 2017)	9
	-	Recycling (use on upland)	Dredged sediments	Depending on the use: Sand/Gravel Fertiliser	(Bates et al., 2015) (Apitz, 2010) (Ferrans et al., 2022).	
	-	Recycling	Individual components (sand, clay)	Sand/Gravel Clay	(Henry et al., 2023)	9
	-	Recycling - stabilisation	Stabilised sediments	Concrete or Sand/Gravel	(Svensson et al., 2022)	9
	-	Recovery - backfilling	Dredged sediments	Sand/Gravel	(Apitz, 2010)	9
	-	Disposal (landfilling/open sea)	-	-	(Bates et al., 2015; Svensson et al., 2022)	9

(1) This can also be considered as recovery – backfilling depending on final use.
(2) Tonini et al. 2013; ARC, 2015; Martinez-Sanchez et al. 2015; Bisinella et al. 2018; Fruergaard et al. 2010.

Source: Own elaboration.

3.1 Established and prospective recycling and preparing for reuse processes

What follows is a description of typical recycling and preparing for reuse processes that may apply to the individual fractions of CDW, but that are not necessarily applied in all circumstances and across all EU Member States. Additionally, possible recycling processes that may apply in a future perspective are also briefly outlined to the extent that these descriptions are available in the techno-scientific literature or from stakeholders. Note that, while preparing for reuse is currently marginally taking place and only for some niche-applications, we nevertheless describe the possible enablers for increased (preparing for) reuse in selective demolition or DfD (largely taken, as mentioned before, from the study of Pristerà et al. (forthcoming)). Note that, for simplicity, we generically talk about 'reuse', indicating both indirect and direct reuse and preparing for reuse. While landfilling and incineration (e.g. for high calorific content waste such as wood or plastics) are also mentioned as possible treatment routes for CDW material fractions, they are not described in detail as these pathways are well-known and thoroughly detailed elsewhere.

3.1.1 Concrete

Following conventional demolition, concrete waste is typically used in the production of recycled aggregates (RA), which are then used mainly for road construction and backfilling although higher value applications are in principle possible (e.g. in concrete production). The use of RA from concrete waste (typically technically referred to as recycled concrete aggregates (RCA) or sometimes coarse recycled concrete aggregates (CRCA)) for the production of structural concrete is currently limited as suggested in a recent JRC study (Pacheco et al., 2023). The authors underline that, in most Member States, recycled aggregate concrete (RAC) is hardly (or not at all) produced (RA account for only 8.2% of all aggregates produced in the EU in the year of 2019). The same authors estimate that 10-20% incorporation of RCA in current structural concrete production would be realistic (i.e. indicating a possible technically feasible recycled content for concrete in the EU market). Note that the European Standard EN 206 allows for RA to be used with a maximum replacement percentage of 30% of the total aggregate bay mass for most concrete applications. If the concrete fraction is contaminated by other materials, notably organics and sulphates, landfilling is also a possible management option. For a thorough description of the state-of-play for RA market, the readers are referred to the detailed study by Pacheco et al. (2023).

Recycling

As for recycling, there are different technologies as described by C. Zhang et al. (2020) or Pacheco et al. (2023). The simplest and most widely applied one relies on a crushing process in order to mostly produce RA, while a minor fraction is RCA. To produce the RCA, the wet process is widely used (note that a dry process is also possible) producing coarse aggregate and two by-products: sieve sand and sludge. The former does not meet the quality standard of fine concrete aggregate; therefore, it cannot be used in new concrete manufacturing, and it is usually either used like RA for road construction or backfilling operations or disposed of in site elevation. Sludge is subject to waste operations and sent to landfill (C. Zhang et al., 2020). When selective demolition is implemented, the inert waste fraction normally has fewer impurities and can be used for the production of RA to be used for the production of structural concrete. This is typically recognised as a higher value application because of the circular and higher value of the end application (bounded use in structural concrete relative to less demanding unbounded use in road basis and backfilling) and because of the higher revenues for the recyclers. Potentially, the material can then be recycled multiple times.

Perspectives for recycling

An enhanced recycling process has been studied in the literature to produce a secondary aggregate that can be used for concrete production (Gebremariam et al., 2020; C. Zhang et al., 2020). This process is an extension of the process producing RA explained before and includes two innovative technologies to improve recycling of concrete waste, namely Advanced Dry Recovery (ADR) and Heating Air classification System (HAS). While ADR is used to sort out clean coarse aggregates for use in the production of structural concrete, HAS is used to produce clean fine aggregates by heating and separating the ultrafine hydrated cement components that can be used to replace virgin cement (i.e. relative to the production of solely RA, this allows to recover a share of the treated concrete waste as cement).

Perspectives for (preparing for) reuse

DfD measures aimed at enabling reuse include modular construction and prefabrication (i.e. the use of precast concrete elements). The use of dry mounting jointing methods, or other removable connections, can further facilitate disassembly and reduce the damages incurred by the concrete elements during the process. DfD can have varying results depending on the building element to which it is applied: e.g. precast columns and beams can often be recovered and reused, though there is a lack of an established market; the right type of joint can make it possible to reuse concrete floor systems and precast concrete facades; interlocking concrete blocks can be used to building walls that can be easily disassembled.

3.1.2 Wood

Wood waste from CDW is sometimes contaminated with preservatives and therefore classified as wood waste to be treated via incineration, other forms of energy recovery or landfilling (e.g. category III-IV in DK, Germany (Höglmeier et al., 2017)).

Recycling

The most common recycling process for wood waste is the production of particleboard panels. During this process, the waste undergoes pre-treatment (wood waste shredding followed by sieving, milling, and sorting of metals and other impurities contained in wood waste), then it is dried down to a ~6% moisture content (as percent of the wet weight, % ww) and sprayed with urea-formaldehyde resin, before being hot-pressed into a mat (Faraca, Tonini, et al., 2019). Wood waste constitutes ca. 43% ww of the feedstock for particleboard in Europe (80% when including primary processing by-products such as sawdust and offcuts), while urea-formaldehyde resin typically constitutes ca. 10% ww of the final product (Rivela et al., 2006). Any wooden by-products originating from the wood waste reprocessing stage are usually combusted in small-size biomass boilers. Foreign materials found in the collected wood waste are generally separated at collection points and sent to recycling (ferrous and non-ferrous metal impurities), landfilling (glass, stones, composite building materials), and incineration (plastics, textiles, cardboard, garden waste). Fly and bottom ashes originating from the combustion and incineration processes are generally sent to landfills for fly and bottom ash, respectively.

Perspectives for (preparing for) reuse

Following selective demolition, structural timber sections hold great potential to be reused with minimal treatment (e.g. cleaning and cutting), provided that they are free of damage. If not suitable for structural elements, timber could still potentially be reused in non-structural elements (Whittaker et al., 2021). Reuse is possible, to some extent, for both lightweight and heavyweight construction; however, large timber elements designed for key structural roles can be challenging to reuse or recycle effectively, as they are often used in combination with other materials from which separation is difficult. Smaller wood elements, as opposite to structural timber sections, are easier to damage during the demolition process and it is therefore more common for them to be recycled, incinerated or landfilled.

DfD measures have different applications in lightweight and heavyweight construction: in the former case, they tend to focus on individual building elements (e.g. walls), while in the latter they also include the production of three-dimensional re-usable modules. DfD approaches aimed at facilitating reuse include prefabrication and modular construction, as well as a focus on the employment of easily identifiable and removable connections (e.g. mechanical connections, metal plate connectors), in order to reduce any potential damage to building components. Moreover, it is often advisable to avoid superfluous treatments and finishes, thus reducing contaminants which may compromise the direct reuse potential of timber elements.

3.1.3 Metals: steel and aluminium

Recycling

Virtually, almost all steel and aluminium are collected for recycling regardless of the type of demolition process. Even when conventional demolition practices are applied, leading to collection of mixed CDW, downstream sorting processes can efficiently recover metals via advanced sorting technologies (X-ray fluorescent, Eddy current, laser induced breakdown spectroscopy, etc.) which are increasingly used to obtain high quality metal scrap fractions. In the recycling process producing iron scrap, iron waste is cut/sheared; a magnetic separator is then used to remove impurities, such as paper, plastics and non-ferrous metals. The separated ferrous metals thus obtained are cleaned at 90–95% and can be sent directly to a steel smelter (Rigamonti et al., 2009). The reprocessing of scrap ferrous metal is a well-established industry. As for aluminium, the recycling process, which results in the production of aluminium ingot, consists of aluminium waste undergoing pyrolysis and then being melted in a rotary kiln fed with natural gas. The resulting ingots are then sent to foundry for remelting (Rigamonti et al., 2009). The reprocessing of scrap aluminium metal is a well-established industry.

Perspectives for (preparing for) reuse

Selective demolition can enable preparing for reuse of selected components as steel purlins, columns and rafters, by reducing the damage incurred when the building is demolished. Moreover, steel products are easily prepared for reuse owing to the methods of construction used, including use of bolted connections (Coelho et al., 2020). Similarly, selective demolition can enable preparing for reuse of aluminium products; that is the case for window frames, which can be collected from a building and prepared for reuse in their original application.

By implementing DfD measures, reuse can be enabled to different degrees, from in-situ reuse without component removal from the structure, to reuse of the whole structure in another location, to reuse of specific building components and their constituent products. Prefabrication and modular construction are effective DfD strategies for steel, as they are for timber and concrete, and bolted connections are recommended as a way to facilitate disassembly and, subsequently, reuse; fire protectants, however, can be an obstacle to the reuse of steel if they cannot be safely removed.

3.1.4 Polyvinylchloride (PVC)

Following conventional demolition, PVC is sometimes landfilled or treated in waste-to-energy treatment plants such as in the RecoChlor project[9], in which chlorine from difficult-to-recycle EoL PVC products is recovered and recycled by producing hydrochloric acid (HCl), which is then reused in the chemical industry to obtain new products.

Recycling

Following conventional demolition, soft polyvinylchloride (PVC) tends to be collected, sorted and sent to recycling to produce roofing sheets, while hard PVC is often recycled to PVC dust, chips and granulate. PVC pipes, in particular, can be recycled into new pipes. Selective demolition generally leads to increased recycling rates and better recycled material quality.

In the mechanical recycling process producing PVC, waste undergoes a pre-treatment where it is shredded and sieved. Metal and other impurities are removed, and the plastic flow is sent to further processing and mechanical recycling. Any remaining materials are incinerated. The plastic reprocessing steps include grinding, washing, drying and pelletising into recycled pellets that substitute for corresponding virgin materials (Faraca, Martinez-Sanchez, et al., 2019).

While mechanical recycling is a more established recycling option due to low energy resource demand, chemical recycling technologies are being developed to foster recycling and avoid reducing the quality of the recycled material relative to the quality required for plastics applications in building and construction (e.g. by solvolysis, dissolution, pyrolysis, gasification). This is exemplified in the VinylPlus project[10].

Perspectives for recycling

Via selective demolition, window profiles made of PVC can be recycled and used in the production of new profiles, though this is a niche application. Direct reuse is a marginal option.

3.1.5 Expanded polystyrene (EPS)

Following conventional demolition, EPS-based insulation is typically landfilled (even if officially banned within the EU) or incinerated. It is important to highlight that polymer-based external thermal insulation composite systems are often difficult to dismantle and individually collect, but EPS-based insulation in other applications such as roofing or flooring is easy to dismantle during demolition since it is mechanically fixed.

Recycling

Expanded polystyrene (EPS) can go through closed-loop recycling, to produce new insulation, or open-loop recycling, to produce lightweight concrete, car parts, etc. In the recycling process producing EPS, plastic EPS is shredded to the right dimension and separated from other impurities. Considering that 98% of EPS actually consists of air, solutions involving EPS compression are recommended. While the residues of waste products are generally incinerated, a recycling process based on a solvent-based separation (dissolution) can be applied (Garcia-Gutierrez et al., 2013). The resulting granules can be melted and remoulded into various products, usually the same product from which it came.

[9] Within VinylPlus project - https://www.vinylplus.eu/
[10] VinylPlus project - https://www.vinylplus.eu/

Perspectives for recycling and (preparing for) reuse

Selective demolition is expected to increase recycling rates of insulation plastic materials, while DfD can be applied to select the most appropriate insulation materials, in order to enable reuse, when possible, or further foster recycling. For instance, sprayed insulation (e.g. cellulose fibre, urea formaldehyde) should be avoided, as it is difficult to salvage during deconstruction. On the other hand, blown insulation can be safely extracted using appropriate techniques. Additionally, slab insulation solutions can be reused, though from a practical perspective the difficulty in reusing them lies mostly in the type of adhesive used.

3.1.6 Gypsum

Following conventional demolition, gypsum is generally landfilled. However, plasterboards could be recycled into new plasterboards (closed loop) or used in cement production or as a soil improver (open loop).

Recycling

In the recycling process of gypsum, recyclers adequately segregate the plasterboard material (around 84% of the material), from cellulose materials (15.2%) and ferrous metals (0.02%) (to be recycled as well). Then, different recycling routes are possible, according to Pantini et al. (2019). In the recycling process producing new plasterboards (up to the recommended maximum content of recycled gypsum in new plasterboards of 30%), gypsum waste is already suitable for the use in the manufacturing process, as natural gypsum, when the dimension is below 15mm. Thus, shredding and secondary milling machines might be needed. It is recycled through a cycle of calcination and rehydration, requiring a relatively pure starting material (Vrancken & Laethem, 2000). Other recycling pathways will produce recycled gypsum to be used as additive in the cement production (with an addition limited at 5% of natural gypsum supply to avoid technical problems and a substitutability factor of 0.99:1 per kg natural gypsum), or in sewage sludge treatment plants for further use in agriculture (with no restriction of dosage as long as the final product has a CaO content of at least 15% and a substitutability factor of 0.9:1 per kg of natural gypsum), and the direct use in agriculture to improve soil properties is also an option (to substitute agricultural lime with a substitutability factor of 0.37:1).

Perspectives for recycling

The results of applying selective demolition on gypsum waste recycling vary depending on the gypsum product being targeted. Where plasterboards are concerned, selective demolition leads to better waste segregation and, therefore, it facilitates recycling to produce new plasterboard. Selective removal of gypsum plaster, on the other hand, is a labour-intensive process, and it is mainly undertaken with the goal of removing impurities from the stony fraction of CDW to better recycle it, rather than to recover the gypsum itself. In this context, landfilling remains a popular option even when selective demolition is applied, largely owing to economic barriers and the relatively low market value of the secondary material. The market for recycled gypsum is indeed impacted by the availability of flue gas desulphurisation gypsum.

Implementing DfD measures (e.g. dry construction methods) can contribute to reducing impurities within the waste fraction and improving its recyclability potential. However, whether recycling will actually occur still depends on the market value of the secondary material.

3.1.7 Bricks

Following conventional demolition, bricks are generally crushed together with other inert materials and used in the production of RA, to be employed in road construction and for backfilling purposes (not structural concrete). They can also be disposed of in a landfill.

Recycling

The recycling process to produce RA consists of several successive steps including handling, crushing, and screening to obtain a more homogenous ready-to-use material without impurities. A minor fraction of residues is resultant and generally sent to landfill. Typically, loading shovels, chain feeders, jaw crushers, conveyor belts, and vibrating devices powered by diesel or electricity are used in this process (Fořt & Černý, 2020). The produced RA are typically used in road construction and for backfilling purposes (not structural concrete; Pacheco et al. (2023)).

Perspectives for recycling

A second recycling process produces material utilised as a Portland cement replacement, and it is an extension of the RA recycling process introduced earlier, with the main scope of producing a finer fraction, typically with

d_{50} of ~50 µm (Fořt & Černý, 2020). For this purpose, additional milling and vibration steps are employed. The last important step aimed at the achievement of pozzolanic activity of such material comprises drying at 70 °C to remove excess water (Fořt & Černý, 2020). A third recycling process produces alkali activated blocks utilised as concrete replacement, and similarly to the previous technology the waste brick needs to be processed first to obtain a fine fraction with d_{50} ~ 50 µm and then dried. Additionally, alkaline activators (e.g. sodium hydroxide, sodium silicate) are used in the mix design. The mixture usually contains sodium hydroxide pellets, sodium silicate (water glass) with SiO_2/Na_2O molar ratio of 1.6, sand, and finely milled brick dust, and provides mechanical performance comparable with common concrete-based composites (Fořt & Černý, 2020). These are however recycling processes with very low TRL value and not fully demonstrated at commercial scale.

Selective demolition leads to a decrease in impurities, and consequently, to the improved quality of the RA produced from this waste fraction. Though not yet implemented at large scale, this opens up the possibility of being able to use these aggregates in structural concrete. Other niche applications include the possibility of using the fine fraction obtained during the recycling process to produce masonry mortar, and the potential use of brick waste to substitute clay soil in the production of new unfired bricks.

Perspectives for (preparing for) reuse

Via selective demolition, it is possible to prepare CDW bricks for reuse. The REBRICK project (REBRICK, 2013) develops and demonstrates the technical viability to produce reusable bricks with market specifications. The CDW passes through an equipment that separates mortar and other materials, such as wires, cement, and wood from the bricks, and thus another system that separates whole bricks from damaged bricks. Thus, through a vibration-based system technology, concrete and cement from old bricks are cleaned and then reused. After being cleaned, they are manually sorted, and automatically stacked and wrapped.

Where DfD is concerned, the main strategy emerging from the literature review consists in the construction of mortar-free structures, in which the bricks are connected by way of steel plates and wall ties. Prefabrication of modular units further increases the reuse potential of this material.

3.1.8 Ceramics and tiles

Recycling

Following conventional demolition, ceramic and tiles are normally used to produce low-quality RA for road construction and backfilling, similarly to bricks. The RA produced from ceramics and tiles cannot be used for the production of structural concrete (Pacheco et al., 2023).

Perspectives for recycling

When selective demolition is applied, ceramics and tiles present several additional recycling opportunities, such as the possibility of being used as supplementary cementitious material or as a precursor to manufacture an alkali-activated binder (when comprising both red brick and tile waste) (Whittaker et al., 2021). These are however recycling processes with a very low TRL value and not fully demonstrated at commercial scale.

Perspectives for (preparing for) reuse

Ceramic normally undergoes the same processes described for (clay) bricks. Following selective demolition, the recovered ceramics can be prepared for reuse as floor or wall tiles through blending the ground ceramic fraction in resin and allowing it to harden in moulds (Whittaker et al., 2021). DfD solutions can target specific ceramic elements, such as floor tiles, to enable their reuse.

3.1.9 Mineral insulation: glass wool and stone wool

Upon conventional demolition, mineral insulation is mostly not recycled according to techno-scientific literature (e.g. see Väntsi & Kärki (2014)) but some stakeholders mention recycling pathways for the brick industry as a possible avenue (specifically, mineral wool can be used in the production of masonry mortar) or as RA. Whenever the waste is contaminated, it is landfilled.

Recycling

Although there are several different options for glass wool and stone wool waste recycling in the literature, most of them refer to mineral wool waste from the production process, being more restricted when the origin is CDW. Thus, Väntsi & Kärki (2014) report the possibility to use mineral wool waste combined with low-melting illite clays to produce composite ceramics. Other products using recycled mineral wool waste, by dispersing them in a solution of cold water and blended with other fillers and binder ingredients could be ceiling tiles substituting other mineral fibre materials (with a substitution ratio of up to 1:1). Another possibility is to use mineral wool waste in cement-based composites as coarse aggregate or fine aggregate substituting natural aggregates significantly improving the compressive strength, splitting tensile strength, absorption, resistivity, and chloride-ion penetration resistance of the cement-based composite. Other recycling options are reported within the WOOL2LOOP project (WOOL2LOOP, 2020) such as the remelting of specifically glass wool waste by burning it with natural gas with the aid of oxygen input and thus using the product again in construction sites, the addition of mineral wool previously pre-processed or milled to the expanded clay manufacturing process, or the complement of virgin materials in stone wool products. Hendriks & Janssen (2001) report a process to recycle glass wool through gasification in an oxygen-free environment (nitrogen), using the output fibres to produce new glass wool.

Perspectives for (preparing for) reuse

According to Rudjord (2018), reuse is not an option for glass wool waste, since the fibres are not of uniform size and the amount of organic content is too large. On the other hand, Väntsi & Kärki (2014) and Hendriks & Janssen (2001) report the possibility of using mineral wool as an artificial substrate to grow various plants in soilless cultures (e.g. stone wool is currently the most widely used soilless medium). However, this would rather be classified as a recycling or recovery pathway (not reuse) as the mineral wool reuse is not meant for the purpose for which the material was originally produced and marketed. For stone wool, similarly, no information on case studies documenting reuse were found in the technical and scientific literature. However, some companies claim that is suitable for reuse as thermal insulation at a new site, provided that the material can be extracted intact from its previous location.

3.1.10 Glass

Following conventional demolition, glass is crushed and landfilled with other waste materials or recovered for low-grade applications (e.g. using glass cullet in the production of RA for road construction). Via selective demolition, flat glass can be recycled into container glass or, more rarely, into new flat glass. Generally, glass has a relatively low reuse potential due to its fragility and the fact that most products are custom-made to satisfy the requirements of the building design.

Recycling

Via conventional demolition, glass ends up in the mixed CDW and can be used to produce RA. In the recycling process producing RA from mineral CDW, the inclusion of fine waste glass aggregates such as glass powder in concrete mixtures is allowed under certain conditions. While the workability of concrete containing crushed glass waste in lieu of conventional fine aggregates is still under discussion, especially concerning its physical properties and associated functionalities, many studies have identified this type of management process as technically feasible (Abd-Allah et al., 2014; Batayneh et al., 2007; Chen et al., 2006; Ismail & AL-Hashmi, 2009). Glass powder is glass that is milled down into very small particles, with a typical median grain size between 30 µm down to as fine as 0.1 µm. In the recycling process producing RA several successive steps including transport, separation and milling to obtain more homogenous ready-to-use material without impurities are applied, while a minor fraction of residues is sent to landfill (Aslani et al., 2023; Mohajerani et al., 2017).

Perspectives for recycling

Via selective demolition, windows are dismantled so that the glass is separated from the frame. The resulting glass waste contains a reduced quantity of impurities and is therefore more likely to be usable in the production of new glass. The recycling process involves different activities such as manual selection, shredding, screening, magnetic and non-magnetic separation to remove impurities and inert materials (ceramics and gravels) and to obtain a proper size distribution. The glass cullet is then delivered to a glass manufacturing plant, where it is used in the production of new glass, together with ordinary virgin raw materials (silica, calcium carbonate, sodium hydroxide, additives) (Rigamonti et al., 2009). The presence of cullet, which is characterised by a lower melting temperature than virgin raw materials, allows the glass furnace to be operated at a lower temperature, thus leading to significant primary energy savings (up to 20% when 80% of cullet is utilised in the kiln feeding). Once the glass cullet has been produced, it can be used either to produce new flat glass, or other types of glass (especially hollow glass, also known as container or packaging glass) or fiberglass (also known as glass wool) (Hestin et al., 2016).

Perspectives for (preparing for) reuse

In terms of DfD strategies, the most common one consists of the use of dry connection methods between glass panel and window frame; this technology facilitates dismantling and can enable reuse. Repurposing is also an option, meaning that glass obtained during disassembly or renovation can be used for indoor applications, which do not require high thermal standards. As mentioned earlier, glass has a relatively low reuse potential due to its fragility. Consequently, recycling is expected to be the most common solution even when DfD is implemented.

3.1.11 Excavated soil

A recent review from Scialpi & Perrotti (2022) summarises the studies related to the sustainable management of excavated soils in urban areas. The excavated soil waste management options reported are the following: preparing for reuse, recycling into individual factions or via stabilisation, and recovery via backfilling.

Recycling

The direct application of excavated soil (e.g. in dams or roadbeds) is usually difficult due to its poor strength and, thus it must be stabilised (Huang et al., 2022). Stabilisation is a technology for improving geotechnical properties in terms of increased strength, reduced permeability and compressibility of soil (Magnusson et al., 2015). This technique, considered herein as recycling, involves the use of blends of soil and binders and the most reported stabilisation methods in the literature are the ones using cement, lime, fly ash, and fibres (Firoozi et al., 2017). Other methods using different types of binders include rice husk ash, bituminous material, geo-textiles and synthetic materials, and several recycled and waste products (Afrin, 2017). The application of different binders and the dosage of the blend depends on the type of soil (e.g. clay content) and the required physical and mechanical properties (e.g. strength and stiffness) of the blend. Namely, the stabilisation with lime is suitable for fine-grained soils, meanwhile the cement can be used for stabilising nearly all types of soils (whenever the organic content is limited to 2%) (Patel, 2019). There are other lesser-known technologies for soil modification such as fluidification that use fluidifiers and plasticisers to stabilise the soil (Michalcikova & Drochytka, 2018). It can be done on-site or off-site. For the former, excavation of the material is not needed. The off-site option involves the excavation of the material and additional transport of the excavated soil to the site where it is going to be used. The process includes the production and transport of the binder to the site and the mixing with the binder.

Excavated soil is recycled in various ways to replace traditional construction materials (N. Zhang et al., 2020). It can be done through on-site recycling plants (that normally function with diesel instead of electricity) or through stationary plants that require additional transport comparing to the on-site plants. The general recycling process (Huang et al., 2022) mainly consists of a screening and sieving phase to assure the adequate size of the soil. It includes an iron removal phase to collect any scrap steel that could cause damage to the separation equipment that extract the recycled sand from the soil and send it to the washer for further cleaning. The possible coarser aggregates are blasted to grind them into sand and washed. In the separation equipment, the other components of the soil (mainly clay and silt) are headed to the slurry tank where flocculating agents are used to precipitate them. Those are sent to the filter press to make the filter clay cakes. The filter cake can be either landfilled or further recycled mixing it with cement and curing agents (e.g. to produce RAC) (Xu et al., 2022). Other authors in literature reported recycling processes for excavated soil in line with the general one explained herein but extended to final products such as recycled bricks (baking free bricks or baked bricks) (N. Zhang et al., 2020), cement treated base materials (Xu et al., 2022), or concrete blocks (Luo et al., 2022). Note that the recycling to their individual components is rarely done due to economic constraints.

Preparing for reuse

A soil can be prepared for reuse and applied either on-site[11] or in other locations/projects, whenever allowed by legislation. Determining whether a material is suitable for reuse in an application depends on several practical and site-specific considerations and factors. Thus, to prepare for reuse and reuse excavation soils, different geotechnical properties (e.g. cohesion and friction angle, grain size distribution, swelling and loss of strength by wetting, organic matter content, hydraulic conductivity) of the receiving project (e.g. roadbeds, paving layers, vegetation cover replacement) have to be fulfilled requiring a characterisation of physical and mechanical properties (e.g. shear strength, dry density), as well as the compliance with environmental safety standards (Kataguiri et al., 2019). This management option includes the relocation of the soil within the receiving site using machinery and the compaction of the material. In the same line, the excavated soil can be used for backfilling either on-site or off-site depending on the geotechnical criteria documented and described in industrial standards (e.g. ASTM standards, AASTHO specifications).

[11] In case it is used on-site, they have to be declared previously as waste.

Recovery

In the context of construction, backfilling is the practice of refilling the excavated pit with material to reinforce and maintain the foundation of a structure or any other structural part. It is important to highlight that complying with the recovery definition, the backfilling operation must replace other materials that are not waste, and in this sense the difference between reuse and backfilling for excavated soil is just semantic, acknowledging the difference in the waste hierarchy.

Finally, other applications have been explored in the literature, such as the possibility of using excavated soils for the creation of technosoils by mixing with compost (i.e. soils designed to mimic natural soil and suitable for vegetation growth) (Fabbri et al., 2021). Further innovative management options are also under development such as using such wastes for energy storage batteries. The EU project NewSETS, funded from the EU's Horizon 2020 programme[12], is focused on different technologies to store energy. Among them, the seasonal heat storage uses sand as storage medium. Excess electricity is used for heating up the sand to a high temperature (around 500 – 600 °C), later the stored energy can be used as heat for industrial steam production or district heating, and even back to electricity, thus working as a "battery". Heat storage is not sensitive to sand grain size and might use grain sizes that are not suitable for the construction industry or were going to be landfilled (preferably high density, low-cost materials that are not from scarce sources).

3.1.12 Dredging spoils

When dredging spoils are non-toxic, they can be used on land for soil filling, construction purposes, coastal nourishment, and as an amendment in agriculture, horticulture, and forestry, as well as disposed directly in landfills or oceans whenever they comply with the pollutant-specific regulations (Crocetti et al., 2022). Thus, the dredging spoil waste management options vary depending on the level of contamination and local regulations. The reported ones in literature (Apitz, 2010; Maryland Department of the Environment, 2017) include the following: i) preparing for reuse, ii) recycling for use on land (either with or without aerobic composting), iii) recycling through stabilisation, iv) recovery through backfilling, and v) disposal. These management options require single treatment processes, or a combination of them, including: i) physical operations such as mechanical separation, dewatering, and washing; ii) chemical processes such as chemical oxidation, and solidification/stabilisation; iii) thermal processes such as desorption, thermal oxidation and immobilisation, and vitrification; and iv) biological processes (Crocetti et al., 2022). Note that most of those processes are commonly used to treat contaminated sediments, and that are typically not applied to non-contaminated dredging spoils.

Recycling

For the recycling and further use of the dredged material it requires some degree of drying or other processing. Minimal processing may consist of dewatering the material that is essential before land use due to the high-water content, and its negative influence on the subsequent transport and possible treatment/use. Different dewatering methods exist, from natural drainage and evaporation, also known as passive dewatering (Bates et al., 2015), that requires vast space and long retention times, to mechanical dewatering units such as filter presses and vacuum filters that increase performance and reduce land demand (Crocetti et al., 2022). The dewatering process can be accompanied by chemical conditioning using products such as iron and aluminium salts, or organic polymers like polyacrylamide, that helps releasing and removing the interstitial water (Crocetti et al., 2022; Zhou et al., 2021). The dewatered material is transported to the receiving site where it can be used in numerous potential beneficial uses including engineering (such as shoreline protection, reclamation and land formation (Bates et al., 2015), capping of contaminated dredged material in confined aquatic disposal or permanent cover of landfills, and agricultural use (either in agriculture for non-food crops, forestry, or horticulture) (Apitz, 2010). The recycling and further use on land includes the use as soil conditioner since the nutrients and organic matter content in theses sediments could improve the chemical status of the soil, mainly as a potential source of phosphorus (Ferrans et al., 2022). The land use can be preceded by a biological process (i.e. aerobic composting) to degrade the organic carbon and nitrogen to create a stable end product (Zhou et al., 2021). However, their use as fertilisers in agriculture is still not allowed due to lack of permissive legislation and consolidated supply chains (Renella, 2021).

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[12] The initiative has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreements no. 646039 and no. 755970 - https://www.newsets.eu/

Through recycling, various products can be obtained from the dredging sediments in different forms such as individual fractions (i.e. sand, silt and clay), or as aggregated materials such as light weight aggregates that are commonly classified as particles with very low density (i.e. clay and silt) (Crocetti et al., 2022). To obtain these products that later might be used for example in cement production, mineral processing techniques are used (Henry et al., 2023). Those can be dry techniques after the materials have been dehydrated, or wet techniques to separate the low-density fraction. These includes operations such as sieving, crushing, flocculation, and filtering (Henry et al., 2023).

The recycling through stabilisation and solidification method is normally used to fixate and encapsulate the contaminants inside the sediment by adding binders, similarly to the process explained in the excavated soil section, lowering the permeability and reducing leaching, and enhancing the strength enough to be used in construction (Svensson et al., 2022). The binders that can be used include cement, lime, fly ash, slag, etc. According to Svensson et al. (2022). In a case study in Sweden, a mixture of binders (i.e. cement and ground-granulated blast-furnace slag) and sediments is used to build the port in Gothenburg.

Recovery

Recovery through backfill as structural and non-structural fill is also a possible use of dredging spoils. Engineered fill for uses such as roadway bed material, parking lot foundation or embankment fill are possible whenever requirements (type of material and gradation, plasticity, permeability, compaction, moisture content) of the engineering plans are fulfilled. To meet the physical and geotechnical requirements, processing might be needed through dewatering, blending or amending (Maryland Department of the Environment, 2017)

Preparing for reuse

The dredging spoil can be prepared for reuse unprocessed or with minimum processing and has been used typically for beneficial use options such as aquatic and wetland habitat development, environmental enhancement (in wildlife or fishery habitats) or beach nourishment (Maryland Department of the Environment, 2017).

4. CDW treatment in the EU

This section illustrates the current situation for the CDW treatment in the EU, gathering information mainly from Eurostat and complementing it with different literature sources, as well as input from stakeholders. A summary is presented in Table 9 and Table 10.

4.1 Mineral fraction

The mineral fraction of CDW (EWC code W12.1) includes concrete, bricks, ceramic and tiles, gypsum, insulation material (here assumed as mineral, i.e. stone and glass wool), mixed construction waste, as well as track ballast and hydrocarbonised road-surfacing materials. Figure 4 shows the treatment of the mineral fraction of CDW in the analysed Member States in 2020. Based on Eurostat data (env_wastrt; European Commission, 2021c), the considered treatment options are disposal, incineration with energy recovery[13], recycling and backfilling. Disposal is herein defined as landfill, incineration without energy recovery[14] and other disposal, of which landfill is typically the dominant (> 99%) disposal method. Figure 4 shows that recycling as of 2020 is the predominant treatment option in most countries. Exceptions are countries where backfilling is the predominant treatment option e.g. Hungary, Denmark, Ireland, and Portugal where 88%, 71%, 73% and 63% of the mineral fraction of CDW, respectively, is reported to be used for backfilling purposes. Other countries such as Poland, France, Spain and Cyprus have waste disposal at higher rates than 20%. The Nordic countries are the only ones where incineration with energy recovery is practiced to some extent (more than 30% in Finland, around 9% in Sweden, and below 4% in Denmark). The official data reported do not provide further insights into the treatment of the individual fractions composing the mineral fraction, being the EU average 79% recycling, 10% backfilling, and 11% landfilling. In the attempt to close this knowledge gap, we herein report insights from the literature to date. Summary may be found in Table 9.

For the case of bricks, grey literature reports cases of reuse occurring mainly in Denmark suggesting that about 3 million bricks per year are prepared for reuse (Santoro, 2020) (corresponding to 6 600 tonnes per year and 3% of the total brick waste in Denmark[15]). This, while currently at EU level is negligible (i.e. 0.08 %), is nevertheless important to be considered for the estimation of the bricks reuse potential. Other studies (Hopkinson et al., 2018; Kay & Essex, 2012; WRAP, 2008) suggest that the rate of bricks from demolished buildings reclaimed for reuse in UK would be between 5% and 10%.

For the case of gypsum, the literature reports that ca. 10% of the gypsum waste generated in CDW is recycled (Deloitte, 2017). The remaining is thus assumed to be landfilled. As for mineral insulation materials, stakeholders pointed out that this fraction should not be assumed as part of the 'mineral fraction of CDW', as this would overestimate recycling of mineral wool. For these material fractions, data reported for 2015 by Wiprächtiger et al. (2020) for Switzerland, acknowledging the geographic restriction of the case study, shows a 2% recycling (mineral wool to mineral wool, calculated after the results presented in their material flow analysis), and since mineral wools should not be incinerated given the negligible calorific value, the remaining 98% is here assumed to be landfilled. This estimation is also supported by other evidence, for example from Flanders (Belgium) where less than 1% recycling is reported (Debacker et al., 2021). Within the same region of Flanders, Monier et al. (2011; Table 50) reported that 0% of gypsum is prepared for reuse. In the same line, ARUP (2021) states that reuse of plasterboard is currently not widespread and there are no suppliers of reclaimed plasterboards in UK.

Box 1 shows the share of the treatment options for the mineral fraction of CDW in the 27 Member States in 2016 and 2018 based on Eurostat data (European Commission, 2021b). Note that the data reported in Eurostat do not provide further insights on the treatment of the individual fractions composing the mineral fraction. Additionally, Box 1 includes a summary of the findings from the critical review of the recovery rates of CDW in the EU performed by Moschen-Schimek et al. (2023) concerning data quality and influencing factors. The authors conclude that changes in the definition of the waste treatment options over time, notably for backfilling, are a key factor to explain the inconsistencies in the reported quantities of recycling, backfilling and disposal. The EU Commission is already aware of the problems concerning backfilling data reporting and a study was published to support the definition of backfilling enabling a uniform application across Member states (European

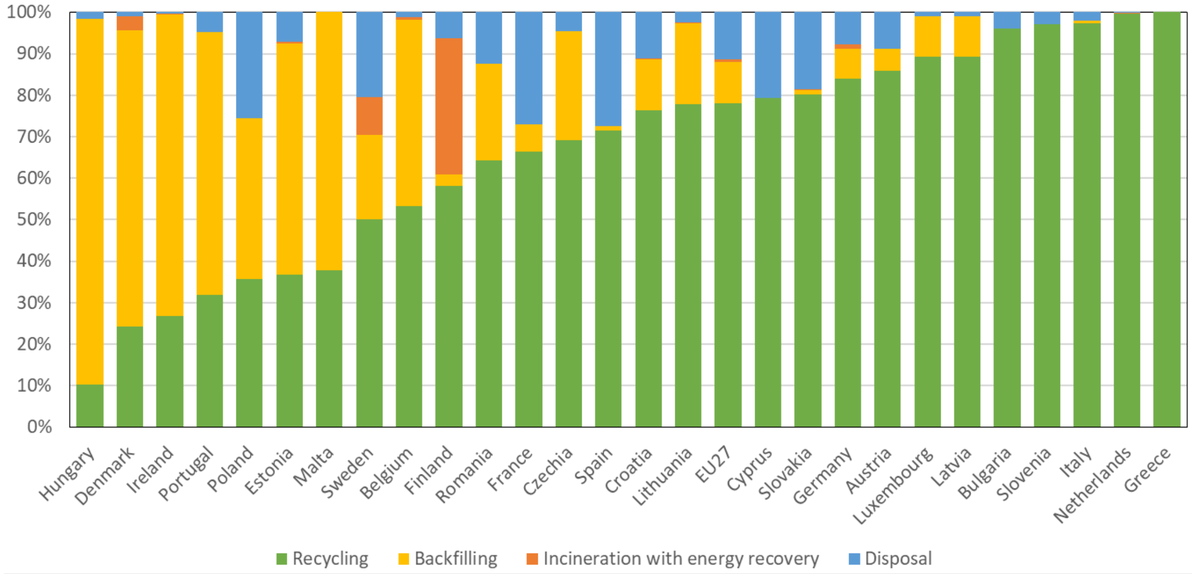
[13] "Incineration with energy recovery" represents waste incineration processes classified as R1 according to 2008/98/EC, Annex II.

[14] "Incineration without energy recovery" represents waste incineration processes classified as D10 according to 2008/98/EC, Annex I.

[15] Calculated using a weigh of a brick of 2.2 kg, and considering the waste brick quantity in Denmark of 212 000 t according to Damgaard et al. (2022).

Commission, 2020c). Figures in Box 1 (year 2016 and 2018) can be compared directly and can be compared also to Figure 4 (year 2020).

Figure 4. Treatment of the mineral fraction of CDW in the analysed EU Member States in 2020. Disposal is defined as landfill, incineration without energy recovery and other disposal. Note that data for Ireland 2020 is not available and data from 2018 has been used.



Source: Own elaboration based on data from Eurostat (env_wastrt).

Box 1. Treatment of the CDW mineral fraction: changes observed following the update of the definition of backfilling.

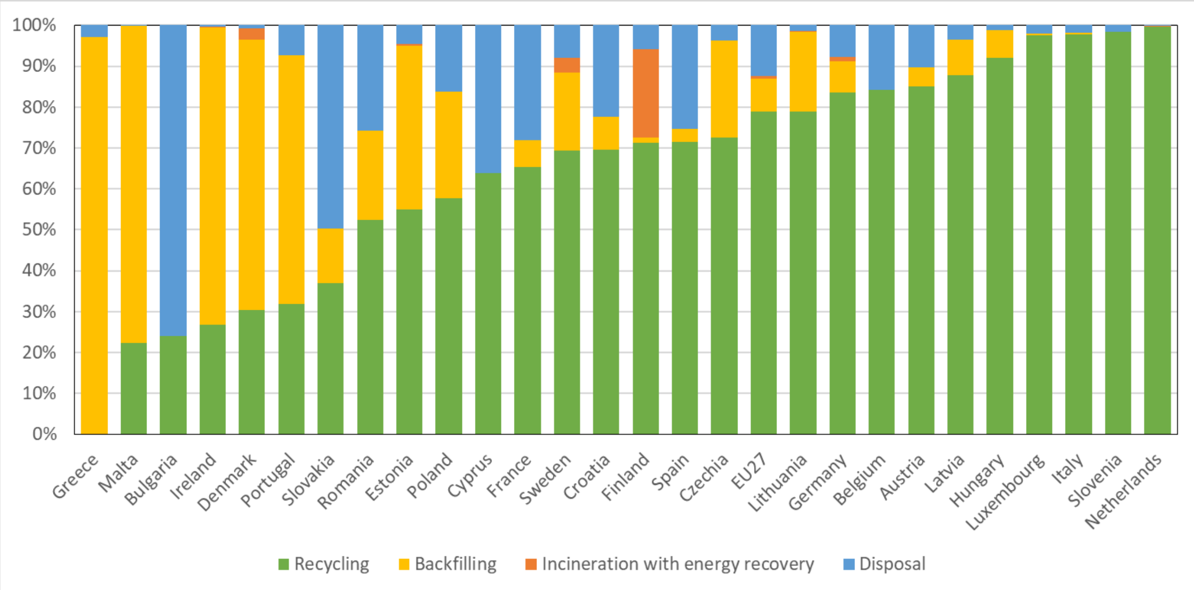
Similarly to 2020 (Figure 4), the predominant treatment option in EU in 2016 (Figure B1) and 2018 (Figure B2) was recycling (80% and 79%, respectively).

Several countries already reported backfilling as a treatment option in 2016, thereby complying with Commission Decision 2011/753/EU establishing rules and calculation methods for verifying compliance with the WFD target[16] for CDW. The Commission decision obliged Member States to report the amount of CDW used for backfilling operations separately from the amount of waste prepared for reuse, recycled or used for other material recovery operations. As described earlier the definition of backfilling in Commission Decision 2011/753/EU has been revised in the WFD as of 2018 (2018/851/EU). The revised definition might have been the cause for a decrease in the recycling rates of some Member States as more focus is put on the distinction between recycling and backfilling, and thus more mineral waste could be reported as being backfilled. Such shift in reporting should, however, not interfere with the overall recovery rate, as backfilling is still considered a recovery operation if the criteria of the definition are fulfilled. This might be the case for Denmark, that in 2016 reported 0% backfilling and in 2018 reported 66% of mineral CDW backfilled at the expense of recycling that decreased from 89% in 2016 to 30% in 2018. Similar changes happened for other Member States, even if delayed in time, not due to an actual shift in treatment methods, but rather because of the revised definition of backfilling in the 2018 amendment of the WFD, which forced Member States to thoroughly re-consider how CDW is reported. For example, Hungary reported a recycling rate of 89% and 92% in 2016 and 2018, respectively, but in 2020 that quantity was reduced to 10%, increasing the reported backfilling from 7% in 2018 to 88% in 2020.

[16] The target set in Article 11(2)(b) of Directive 2008/98/EC reads: "by 2020, the preparing for reuse, recycling and other material recovery, including backfilling operations using waste to substitute other materials, of non-hazardous construction and demolition waste excluding naturally occurring material defined in category 17 05 04 in the list of waste shall be increased to a minimum of 70 % by weight."

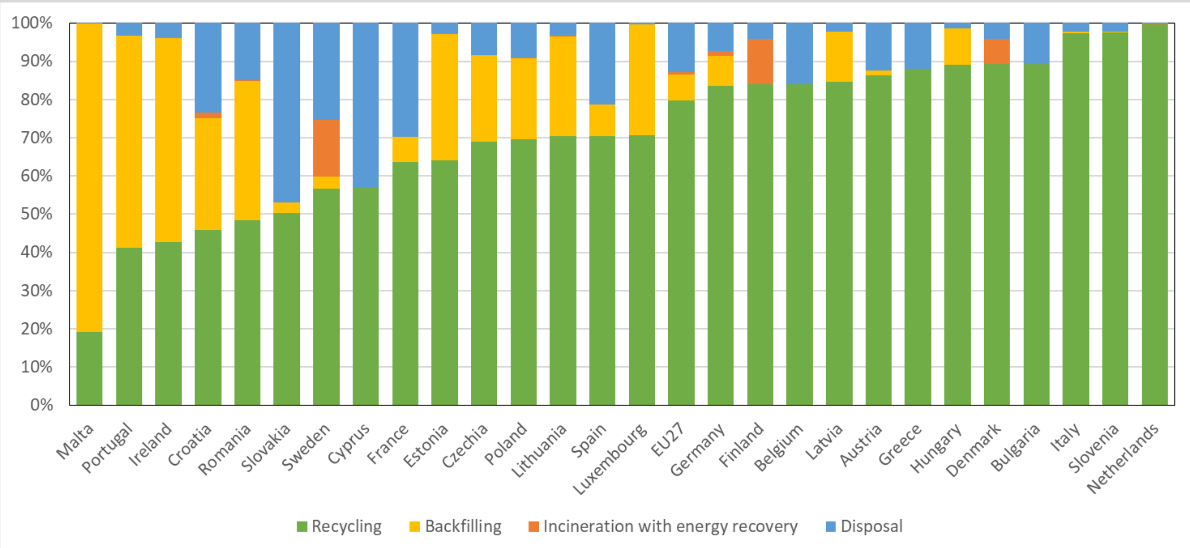
In addition, more clarification on the definitions and instructions of the reporting seem to be still needed. It is observed from the data that some countries changed the reported quantities dramatically from one reporting year to the other between the different treatment options (including recycling, backfilling and disposal). For example, Bulgaria reported in 2016 recycling rates and disposal rates of 89% and 11%, respectively. But in 2018, the rates shift to the opposite (recycling rate of 24% and disposal rate of 76%), just to shift back in 2020 to similar figures as 2016 (recycling rate of 96% and disposal rate of 4%). Greece reported in 2016 recycling rate and disposal rate of 88% and 12%, respectively. But in 2018, the recycling rate was null shifting all to backfilling (97%), just to shift back in 2020 to recycling rate of 100%. Slovakia reported in 2018 recycling rate and disposal rate of 37% and 50%, respectively. But in 2020, the recycling rate increased to 80% and the disposal rate decreased to 19%.

Figure B1. Treatment of the mineral fraction of CDW in the analysed Member States in 2018. Disposal is defined as landfill, incineration without energy recovery and other disposal.



Source: Own elaboration based on data from Eurostat (env_wastrt).

Figure B2. Treatment of the mineral fraction of CDW in the analysed Member States in 2016. Disposal is defined as landfill, incineration without energy recovery and other disposal.



Source: Own elaboration based on data from Eurostat (env_wastrt).

In a critical review on the recovery rates of CDW in the EU from Moschen-Schimek et al. (2023), the authors investigate, for 12 countries, whether they have achieved a real increase in CDW recovery rate based on technological and legal changes, or whether the increase was due to methodological changes in data reporting, and analyse three potential influencing/limiting factors: (a) not harmonised data collection methods in the analysed countries, (b) differences in national waste code systems, and (c) not harmonised application and insufficient definition of backfilling activities. The study is grounded on the evaluation of the quality reports (between 2012 and 2016) that according to the authors present good quality and can be regarded as a reliable database. The authors conclude that two of the three influencing factors, namely (a) and (c), are the most meaningful for the interpretation of the CDW recovery rates published by Eurostat. Misallocation of waste types to waste codes or treatment activities to treatment operations codes hampers comparisons both between EU countries and within one country over time. High growth rates of the recovery rates can be linked to the change of treatment codes (from disposal to backfilling codes), the change of data collection methods and the adjusted allocation of waste streams to the correct waste code.

4.2 Metals, plastic, wood and glass

CDW also contains other fractions besides those listed under the mineral fraction (EWC code W12.1), notably metals, plastic, wood, and glass. As for aluminium treatment, a study by Delft University of Technology (2004) suggests that aluminium is already largely separated and recycled from CDW (the study suggests that ca. 95% of the aluminium available in CDW was collected for recycling, as of 2004). As for steel, the high market value of steel scrap relative to other materials, make this fraction also being largely separated and recovered from mixed CDW - either during demolition or at subsequent sorting plants treating mixed CDW, in a similar fashion as aluminium. Besides, for both aluminium and steel, a quantity between 5 and 15% is prepared for reuse according to the stakeholders. This is also supported by the study of Diyamandoglu & Fortuna (2015), reporting current metal waste treatment practice at EU level as 10% preparing for reuse, 84% recycling and 6% landfill.

As for wood waste, a study by Ramboll (2018) (mainly on wooden windows and doors) reports eight countries' specific estimations of wood waste treatment for 2018. There are great variations among them, for example Austria incinerates 100%, Sweden and Germany incinerate 95% with 5% recycled, and Italy presents 80% recycled and 20% incinerated. Landfilling is estimated to 25% for France (40% recycled and 35% incinerated) and 5% for Czechia (40% recycled and 55% incinerated). To calculate an EU average, based on the construction wood waste reported in Eurostat for 2018 and based on the eight country estimations from Ramboll (2018), we perform a weighted average resulting in 73% incinerated, 21% recycled (mainly to particleboard), and 6% landfilled. The recycled quantity is then corrected to 30% following the indications of Deloitte (2017) that has a broader EU scope than Ramboll's study (however, the magnitudes are comparable). Even if some wood could in practice be prepared for reuse, as shown in some regional studies for Flanders (Belgium) that estimate a preparing for reuse of timber of 1.89% (Monier et al., 2011; Table 50), there is no data available for this treatment option at EU level. A case study from the UK (ARUP, 2021) reported that 30% of the wood available in CDW was salvaged (including softwood studding, modern staircases, mouldings, scrap timber and cheap modern furniture).

Deloitte (2017) further estimates the recycling rate of glass waste (to new glass) at 6%, validated by Glass Europe, while not reporting the end treatment of the remaining 94%. A report from Ramboll (2018) provides rough estimates for the treatment of glass waste from windows, suggesting that landfilling is the main treatment pathway (70%) complemented by recycling (30%). On this basis, we here assumed that 6% of glass waste is sent to glass-to-glass recycling while the remaining glass waste is split between landfilling (70%) and ending up as foreign material in the production of RA for backfilling (24%). Note that this is an assumption due to lack of specific data, as the RA containing glass could as well be used for road construction and thus reported as recycled. Concerning preparing for reuse of glass waste, no data at EU level is available. For the Flanders (Belgium), Monier et al. (2011; Table 50) reported that close to 0% of glass waste is prepared for reuse.

As for plastic waste, PVC and EPS are by far the main plastic fractions in CDW representing altogether about 80% of the total plastic waste found typically in CDW[17]. Concerning EPS, according to 2019 estimates by EUMEPS[18], about 10% is recycled, 66% incinerated and the remaining 24% landfilled. As for PVC, the material flow analysis of the EU PVC cycle by Ciacci et al. (2017) concludes that, for the period 1960-2012, ca. 30% of

[17] These shares are taken from Plastic Europe and used also in the previous JRC report on CDW by (Damgaard et al., 2022).

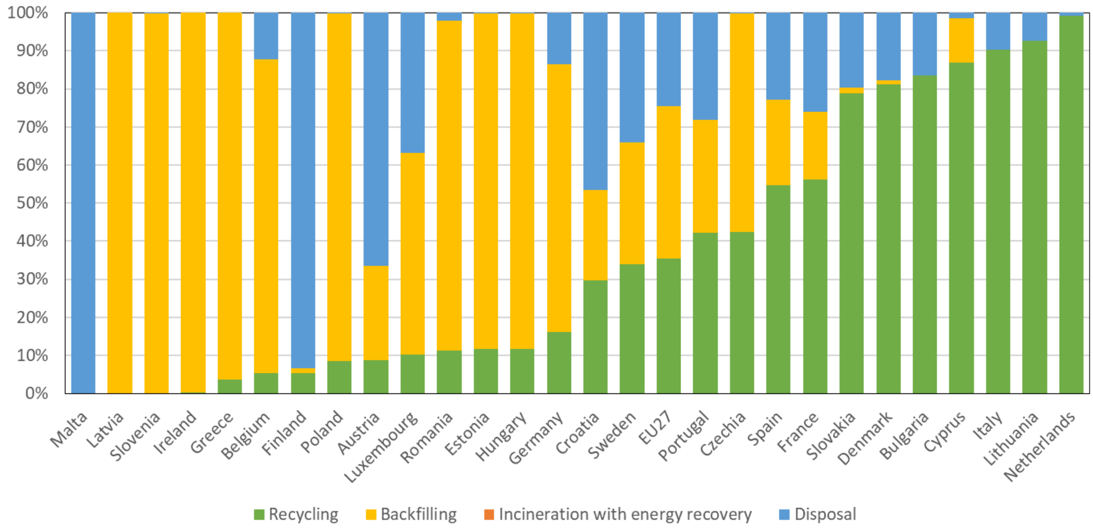
[18] Data provided to JRC by EUMEPS during an expert workshop held in 2023. Similar data are reported in a presentation power point of a study by BASF (available at https://www.psloop.eu/wp-content/uploads/2021/01/Conversio-study.pdf)

the PVC waste (from all origins, i.e. construction, packaging, and other sectors) has been collected for recycling, 12% sent directly to incineration and 58% to landfill. Notably, out of the amount collected and sent to recycling, a substantial share is rejected in the recycling process and then sent to incineration or landfill (ca. 30-50% of the PVC entering recycling) as suggested in the studies of Damgaard et al. (2022) and Ciacci et al. (2017). Concerning preparing for reuse, no data at EU level is available but within the region of Flanders (Belgium), Monier et al. (2011; Table 50) reported that around 1% of plastic is prepared for reuse. An overview of the treatment pathways for all fractions may be found in Table 9.

4.3 Soil waste

For excavated soils, we refer to the treatment data available in Eurostat (env_wastrt; European Commission, 2021c). For 2020, Figure 5 shows the treatment of the excavated soil fraction (non-hazardous) of CDW in the Member States. It is clear that there is no predominant treatment option at EU level. Some countries report shares of recycling higher than 80% such as Denmark, Bulgaria, Cyprus, Italy, Lithuania and Netherlands. On the other hand, other countries report negligible recycling shares with either 100% disposal (Malta) or 100% backfilling (Latvia, Slovenia, Ireland). Other countries present high disposal shares for this fraction such as Finland (93%), Austria (66%), and Croatia (46%). As an average, for the EU in 2020, the recovery rate of soils (calculated by the authors, even though it is not included in the official recovery rate published by Eurostat) equals about 76%. However, the quality of this data needs to be further assessed and probably the limiting factors referenced in Box 1 might also apply to soil. According to Moschen-Schimek et al. (2023), some countries such as Bulgaria and Belgium do not report any backfilling activities of soil waste even if those take place. Thus, these amounts are either reported as recovered amounts, are not classified as waste (i.e. they reach an end-of-waste (EoW) status) or they are not reported at all. This is somehow reflected in the validation report of the Eurostat data (Noel et al., 2021) for 2018 that highlights for Bulgaria that there is a significant difference between excavated soil waste generations (74 kt - see Table 2) and reported as treated (only 9%, i.e. 6.4 kt). Other countries, apart from Bulgaria, that reported in 2018 less than 50% of the generated excavated soil waste as treated are Malta (0%), Belgium (25 %), Cyprus (44%) and Lithuania (46%). Concerning preparing for reuse, no data at EU level is available but within the region of Flanders (Belgium), Monier et al. (2011; Table 50) reported that around 10% of excavated material is prepared for reuse. The Earth Cycle project lead by the city of Sevran within the Grand Paris area (France) reported that around 5% of the excavated earth managed as waste was reused (Diab, 2020).

Figure 5. Treatment of the excavated soil fraction of CDW in the Member States in 2020. Disposal is defined as landfill, incineration without energy recovery and other disposal. Note that data for Ireland 2020 is not available and data from 2018 has been used instead.

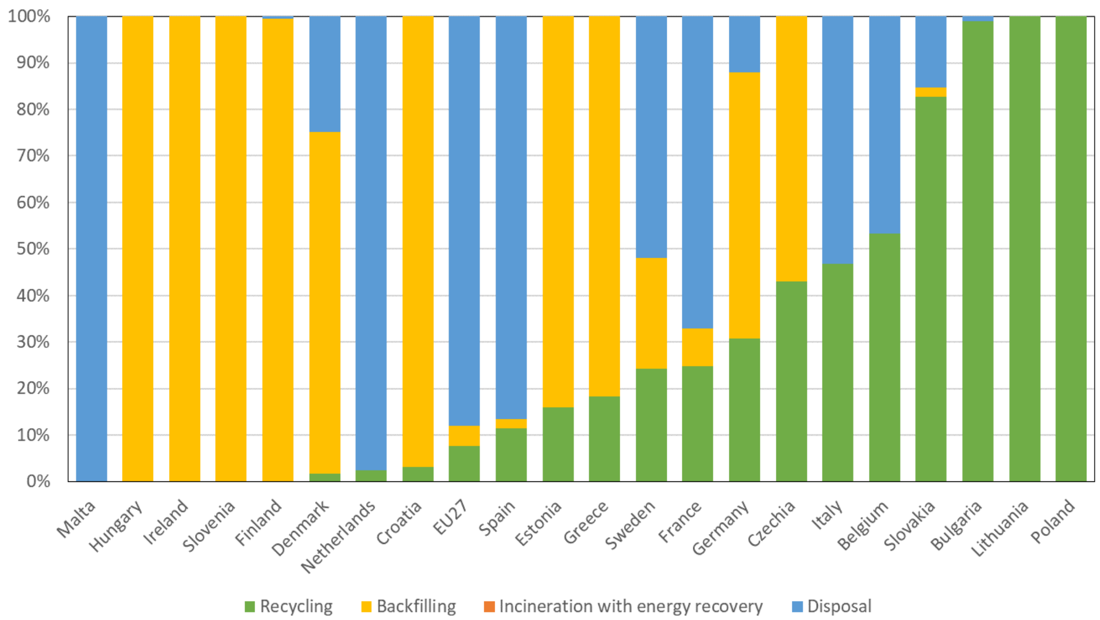


Source: Own elaboration based on data from Eurostat (env_wastrt).

4.4 Dredging spoils

Similarly, for dredging spoil, we refer to the treatment data available in Eurostat (env_wastrt; European Commission, 2021c). Figure 6 shows the treatment data for this fraction of (non-hazardous) CDW in the Member States in 2020. According to the data, there are three countries that recycle 100% of dredging spoils (Bulgaria, Lithuania, and Poland), while Slovakia recycles more than 80%. However, the EU average shows that recycling accounts only for 8%, disposal for 88%, and backfilling for 4%. This is due to the weight of Netherlands where 85% of dredging spoils within the EU are generated and the treatment followed within this country is mainly disposal (98%). The limiting factors referenced in Box 1 might also apply to the case of dredging spoils. Note that six countries reported no data for this fraction (i.e. Austria, Cyprus, Latvia, Luxembourg, Portugal and Romania). The validation report of the Eurostat data (Noel et al., 2021) for 2018 points out that, in the case of the Netherlands, for some disposal methods such as disposal of dredging spoils at sea (code D6), treatment capacities are less straightforward to be determined and thus they have not been reported.

Figure 6. Treatment of the dredging spoil fraction of CDW in the Member States in 2020. Disposal is defined as landfill, incineration without energy recovery and other disposal. Note that data for Ireland and Czechia 2020 is not available and data from 2018 has been used instead.



Source: Own elaboration based on data from Eurostat (env_wastrt).

4.5 Overview of current CDW treatment pathways and recovery rates in the EU

Based on the details provided per each individual material fraction along the whole Section 4, Table 9 summarises the current treatment pathways in the EU. The overall recovery rate for CDW (based on the mineral waste fraction) as reported to Eurostat (env_wastrt) for the attainment of the WFD CDW recovery target is shown in Table 10. It varies from 63% for Finland to 100% for Malta, Greece and the Netherlands, and equals on average 89% for the EU in 2020.

Table 9. Overview of the current treatment pathways for individual material fractions of CDW in the EU based on data reported by Member States and on the available techno-scientific literature. Values are rounded.

Material fraction	Preparing for reuse	Recycling	Backfilling	Incineration	Landfilling
Mineral fraction					
Concrete	0%	79%	10%	0%	11%
Bricks	0%	79%	10%	0%	11%

Ceramics and tiles Insulation material Gypsum	0% 0% 0%	79% 2% 10%	10% 0% 0%	0% 0% 0%	11% 98% 90%
Metals Aluminium Steel	 10% 10%	 84% 84%	 0% 0%	 0% 0%	 6% 6%
Plastic PVC EPS	 0% 0%	 30% 10%	 0% 0%	 12% 66%	 58% 24%
Wood	0%	30%	0%	64%	6%
Glass	0%	6%(1)	24%(2)	0%	70%
Soil waste	0%	35%	40%	0%	25%
Dredging spoils	0%	8%	4%	0%	88%

(1) *6% only is estimated to be glass to glass recycling (https://ec.europa.eu/environment/pdf/waste/studies/CDW_Final_Report.pdf).*
(2) This is an assumption due to lack of specific data as the RA containing glass could be used for road construction and reported as recycled.

Source: Own elaboration – see detailed description and references in Sections 4.1 to 4.4.

Table 10. Overall recovery rate for CDW across the EU Member States (in %); na: not available.

	2010	2012	2014	2016	2018	2020
European Union - 27 countries (from 2020)	na	na	87	87	88	89
Belgium	17	18	32	95	97	99
Bulgaria	62	12	96	90	24	96
Czechia	91	91	90	92	na	96
Denmark	na	91	92	90	97	97
Germany	95	94	na	na	93	94
Estonia	96	96	98	97	95	93
Ireland	97	100	100	96	100	na
Greece	0	0	0	88	97	100
Spain	65	84	70	79	75	73
France	66	66	71	71	73	74
Croatia	2	51	69	76	78	89
Italy	97	97	97	98	98	98
Cyprus	0	60	38	57	64	79

Latvia	na	na	92	98	97	99
Lithuania	73	88	92	97	99	98
Luxembourg	98	99	98	100	98	99
Hungary	61	75	86	99	99	98
Malta	16	100	100	100	100	100
Netherlands	100	100	100	100	100	100
Austria	92	92	94	88	90	91
Poland	93	92	96	91	84	74
Portugal	58	84	95	97	93	95
Romania	47	67	65	85	74	88
Slovenia	94	92	98	98	98	97
Slovakia	na	na	54	54	51	81
Finland	5	12	83	87	74	63
Sweden	78	81	55	61	90	74

Source: (Eurostat; env_wastrt; CEI, "Circular Economy Indicator WM_040"; accessed May 2023).

5. Assessment of environmental and economic impacts of CDW management

This section presents materials and methods to conduct an environmental and economic impact assessment of CDW management, as well as the results derived from it. The methodology subsection includes all the elements needed to properly perform the assessment, such as the definition of goal and scope, functional unit, system boundaries, selection of impact categories and scenarios to be assessed, inventory, and finally uncertainty handling.

5.1 Material and methods

5.1.1 Goal, scope and functional unit

Life Cycle Assessment (LCA) methodology is applied in accordance with ISO 14040/14044 standards. Complementarily, LCC methodology presented in Martinez-Sanchez et al. (2015) is employed to perform the Environmental Life Cycle Costing (ELCC) and the Societal LCC (SLCC). The former consists in a financial assessment including relevant environmental taxes (e.g. landfill and incineration taxes) and the latter is also known as "welfare-economic" assessment that includes the costs of marketed goods along with the effects on the welfare of the society caused by externalities. Note that here we only include external cost of the environmental emissions to air, water, and soil[19]. The goal of the study is to assess the environmental and economic impacts of alternative management options for the different fractions of CDW, focusing on preparing for reuse and recycling. To this aim, the functional unit (FU) is the management of one tonne of each individual fraction from CDW. Based on this FU, for each individual waste material fraction we compare alternative waste management pathways; these management and technology pathways do not necessarily produce the same end-products. This is evident for the case of recycling and landfill or incineration pathways. To credit the reuse or recovery of materials and energy via the alternative EoL management pathways, system expansion is applied as commonly done in waste management LCAs (Ekvall, 2002) and in accordance with ISO 14040/14044 and the Environmental Footprint methodology. In line therewith, to quantify the impacts related to the use of a reused product or material (i.e. multi-cycle product or material), the approach put forward in the European standard EN 15804+A2, which sets the rules for the product category of construction products, includes an additional phase (module D – benefits and loads beyond the system boundaries) that can be used to assess and express the potential benefits and loads of future loops. This method assumes that the savings from reusing a material are equal to the impact related to the production of the original material (assumed herein coming from virgin sources) minus the impact of the processes required to prepare the product for reuse (e.g. cleaning) (Etienne et al., 2022). This method builds on the assumption that benefits associated to the use of a reused product are more important than benefits associated to the use of a (potentially) reusable product (i.e. avoiding the production related impact is more relevant than limiting the EoL impact by using a reusable product.). It is necessary to keep in mind that the reusability of a product is hypothetical and cannot be guaranteed when using a product (the benefit is "potential"), while for a reused product, the benefit is certain. Note that system expansion only assumes one cycle of reuse or recycling; any further cycles are neglected (Malabi Eberhardt et al., 2020). Also, it does not account for reference service life, maximum number of reuses, or the impact allocation from the final disposal of the material. Other methods are available in the literature as shown by Allacker et al. (2017) and De Wolf et al. (2020), but no consensus exists.

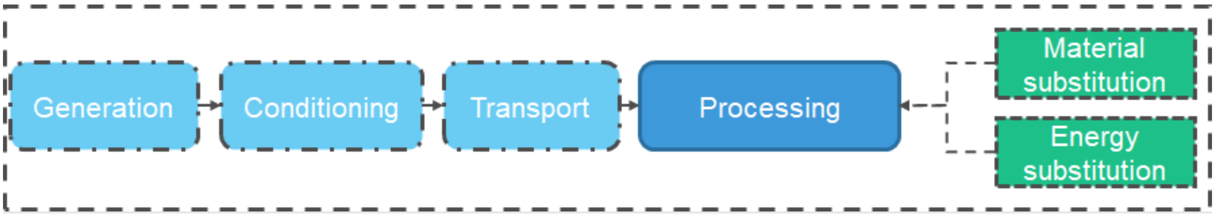
5.1.2 System boundaries

The general system boundaries of the study are shown in Figure 7. Light-blue dash-dotted squares denote life cycle stages that, depending on the scenario, might not be needed, dark-blue solid squares are life cycle stages that occur in all scenarios and green dotted squares represent substitution processes which might occur depending on the final output produced. The generation stage includes demolition processes that can be either conventional or selective demolition, excavation processes, or dredging processes. The conditioning stage includes all processes needed to adequate the wastes generated before transport or processing (e.g. dewatering

[19] The authors are aware that recent projects and literature suggest a different naming for this assessment (full Environmental Life Cycle Costing; fELCC) as only environmental emissions are fully monetised. However, given the lack of consensus on the terminology, we prefer to maintain the naming of SLCC in this report, as this has also been used in a parallel publication (Caro et al., 2024) and throughout the project workshops and consultations.

processes for dredging spoils). The transport stage includes the handling of material as well as the transport of input-waste from the generation site to the receiving site or treatment facilities comprising different transport vessels such as lorry, train or barge. As the impact of transport is highly dependent on the distance and, depending on the scenario, might not be needed (e.g. on-site treatment of wastes), it will be subject to a dedicated sensitivity analysis. The processing stage includes the preparing for reuse, recycling and recovery operations (depending on the scenario) needed to transform the waste material into a reusable product or a recyclate and emissions. This stage also includes the landfilling process. Consecutive (subsequent or multiple) use of the reused materials and recyclates has not been taken into consideration in this study, except for wood which is analysed in a dedicated sensitivity analysis (see Section 5.1.7). The processing stage also considers further treatment of any non-targeted material fractions separated and recovered during recycling (e.g. metals, when included in the input-waste), as well as the handling of separated non-recyclable material fractions, residues and losses from recycling. Residues and losses from recycling were assumed to be landfilled. The input-waste was assumed to carry no environmental burdens from the respective upstream life cycle stages (that according to EN 15978 would be from A1 to B7, covering the product stage, the construction process stage and the use stage). This follows the common "zero-burden" assumption applied in waste management LCA, as the impact of production is the same across the alternative management scenarios. The assessment was conducted by employing the EASETECH software (Clavreul et al., 2014).

Figure 7. General system boundaries of the Life Cycle Assessment and Life Cycle Costing. Waste is assumed to be generated burden-free as the upstream impacts of the production would be the same across all management scenarios. All operations involving generation are included (e.g. demolition) (blue boxes) but depending on the scenario some might not be needed (blue dashed boxes). System expansion is applied to credit material and energy recovery (green dashed boxes).



Source: Own elaboration.

5.1.3 Impact categories

The impact assessment was performed for the 16 impact categories included in the Product Environmental Footprint method (Zampori & Pant, 2019). Herein, only Climate Change results are presented and discussed. The results for the remaining environmental impact categories are summarised in Annex 1 and further reported in Caro et al. (2024) and Cristóbal et al. (forthcoming). Moreover, economic impacts were assessed through ELCC and SLCC. The ELCC accounts for internal costs (annualised cost of capital along with operational expenditures and revenues) and internalised environmental taxes (i.e. already paid by companies). The SLCC includes internal costs (expressed as shadow prices, i.e. removing taxes and subsidies) summed up to external costs (also expressed as shadow prices; i.e. monetised environmental emissions to air, water, soil).

5.1.4 Scenario definition

The assessment considered five classes of waste management options possible for the different fractions: preparing for reuse (REU), recycling (REC), recovery-backfilling (RCB), landfilling (LAN), and incineration (INC). Note that for some fractions such as excavated soils and dredging spoils, backfilling is similar to preparing for reuse and recycling, respectively, in terms of modelling, because both pathways substitute non-waste materials (e.g. natural aggregates). The selected scenarios relying on alternative pathways and technologies for treatment of individual CDW material fractions are shown in Table 11. Landfill was considered for all CDW fractions, and incineration was considered only for those fractions having positive calorific value. Table 11 also presents the substitutability factor applied to account for the replacement of primary material via secondary material (reused or recycled) in the market. Sustainability factors are taken from literature and consider both the quality of the material and the market demand (Borghi et al., 2018). The quality of the material is considered in terms of “clean composition” and presence of impurities (through a coefficient named Q_1), as well as technical characteristics compared to those of the substituted material in relation to the specific application (through a coefficient named Q_2). In contrast, the market demand captures the ratio of the amount of material sold and of the amount produced at the preparing for reuse/recycling/recovery process in a specific time period (through a coefficient named M). Thus, the substitutability factor is calculated by multiplying the three coefficients (i.e. Q_1* Q_2*M).

Table 11. Description of the scenarios modelled in this study for the different fractions with details on life cycle stages and product substitutability factors (where applicable).

Fraction	Details of life cycle stage					Scenario Code(1)
	Generation	Processing	Products & Outputs	Products substituted	Substitutability factor	
Concrete (CON)	Selective demolition	Preparing for reuse	Concrete	Concrete	1	REU
	Selective demolition	Recycling	Cementitious material Recycled aggregates	Cement Natural aggregates	0.71 0.85	REC-CEM
	Selective demolition	Recycling	Recycled aggregates	Natural aggregates	0.85	REC-RA
	Conventional demolition	Landfill	-	-	-	LAN
Wood (WOD)	Selective demolition	Preparing for reuse	Wood	Timber	1	REU
	Selective demolition	Recycling	Particle board	Particle board	1	REC-PBD
	Conventional demolition	Landfill	-	-	-	LAN
	Conventional demolition	Incineration	Electricity & heat	Electricity & heat	1	INC
Steel (STE)	Selective demolition	Preparing for reuse	Steel	Steel	1	REU
	Selective demolition	Recycling	Iron scrap	Iron scrap	0.75	REC-STE
	Conventional demolition(2)	Landfill	-	-	-	LAN
Aluminium (ALU)	Selective demolition	Preparing for reuse	Aluminium	Aluminium	1	REU
	Selective demolition	Recycling	Aluminium ingot	Aluminium ingot	0.85	REC-ALU
	Conventional demolition(2)	Landfill	-	-	-	LAN
Plastic PVC (PVC)	Selective demolition	Recycling	Polyvinylchloride	Polyvinylchloride	0.69	REC-PVC
	Conventional demolition	Landfill	-	-	-	LAN

	Conventional demolition	Incineration	Electricity & heat	Electricity & heat	1	INC
Plastic EPS (EPS)	Selective demolition	Recycling	Polystyrene	Polystyrene	0.69	REC-EPS
	Conventional demolition	Landfill	-	-	-	LAN
	Conventional demolition	Incineration	Electricity & heat	Electricity & heat	1	INC
Gypsum (GYP)	Conventional demolition	Recycling	Plasterboard	Plasterboard	0.88	REC-GYP
	Conventional demolition	Landfill	-	-	-	LAN
Ceramics & tiles (C&T)	Selective demolition	Preparing for reuse	Ceramic	Ceramic	1	REU
	Selective demolition	Recycling	Cementitious material	Cement	0.71	REC-CEM
	Selective demolition	Recycling	Recycled aggregates	Natural aggregates	0.83	REC-RA
	Conventional demolition	Landfill	-	-	-	LAN
Glass wool (GLW)	Selective demolition	Recycling	Glass wool fibres	Virgin rock	0.83	REC-GLW
	Conventional demolition	Landfill	-	-	-	LAN
Stone wool (STW)	Selective demolition	Recycling	Stone wool fibres	Virgin rock	0.83	REC-STW
	Conventional demolition	Landfill	-	-	-	LAN
Bricks (BRK)	Selective demolition	Preparing for reuse	Brick	Brick	1	REU
	Selective demolition	Recycling	Cementitious material	Cement	0.71	REC-CEM
	Selective demolition	Recycling	Alkali activated blocks	Concrete	0.65	REC-CON
	Selective demolition	Recycling	Recycled aggregates	Natural aggregates	0.83	REC-RA
	Conventional demolition	Landfill	-	-	-	LAN
	Selective demolition	Preparing for reuse	Glass	Glass	1	REU

Glass (GLA)	Selective demolition	Recycling	Flat glass	Flat glass	1	REC-GLA
	Selective demolition	Recycling	Recycled aggregates	Natural aggregates	0.83	REC-RA
	Conventional demolition	Landfill	-	-	-	LAN
Excavated soil & rocks (ESR)	Excavation	Preparing for reuse	Soil	Natural aggregates	0.65	REU-RCB(3)
	Excavation	Recovery backfill				
	Excavation	Recycling – stabilisation (with lime)	Stabilised soil	Concrete	1	REC-LIM
	Excavation	Recycling – stabilisation (with cement)	Stabilised soil	Concrete	1	REC-CEM
	Excavation	Recycling	Individual components (sand, clay)	Natural sand Natural clay	1	REC-IND
	Excavation	Landfill	-	-	-	LAN
Dredging spoil (DDS)	Dredging	Preparing for reuse	Dredged sediments	Natural aggregates	1	REU
	Dredging	Recycling (use on land)	Dredged sediments	Natural aggregates	0.75	REC-RCB(3)
	Dredging	Recovery backfill				
	Dredging	Recycling – stabilisation (with cement)	Stabilised sediments	Concrete	1	REC-CEM
	Dredging	Recycling	Individual components (sand, clay)	Natural sand Natural clay	1	REC-IND
	Dredging	Landfill (upland)	-	-	-	LAN

(1) The code 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) or the material used as binder for the stabilisation technique for excavated soils and dredging spoils (LIM=lime; CEM=cement).

(2) While the landfilling scenario for metals has been modelled for illustration in the present study, it does not reflect today's reality, as metals are mostly recycled.

(3) Complying with the recovery definition in the WFD, the backfilling operation must replace other materials that are not waste. In this sense, for excavated soils, backfilling is modelled equally as preparing for reuse (Iacovidou et al., 2020), and for dredging spoils it is modelled as recycling (use on land).

Note: Transport is included in all scenarios (road transport) while conditioning (dewatering) is only considered for dredging spoils.

Source: Own elaboration.

5.1.5 Inventory

To describe the foreground system, we use the data obtained from the techno-scientific literature on i) waste characterisation (CDW composition and flows), and ii) technologies and processes inventory (energy, electricity, material, fuels and resource provision). For more information about the data used for each scenario listed in Table 11, the reader is referred to Caro et al. (2024) and Cristóbal et al. (forthcoming). Complementary background data for modelling waste treatment technologies were taken from the Ecoinvent database 3.7.1 (Ecoinvent, 2023).

Transport distances from the generation site to preparing for reuse/recovering/recycling sites or landfills were assumed to be 50 km (Magnusson et al., 2015; N. Zhang et al., 2020). For other transport processes, for example the transport of soil components (e.g. sand) after recycling, a lower distance of 20 km was considered following the assumptions of Magnusson et al. (2015). For transport of materials required for management options, e.g. lime or cement for stabilisation, 10 km was used. While shorter or longer distances may occur with different treatment options across the 27 Member States, the same distance was assumed for all scenarios to capture the differences in the performance of individual management technologies, rather than focusing on specific regional contexts. However, these assumptions on the distances are tested in a sensitivity analysis. Note that we considered the volume-limiting factor for transport of voluminous and light material such as EPS using the utilisation rate of the cargo, following the approach explained in Lu et al. (2021) (considering bulk density of the material and weight and volume of the cargo). We assumed that conventional demolition preceded incineration, landfill or any recovery operation, while selective demolition was a prerequisite for recycling and preparing for reuse. The modelling of these two types of demolition involved different energy consumptions and costs (Caro et al., 2024).

For all scenarios, one or more reference studies were used to model the technologies (see Table 8). Detailed (life cycle) inventories for technologies and processes may be found in separate documents complementing this study (Caro et al., 2024; Cristóbal et al., forthcoming).

5.1.6 Uncertainty propagation analysis

Parameter uncertainty was addressed using uncertainty propagation, following the approach suggested in Bisinella et al. (2016). 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 the 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. The first is assigned to all parameters, mainly following a uniform distribution and the range assigned is either based on literature, when available, or assumed to be +/- 20%, when not available. The additional uncertainty on 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. For more information about the data used, the reader is referred to Caro et al. (2024) and Cristóbal et al. (forthcoming).

5.1.7 Sensitivity analyses

On top of the uncertainty propagation analysis, we perform four sensitivity analyses focusing on different groups of parameters, by altering key parameters *one-at-a-time* to see their effect on the results. The first group of parameters is related to transport. Since the key parameter in this case is the distance, three different assumptions are made. The first one is the base case in which the distance to treatment is assumed to be 50 km. The second is treating the waste in-situ (i.e. transport distance null, whenever possible). Note that this in-situ treatment assumes that the treatment plant functions with diesel instead of electricity, and in some cases the generation process is not needed, such as for excavated soils for which the stabilisation technique can be used without excavation. The third is an increase in the distance of the transport to subsequent treatment/disposal set to 100 km (double the base case). Transport distances for all remaining products (e.g. used for processing) are maintained as in the base case. The second group of parameters tested is related to the substitutability factors in the model. These have been halved relative to the value used in the base case (see Table 10). The third group is on the use of a low-carbon energy mix in line with the upcoming policy framework (i.e. year 2050) as reported by Keramidas et al. (2021). Finally, a specific sensitivity analysis is performed for the wood waste fraction introducing the cascading principle to identify the effects on the results. This sensitivity is performed following the approach of Faraca, Tonini, et al. (2019) that studied the cascade of wood waste. For this analysis, we assume that only high-quality wood waste (Q1 and Q2 of the

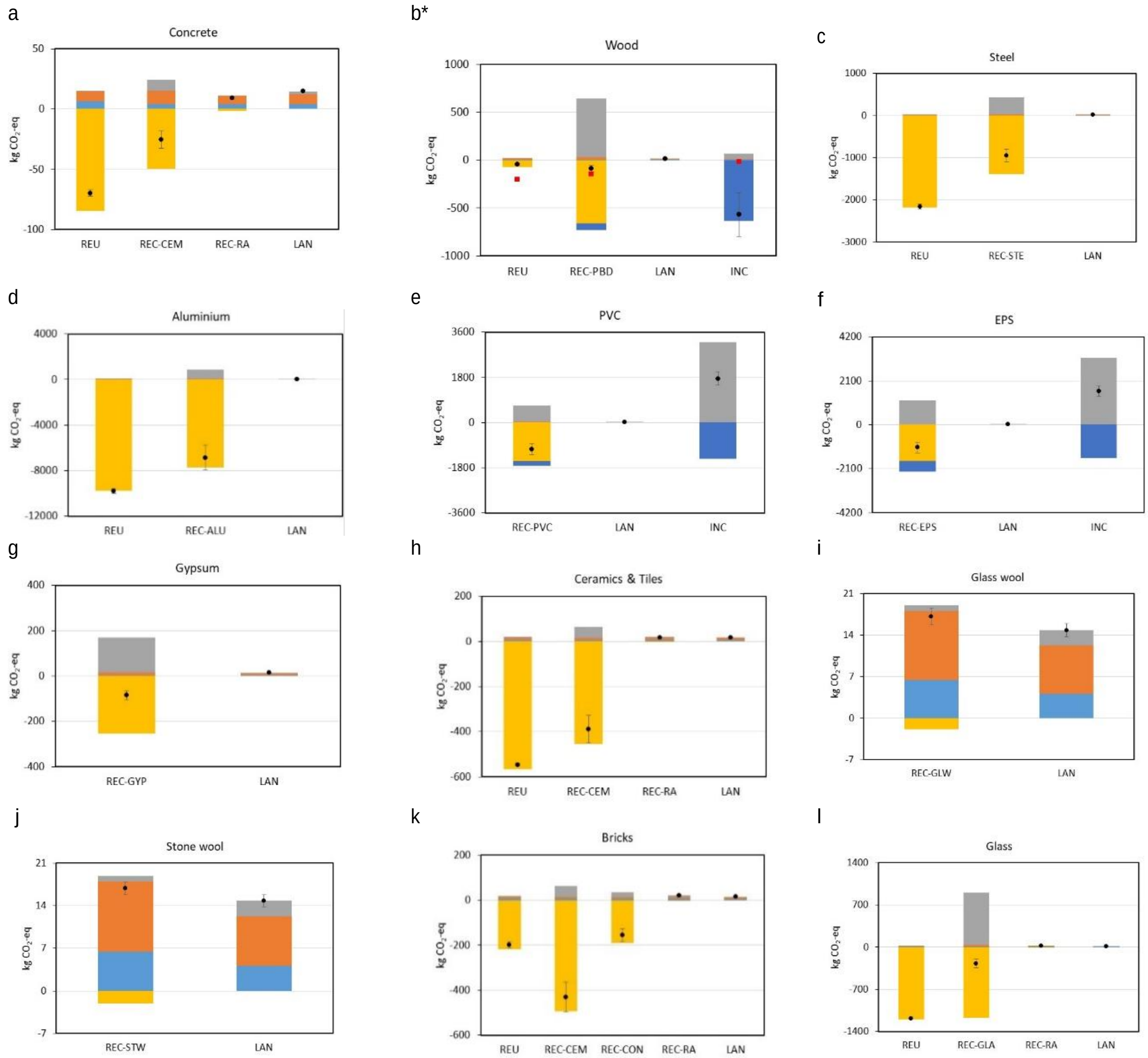
German classification, see Faraca, Tonini, et al. (2019)) can be used for subsequent cascading applications in particleboard production. This constitutes ca. 60% of the wood waste generated at each cycle, the rest being sent to incineration for energy recovery. We do not consider impacts from the use phase and transport in the subsequent life cycles (2^{nd}, 3^{rd}, 4^{th}), as we are only interested in the net difference between the incineration and recycling scenario (we assume that both scenarios have same transport and use impacts). We stop at the 4^{th} life cycle, as in Faraca, Tonini, et al. (2019).

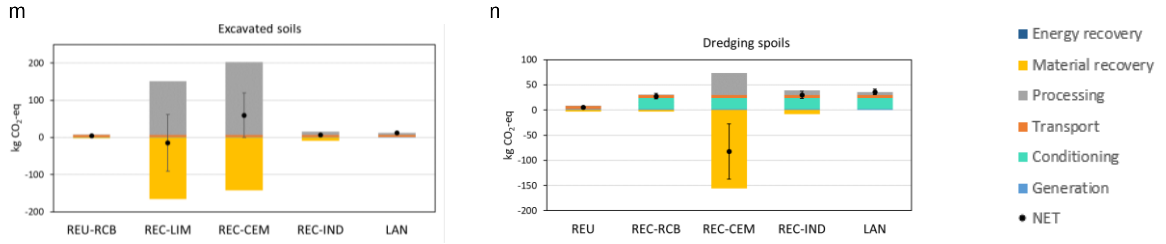
5.2 Assessment results

5.2.1 Life Cycle Assessment results

Figure 8 shows the results for the category Climate Change for all scenarios analysed. The contribution breakdown presents the following aggregation: i) waste generation (including demolition, excavation, dredging); ii) conditioning processes; iii) transport; iv) processing; v) material recovery; vi) energy recovery.

Figure 8. Characterised Climate Change results per tonne of CDW fraction managed with breakdown of the contributions in 2020. Values above zero represent burdens, while values below zero represent savings. The final net impact, per each individual category, is the sum of burdens and savings and is represented with a black dot. The error bars represent the standard deviation around the net result. For the abbreviations used, please refer to Table 11.





*For wood, the red square (■) indicates the result when cascading is accounted for (four cycles; sensitivity analysis).

Source: Own elaboration.

On Climate Change, the waste hierarchy is fully respected for nine CDW material fractions, and partially respected for the three fractions that present the incineration option (i.e. PVC and EPS, where landfilling performs better than incineration, and wood, where incineration performs better than reuse and recycling (only if the cascading is not considered, because otherwise the hierarchy is fully respected). The two fractions that do not follow the waste hierarchy are glass wool and stone wool, where landfilling performs better than recycling (see the paragraph below). Thus, preparing for reuse is the best performing option for almost all fractions that present the option, capitalising on the low processing impacts and on the high savings from material recovery (e.g. 9 770 kg CO_2 eq. t^{-1} and 2 166 kg CO_2 eq. t^{-1} for aluminium and steel, respectively). Only for bricks and dredging spoils, a recycling option performs better than preparing for reuse, with the latter being the second best performing option.

For wood (Figure 8b), incineration performs better than preparing for reuse and recycling, owing to high energy recovery and low GHG emissions because of the carbon neutrality assumption for biogenic carbon in wood. However, Figure 8b shows that when the calculation for cascading is performed, wood results significantly change with preparing for reuse becoming the best option followed by recycling and finally incineration. Although results showed in Figure 8b are calculated by using four cycles of cascading, an inversion of the trend is already observed with only two cycles (see Annex 2 for further info). For stone wool (Figure 8j) and glass wool (Figure 8i), the recycling scenarios offer limited GHG savings. This result can be extended to the other material fractions where scenarios producing RA are considered. We observe that recycling to RA is comparable to landfilling, i.e. credits from material recovery are not sufficient to compensate for the burdens of collection, sorting, transport and recycling operations. For instance, recycling of concrete waste to cement (Figure 8a) records a total net GHG saving of 26 kg CO_2 eq. t^{-1} which is substantially higher than the net burden obtained when recycling concrete to RA (9 kg CO_2 eq. t^{-1}). Another example, the closed-loop recycling producing flat glass from glass waste achieves a total net saving of 272 kg CO_2 eq. t^{-1} (Figure 8l), again, considerably higher than the net burden obtained when recycling glass to RA (23 kg CO_2 eq. t^{-1}). The results are similar for ceramics/tiles (Figure 8h) and bricks (Figure 8k), where recycling to RA performs better than landfill and incineration, but results in limited savings relative to them.

The highest GHG savings are achieved for preparing for reuse and recycling metals (Figure 8c for steel and Figure 8d for aluminium). While this is expected, due to the carbon-intensive production process of these materials and the consequent substantial savings in recycling them, aluminium and steel are currently already separated and recycled to a large extent, owing to their market value. Recycling of EPS (Figure 8f) and PVC (Figure 8e) save 1 088 and 1 058 kg CO_2 eq. t^{-1}, respectively, resulting in a significant reduction of GHG emissions relative to the net burden incurred by landfilling (15 kg CO_2 eq. t^{-1}) and incineration (1 605 and 1 747 kg CO_2 eq. t^{-1}, respectively). Recycling gypsum to plasterboard (saving 85 kg CO_2 eq. t^{-1}) performs better than landfilling (net burden of 15 kg CO_2 eq. t^{-1}; Figure 8g). Recycling bricks or concrete to cement also generates important net GHG savings (431 kg CO_2 eq. t^{-1} and 26 kg CO_2 eq. t^{-1}, respectively) relative to the net burden from landfilling (15 kg CO_2 eq. t^{-1}). However, these scenarios are based on experimental set ups with low Technology Readiness Level and are thus highly uncertain (i.e. ±66.4 kg CO_2 eq. t^{-1} and ±7.3 kg CO_2 eq. t^{-1}, respectively).

Across all scenarios investigated, the most important contribution to climate burdens from recycling is the recycling process itself, while the most notable contribution to the savings is the substitution of materials with a substantial difference between substituting natural aggregates (low savings) and substituting cement or materials in a closed loop (high savings). However, in scenarios with recycling to RA 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 impacting parameter in CDW management. In selected recycling pathways, the substitution of energy through energy recovery also becomes important when CDW is diverted to incineration. This is the case for fractions with high calorific value, i.e. wood, EPS and PVC. For PVC and EPS, however, notwithstanding the energy recovery savings, the overall GHG balance is a net burden (GHG emissions are higher than GHG savings). For incineration, the most important contribution to the burdens is the process itself (combustion and related emissions); the same holds for landfilling (on-site operations). The GHG contribution of demolition is negligible compared to the other process stages.

With respect to excavated soils (Figure 8m), the scenario leading to the highest climate benefits is recycling through stabilisation with lime. This was the only one contributing with net savings (14 kg CO_2 eq. t^{-1}), followed by preparing for reuse/backfilling (burden of 4 kg CO_2 eq. t^{-1}), recycling (burden of 6 kg CO_2 eq. t^{-1}), landfilling (burden of 12 kg CO_2 eq. t^{-1}), and finally recycling through stabilisation with cement (burden of 60 kg CO_2 eq. t^{-1}), all of them net burdens. It is important to note that both recycling through stabilisation scenarios present high uncertainty (±76 kg CO_2 eq. t^{-1} for stabilisation with lime and ±60 kg CO_2 eq. t^{-1} for stabilisation with cement), mainly due to the excavated soil density parameter which influences the quantities of binder added and thus the whole modelling. Across all scenarios, processing (including landfilling) contributes the most to the net impact except for reuse/backfilling where the main contribution is transport. Specifically, the burdens from recycling through stabilisation were driven by the carbon dioxide in the production of hydrated lime and the cement added as a binder. In contrast, the main contribution to GHG savings was material recovery, due to the displacement of natural aggregates, and concrete for reuse/backfilling and stabilisation with lime scenarios, respectively. The contributions from transport were significant for reuse/backfilling and recycling. For the remaining scenarios, albeit not negligible, they were minor relative to the other processes. Following the same tendency, contributions from excavation are small compared to the other processes.

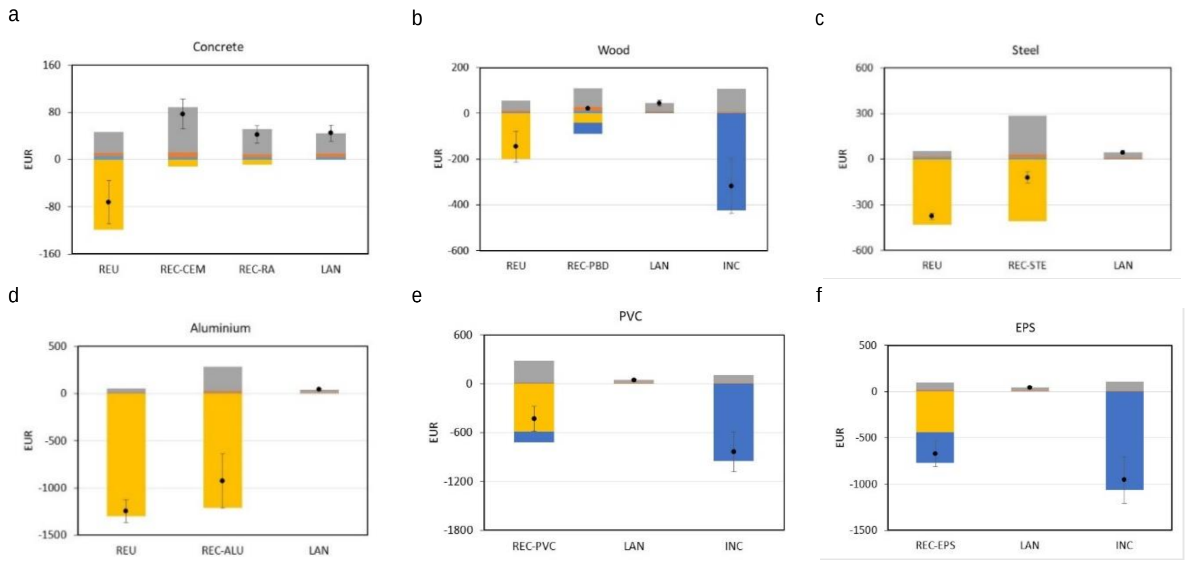
For dredging spoils (Figure 8n), almost all scenarios lead to climate burdens, except for recycling through stabilisation with cement, yielding savings of 82 kg CO_2 eq. t^{-1}, followed by preparing for reuse (burden 5 kg CO_2 eq. t^{-1}), backfilling (burden 27 kg CO_2 eq. t^{-1}), recycling (burden 30 kg CO_2 eq. t^{-1}) and landfilling (burden 36 kg CO_2 eq. t^{-1}). The main contributions are from conditioning except for the case of stabilisation with cement, where processing also plays an important role. The contribution of transport and generation processes are small relative to the others.

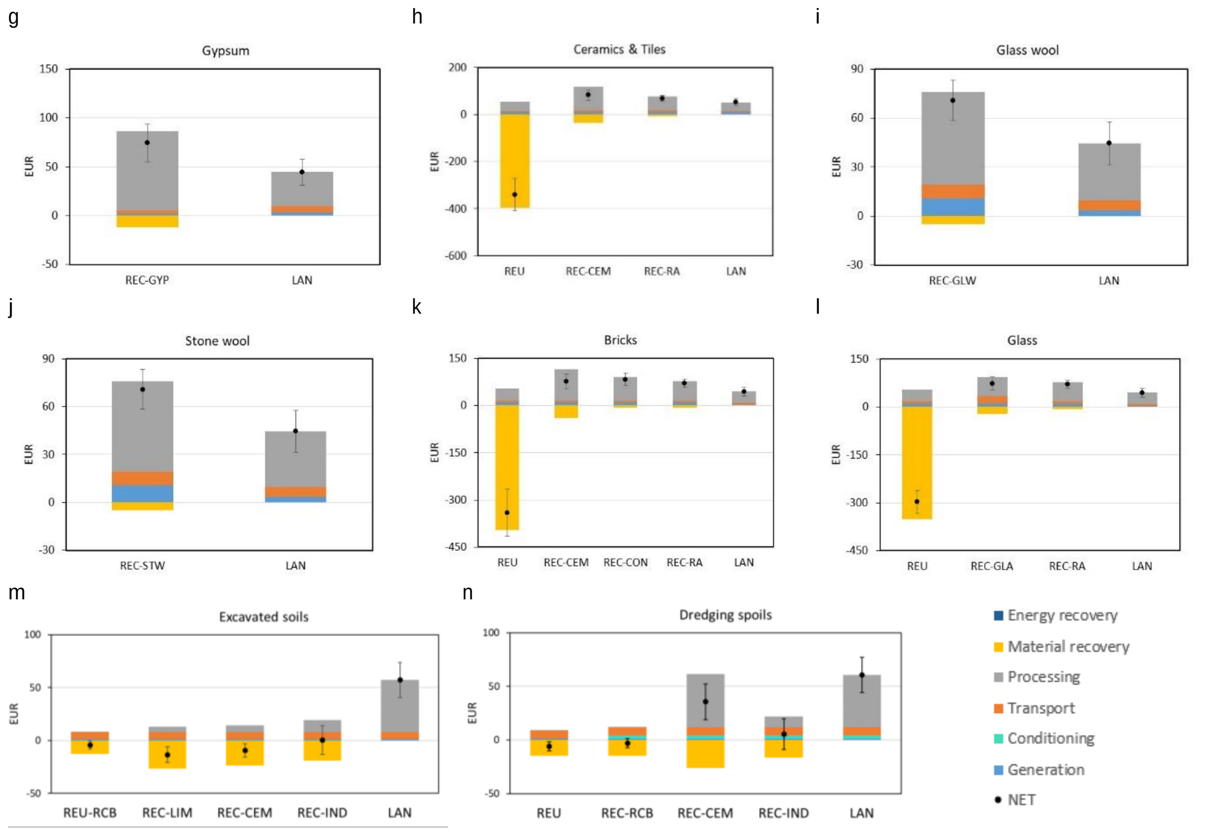
Detailed results for the other key impact categories analysed on top of Climate Change can be found in Annex 1. The impacts on the other categories, except for ozone depletion, land use and resource use, follow a similar trend to that of Climate Change with respect to the ranking of the scenarios and impact contributions.

5.2.2 Environmental Life Cycle Costing results

Figure 9 shows the Environmental Life Cycle Costing results for the scenarios analysed. The contribution breakdown is the same as for the LCA results: i) waste generation (including demolition, excavation, dredging); ii) conditioning processes; iii) transport; iv) processing; v) material recovery; vi) energy recovery.

Figure 9. Environmental Life Cycle Costing results per tonne of CDW fraction managed with breakdown of the contributions. Values above zero represent costs, while values below zero represent revenues. The final net impact, per each individual category, is the sum of costs and revenues and is represented with a black dot. The error bars represent the standard deviation around the net result. For the abbreviations used please refer to Table 11.





Source: Own elaboration.

Preparing for reuse is the best option for almost all fractions that present this option, capitalising on the low processing costs and/or on the high revenues from material recovery (e.g. EUR 1 246 t^{-1} and EUR 376 t^{-1} for aluminium and steel, respectively). Only for excavated soils, the recycling through stabilisation scenarios perform better than preparing for reuse, the latter being the second-best performing option.

We find that recycling of concrete (Figure 9a), ceramics/tiles (Figure 9h) and bricks (Figure 9k) to cement and RA is more expensive than landfilling (note that a landfill tax of EUR 19 t^{-1} is included). The same holds for the cost of closed-loop recycling of gypsum to plasterboard and glass to flat glass compared to landfilling (Figure 9g and Figure 9l, respectively). Recycling of steel (Figure 9c) and aluminium (Figure 9d) expectedly stands out as clearly economically favourable. For plastics, recycling of PVC (net income of EUR 431 t^{-1}) is less costly than landfilling (EUR 45 t^{-1}) but more expensive than incineration (net income of EUR 839 t^{-1}). Although an incineration tax is included (EUR 21 t^{-1}), this result is mainly due to the higher revenues obtained from energy recovery via incineration (Figure 9e). Recycling of EPS (net income of EUR 674 t^{-1}) is economically favourable to landfilling (EUR 45 t^{-1}), but still more costly than incineration (net income of EUR 956 t^{-1}; note that a part of the revenues of recycling comes from energy recovery, because of the significant amount of recycling residues incinerated) (Figure 9f). Recycling of stone wool (Figure 9j) and glass wool (Figure 9i) is more expensive than landfilling. Across all scenarios investigated, the most important contribution to the costs is the recycling process itself (processing). Similarly, for landfilling and incineration the process is the most expensive stage. The revenues stem from the sales of materials and energy.

The scenarios with the lowest costs for excavated soils (Figure 9m) are recycling through stabilisation with lime (net revenues of EUR 14 t^{-1}) and with cement (net revenues of EUR 9 t^{-1}), followed by preparing for reuse/backfilling with net revenues of EUR 4 t^{-1}. The recycling scenario presents a net cost of EUR 0.5 t^{-1} (with high uncertainty, i.e. EUR ±13 t^{-1}), and for the landfilling scenario the net cost equals EUR 57 t^{-1}. The main contribution to the landfilling scenario is the processing (accounting for CAPEX and OPEX), as well as the landfill tax mentioned before. For the remaining scenarios, processing constitutes a lower share of the total costs, being almost negligible in preparing for reuse/backfilling. Transport processes represent a high contribution in the preparing for reuse/backfilling scenario and a moderate contribution in the rest of them. Contributions from excavation are very low in all scenarios. Finally, revenues are obtained from material recovery.

For dredging spoils (Figure 9n), the lowest cost are obtained for preparing for reuse with a net income of EUR 6 t^{-1}, followed by recycling for use on land/backfilling with a net income of EUR 3 t^{-1}, recycling with net costs of EUR 5 t^{-1} (with very high uncertainty, i.e. EUR ±14 t^{-1}), recycling through stabilisation with net costs of EUR 36 t^{-1}, and finally landfilling with costs of EUR 61 t^{-1}. In line with the excavated soils, the main contribution to the landfilling scenario is the processing stage, which accounts for the CAPEX and OPEX, as well as the landfill tax. The main contribution for recycling through stabilisation is processing. Meanwhile, material recovery is the main contributor to the total costs of recycling for use on land/backfilling (i.e. REC-RCB), as well as of the recycling scenario (i.e. REC-IND). The summed up contribution of the remaining stages (i.e. generation, conditioning and transport), even if not negligible, is small compared to the contribution of the other processes.

Findings reveal that a trade-off between environmental (Figure 8) and economic (Figure 9) impacts seems to currently exist at EU level, since, aside from the preparing for reuse, the least favourable environmental performance often coincides with the cheapest management pathway. This is the case for concrete, gypsum, ceramic and tiles, bricks, and glass (see Table 12).

Table 12. Combination of results obtained in Figure 8 and Figure 9, reporting the outcome of the net impact on Climate Change (CC) category and the Environmental Life Cycle Costs (Costs), per individual fraction and associated management.

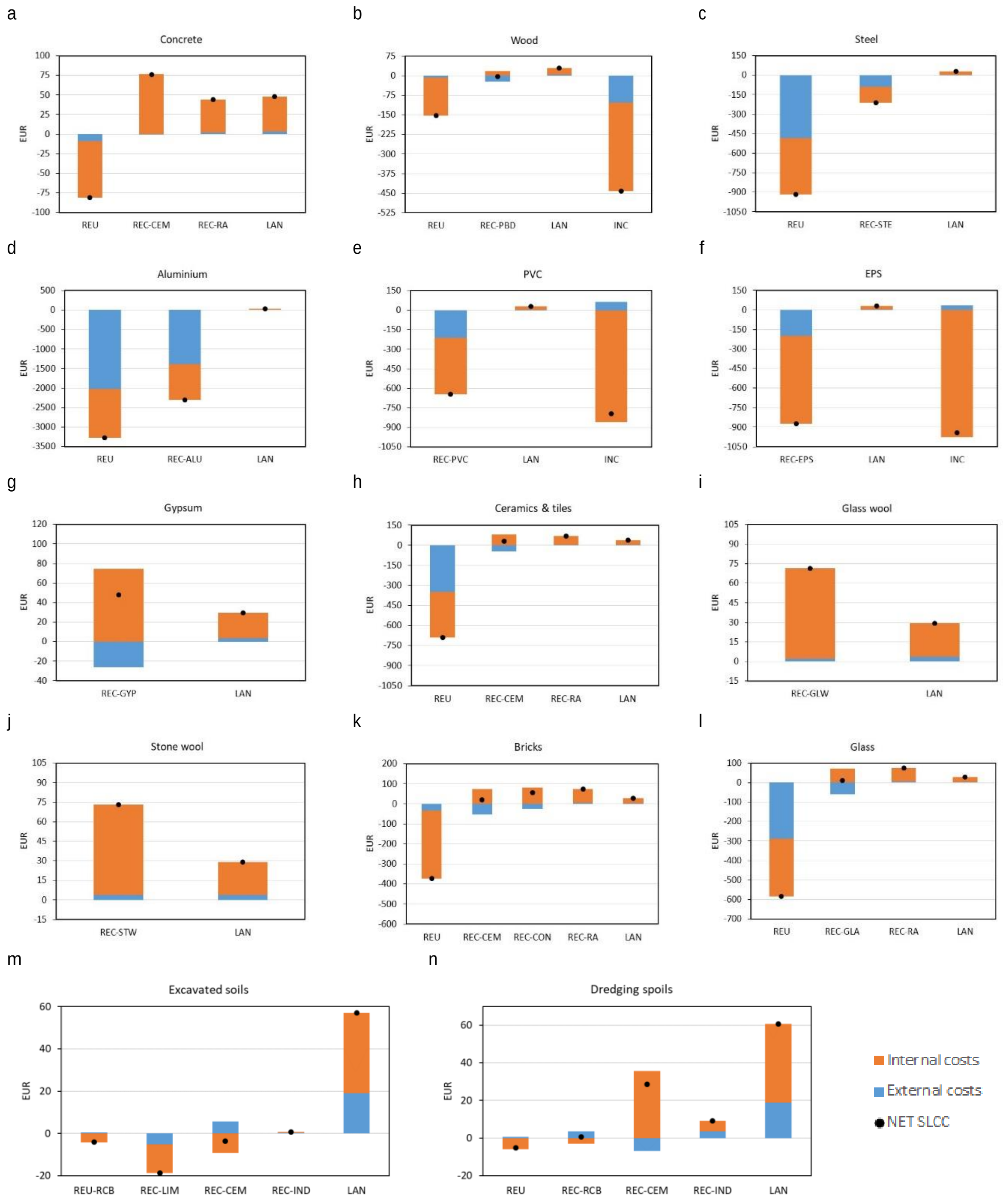
Material	Preparing for reuse		Recycling		Landfill		Incineration	
	CC	Costs	CC	Costs	CC	Costs	CC	Costs
Concrete	(1)☺	(1)☺	(2)😐	(3)☹	(3)☹	(2)😐	-	-
Wood	(2)😐	(2)😐	(3)😐	(3)😐	(4)☹	(4)☹	(1)☺	(1)☺
Wood*	(1)☺	(1)☺	(2)😐	(2)😐	(4)☹	(4)☹	(3)😐	(3)😐
Steel	(1)☺	(1)☺	(2)😐	(2)😐	(3)☹	(3)☹	-	-
Aluminium	(1)☺	(1)☺	(2)😐	(2)😐	(3)☹	(3)☹	-	-
Plastic PVC	-	-	(1)☺	(2)😐	(2)😐	(3)☹	(3)☹	(1)☺
Plastic EPS	-	-	(1)☺	(2)😐	(2)😐	(3)☹	(3)☹	(1)☺
Gypsum	-	-	(1)☺	(2)☹	(2)☹	(1)☺	-	-
Ceramics & Tiles	(1)☺	(1)☺	(2)😐	(3)☹	(3)☹	(2)😐	-	-
Glass wool	-	-	(2)☹	(2)☹	(1)☺	(1)☺	-	-
Stone wool	-	-	(2)☹	(2)☹	(1)☺	(1)☺	-	-
Bricks	(2)😐	(1)☺	(1)☺	(3)☹	(3)☹	(2)😐	-	-
Glass	(1)☺	(1)☺	(2)😐	(2)😐	(3)☹	(3)☹	-	-
Excavated soil	(1)☺	(1)☺	(2)😐	(2)😐	(3)☹	(3)☹	-	-
Dredged spoil	(1)☺	(1)☺	(2)😐	(2)😐	(3)☹	(3)☹	-	-

Note: Symbol ☺ along with (1) represents the best performance from a Climate Change and ELCC perspective. Symbol 😐 along with (2) (and eventually (3)) represents the second-best performance from a Climate Change or ELCC perspective. Finally, symbol ☹ along with (3) (and eventually (4)) represents the worst performance from a Climate Change or ELCC perspective.
**These consider cascading and/or a cleaner energy mix (see sensitivity analysis in Section 5.2.3).*

Source: Own elaboration.

The results for the societal costs (SLCC) are depicted in Figure 10. Overall, the societal cost follows a similar trend to that of the ELCC. Recycling pathways reduce the societal costs compared to landfilling with some exceptions, such as concrete, gypsum, glass wool and stone wool. Preparing for reuse is the best option for almost all fractions that present this option, similarly to the Climate Change impact and ELCC. In terms of SLCC, we find that recycling of concrete (Figure 10a) to cement is more expensive than landfilling; the same holds for the cost of closed-loop recycling of gypsum to plasterboard and glass wool and stone wool compared to landfilling (Figure 10g, Figure 10i and Figure 10j, respectively). Recycling of steel (Figure 10c) and aluminium (Figure 10d) expectedly stand out as economically favourable. For plastics, recycling of PVC (net income of EUR 647 t^{-1}) is less costly than landfilling (EUR 29 t^{-1}), but more expensive than incineration (net income of EUR 794 t^{-1}) (Figure 10e). Recycling of EPS (net income of EUR 874 t^{-1}) is favourable to landfilling (EUR 29 t^{-1}), but still more costly than incineration (net income of EUR 942 t^{-1}) (Figure 10f). Positive external cost for the stabilisation of excavated soils with cement (Figure 10m) is mainly due to the higher quantity of cement used for the mixing process, compared to the quantity of cement within the concrete substituted. For dredging spoils (Figure 10n), positive external costs are mainly due to the production of the flocculant (i.e. polyacrylamide) added in the conditioning process. Across all scenarios investigated, the most important contribution to the costs are generally internal costs. Similarly, for landfilling and incineration the internal costs are more significant than the external ones.

Figure 10. Societal Life Cycle Costing (SLCC) results per tonne of CDW fraction managed with breakdown of the contributions representing internal and external costs. Values above zero represent costs, while values below zero represent revenues. The final net SLCC is the sum of costs and revenues and is represented with a black dot. For the abbreviations used, please refer to Table 11.



Source: Own elaboration.

5.2.3 Sensitivity analysis results

This section presents the results of the four sensitivity analyses. This is done by comparing the impacts of the sensitivity scenario(s) with those of the base case assessment. First, doubling the distance to treatment or disposal does not affect the ranking of the scenarios, across all material fractions investigated. However, it is important to highlight that pathways producing recycled aggregates (REC-RA) are more sensitive to distance variations. This is due to a lower relevance of other parameters, such as material substitution and processing for this specific recycling pathway. Similarly, applying *in-situ* treatments (only tested on the excavated soils fraction) does not affect the ranking on the Climate Change category and cost.

Regarding the sensitivity on the substitutability factors, results for Climate Change show that most of the CDW fractions are extremely sensitive to the substitutability factor. However, pathways leading to recycled aggregates (REC-RA), are less sensitive to the substitutability factor. This is due to a lower relevance of this parameter for this specific recycling pathway, as the saving contribution from substitution of natural aggregates is indeed limited, making processing and transport relatively more important. For the excavated soils, halving the substitution factor leads to a change in the ranking of the scenarios, making preparing for reuse/backfill and recycling the best options and recycling through stabilisation worse than landfilling. Results on costs show that halving the substitutability parameters has limited influence on the results and does not affect the ranking of scenarios.

The sensitivity analysis focusing on a low-carbon EU energy mix (i.e. year 2050, based on electricity and heat with a higher percentage of renewable energies compared to 2020) shows that the influence on Climate Change for the scenarios consuming high quantities of electricity (e.g. recycling) or leading to energy recovery (e.g. incineration) is paramount. In general, the performance of energy recovery scenarios is drastically reduced in 2050, because less credits are achieved substituting energy (assumed to be cleaner). Contrarily, the performance of recycling is only partly affected as, while it is true that the impact of virgin production (substituted via recycling) is reduced, processing impacts associated with recycling technologies also decrease (cleaner energy used). This is evident for the case of wood waste. For excavated soils, the use of a cleaner energy mix can foster recycling, which would be the best option performing, favourable to preparing for reuse. Since no change in the price of the electricity has been modelled, this aspect would not influence the cost. The results of this sensitivity analysis may be consulted in Annex 2 (Figure A1). Finally, the results of the sensitivity analysis on wood waste including further cascading uses were shown in Figure 8b and commented on Section 5.2.1. Further details on the results can be found in Annex 2 - Table A15. We included the analysis of the possible effect of a low-carbon EU energy mix on treating wood waste including further cascading uses as previously commented (see Annex 2 – Table A16).

6. Potential for preparing for reuse and recycling in the EU

Sections 6.1 and 6.2 summarise the information available in the techno-scientific literature with respect to recycling and reuse potential (generally intended as theoretical/technical potential) for the individual fractions of CDW. The information detailed in these sections is the basis for the assumptions made to quantify GHG savings and costs associated with increased recycling (scenario: MRP) and increased preparing for reuse and recycling (scenario: MPP) in Section 6.3.

6.1 Potential for recycling

The recycling potential of several material fractions has been estimated on the basis of a literature review and stakeholder consultations. The resulting values are collected in Table 13.

For concrete waste, 100% recycling (into RA and/or cementitious materials) appears achievable, as suggested in Wahlström et al. (2014). We apply this value for further calculations. Where wood is concerned, the literature flags that recycling is possible only for certain types of wood waste (often referred to as high-quality wood waste) and the recycling potential varies between 60%, for a case study in Denmark (Faraca, Tonini, et al., 2019), and 44%, for a case study in Germany (Höglmeier et al., 2017), in relation to the total wood waste generated. We apply the latter value for further calculations. As for steel and aluminium, stakeholders (i.e. Metals in Buildings and European Aluminium) suggest that it is feasible to collect and send to preparing for reuse/recycling up to 99% of the waste produced. For gypsum, recovery rates of up to 95% are reported by Vrancken & Laethem (2000). Similar to concrete, for bricks and ceramic and tiles, a maximum recycling of 100% is possible. For glass and insulation materials (i.e. glass wool and stone wool), Mulders (2013) and C. Zhang et al. (2021) report that 100% was sent to recycling in selected case studies in the Netherlands. For PVC waste, Lase et al. (2023 - Table S5) report a recycling potential of ca. 90%[20]. For EPS waste, Lase et al. (2023 - Table S5) suggest that 14% recycling is realistically achievable, while a study by Conversio reports that in some EU countries EPS recycling from CDW is already happening at rates of around 20-27% (e.g. Czechia 27%, Austria 20%; Lindner et al., 2020). We thus assume that this figure is realistically achievable based on best practices, and we apply it to our calculations. For excavated soils and dredging spoils, the literature suggests that 100% recycling is possible. Table 14 summarises the assumptions on the recycling rates taken for the quantification of GHG savings and costs associated with increased recycling in Section 6.3.

6.2 Potential for preparing for reuse

The reuse potential of several material fractions has been estimated on the basis of a literature review and stakeholder consultations. The resulting values are described in this section and collected in Table 13.

Concrete: based on the literature review, it is assumed that only precast concrete can be reused. The amount of precast concrete has therefore been estimated and multiplied by its the reuse potential, which, according to Iacovidou & Purnell (2016), is around 50%. Further details on the calculation are available in Table 13.

Wood: as timber can perform a variety of functions within a building, estimating its reuse potential is particularly complex. Even within the context of one specific building element, reuse potential values can sometimes vary from one source to another. Wooden flooring, for instance, has a reuse potential that ranges between around 50% and 85% (Gorgolewski & Ergun, 2013; Iacovidou & Purnell, 2016; Sassi, 2002). Doors have a reuse potential of around 50%, while the figure is lower for window frames (<50%) (Gorgolewski & Ergun, 2013). As structural timber represents the highest share of wood in buildings, it has been used in the present report as a proxy to estimate the reuse potential of construction timber as a whole. Several values are available in literature: Höglmeier et al. (2017) estimate a 25% reuse potential based on a 2011 German case study; ARUP (2021) states that a survey conducted in 1998 the UK showed a 30% salvaged wood rate; the Institute for Local Self-Reliance (2006) assumes a reuse potential of 39% in buildings in fair condition and 59% in buildings in good conditions (data relative to the US); finally, Iacovidou & Purnell (2016) estimate the reuse potential as amounting to >50%. The study from Höglmeier et al. (2017) is used in the present report, as it provides a precise number, refers to a European country, and is relatively recent.

[20] Calculated with the Max transfer coefficients of collection and on-site sorting.

Steel and aluminium: according to Cooper & Allwood (2012), structural steel alone has a reuse potential of 79% on a global level; however, that value decreases to 38% when considering the total steel content in buildings, and to 29% when taking into account the construction sector as a whole. According to the same source, aluminium has a reuse potential of 50%. These values, which have been used in the present work, are largely in line with the 40% overall reuse potential estimated by way of a stakeholder survey by Hartwell et al. (2021) for building frame materials (i.e. steel and aluminium).

PVC: on the basis of the literature review in general, and Iacovidou & Purnell (2016) in particular, PVC has been assumed to have no potential for reuse, being better suited to recycling.

EPS: based on the literature review, it has been assumed that EPS has no potential for reuse.

Gypsum: as a category, gypsum includes both plaster and plasterboard. The former has no reuse potential. Information on the latter is less well defined. According to Gorgolewski & Ergun (2013) and Iacovidou & Purnell (2016), its reuse potential is “weak” (<50%). Given a variety of other sources indicating that reusing gypsum plasterboard is either not an option, or theoretically but not practically feasible (Klinge et al., 2022; Monier et al., 2011; Pristerà et al., forthcoming; Sandin et al., 2021; Thormark, 2000), the assumption in the present work is that the material has a reuse potential of 0%.

Ceramics & Tiles: there is a distinction to be made between roof and floor tiles. Roof tiles seem to have a higher reuse potential (>50% according to Iacovidou & Purnell (2016), and 60% according to Blomaard (2020), though it should be noted that the latter includes both cement and clay tiles), while this value is much lower for ceramic floor tiles (2%, according to Sassi (2002)). However, the share of each of these elements in the corresponding waste fraction is not known. Using stakeholder consultations as a basis, an overall 10% reuse potential has been estimated for this category (further details are available in Table 13).

Glass wool and mineral wool: based on the literature review, it has been assumed that glass wool has no potential for reuse. According to Gorgolewski & Ergun (2013) and Iacovidou & Purnell (2016), stone wool has low reuse potential (<50%). However, a variety of other sources indicate that stone wool, and mineral wool in general, has no potential for reuse and is better suited for recycling (Domonkos et al., 2022; NFDC, 2023; Thormark, 2000; WOOL2LOOP, 2020); it has therefore been assumed that the reuse potential of stone wool is also equal to 0%.

Bricks: according to Iacovidou & Purnell (2016), clay bricks bonded with cement mortar have a 0% reuse potential, while those bonded with lime mortar have a high reuse potential (>50%). However, other factors are also to be taken into account in this analysis, such as the size and technical characteristics of the bricks. In order to develop a valid estimate of the reuse potential of bricks at the EU level, stakeholders have been consulted, leading to establishing values for Denmark in particular. These values have then been scaled up to EU27 as detailed in Table 13.

Glass: according to Gorgolewski & Ergun (2013) and Iacovidou & Purnell (2016), glass has a low reuse potential (<50%); Rota et al. (2023) estimated the reuse potential of insulated glass units at 33%, while according to Hartwell et al. (2021), glazing glass has an average reuse potential of 20%. The latter value has been used in the current work, as it refers to the overall glass content in buildings and is rather recent.

Table 13. Estimation of the share of each waste material fraction that can potentially be sent to recycling and 'preparing for reuse' (columns #3 and #6), compared to the quantities reported in the baseline (columns #2 and #5). Quantities in brackets report the percent of material available after recycling and reuse, i.e. corrected for process losses.

Material fraction	Baseline: currently sent to recycling	Potential sent to recycling (% waste recycled)	Reference	Baseline: currently sent to preparing for reuse	Potential sent to preparing for reuse (% waste reused)	Reference
Concrete	79% (79%)	100% (100%)	For concrete waste, 100% recycling (into RA and/or cementitious materials) appears possible, as suggested in Wahlström et al. (2014).	0%	13% (13%)	It is assumed that only precast concrete can potentially be reused, based on the review of (Pristerà et al., forthcoming). The amount of precast concrete is calculated based on Business Market Insight[21], which estimates the EU ready mix concrete market value in 2022 as 113 140 million US$ and the precast concrete market value in 2022 as 29 799 million US$ (i.e. the share would be equal to ca. 26%). Iacovidou & Purnell (2016) estimate that precast concrete has a reuse potential of around 50%; the overall reuse potential of concrete in buildings is therefore estimated at 13%.
Wood	30% (25%)	44% (37%)	The literature flags that recycling is possible only for specific types of wood waste (often referred to as high-quality wood waste), with a recycling potential varying between 60% for a case study in Denmark (Faraca, Tonini, et al., 2019) and 44% for a case study in	0%	25% (25%)	A 25% rate is assumed, based on the findings of a case study in Germany detailed by Höglmeier et al. (2017).

[21] https://www.businessmarketinsights.com/reports/europe-precast-concrete-market.

			Germany (Höglmeier et al., 2017) out of the total wood waste generated. We use the latter for further calculations.			
Steel	84% (70%)	89% (75%)	Stakeholders (i.e. Metals in Buildings and European Aluminium) suggest that up to of 99% of steel could feasibly be collected and sent for recycling. Note that 10% is already sent to preparing for reuse.	10% (10%)	29% (29%)	Value based on Cooper & Allwood (2012).
Aluminium	84% (77%)	89% (82%)	Stakeholders (i.e. Metals in Buildings and European Aluminium) suggest that up to of 99% of aluminium could feasibly be collected and sent for recycling. Note that 10% is already sent to preparing for reuse.	10% (10%)	50% (50%)	Value based on Cooper & Allwood (2012), who assessed the potential reuse of aluminium in the construction sector on a global level. As construction technologies do not undergo significant regional variations with regards to aluminium, it is assumed that the same reuse potential can be applied at the global and European level.
Plastic PVC	30% (26%)	90% (78%)	For PVC waste, Lase et al. (2023) report a recycling potential of ca. 90%.	0%	0%	It is assumed that reuse is not feasible, based on data from Iacovidou & Purnell (2016).
Plastic EPS	10% (7%)	27% (19%)	For EPS waste, Lase et al. (2023) suggest that 14% recycling is realistically achievable, while a study by Conversio reports that in some EU countries EPS recycling from CDW is already happening at rates of around 20-27% (e.g. Czechia 27%, Austria 20%; Lindner et al., 2020). We thus assume that this figure is realistically achievable based on best	0%	0%	It is assumed that reuse is not feasible, based on the analysed literature.

			practices, and we apply it to our calculations.			
Gypsum	10% (10%)	95% (94%)	For gypsum, recovery rates of up to 95% are reported by Vrancken & Laethem (2000).	0%	0%	It is assumed that reuse is not feasible, based on the analysed literature.
Ceramics & Tiles	79% (73%)	100% (92%)	Similar to concrete, a maximum recycling potential of 100% is possible for ceramic and tiles.	0%	10% (7%)	Stakeholders pointed to some niche businesses collecting roof tiles and preparing them for reuse. On this basis, knowing the market for ceramics, we calculated that about 10% of the total generated C&T waste consist of roof tiles and be potentially available for preparing for reuse. As for the reuse process itself and related losses, in the absence of specific data, we assume the same process losses as for bricks.
Glass wool	2% (2%)	100% (80%)	Mulders (2013) and C. Zhang et al. (2021) report that 100% of insulation materials (i.e. glass wool and stone wool) was sent to recycling in selected case studies in the Netherlands.	0%	0%	It is assumed that reuse is not feasible, based on the analysed literature.
Stone wool	2% (2%)	100% (80%)	Mulders (2013) and C. Zhang et al. (2021) report that 100% of insulation materials (i.e. glass wool and stone wool) was sent to recycling in selected case studies in the Netherlands.	0%	0%	It is assumed that reuse is not feasible, based on the analysed literature.
Bricks	79% (73%)	100% (93%)	Similar to concrete, a maximum recycling potential of 100% is possible for bricks.	0%	59% (40%)	To estimate the *preparing for reuse* potential at the EU level, we assume that, similarly to Denmark, solid bricks (in contrast to engineering bricks for non-visual purposes) can be reused in countries with similar

						building practices as Denmark[22]. Relying on the figures from Damgaard et al. (2022; Table F25), the waste brick quantity in the EU amounts to ca. 8.5 million t from which 5.3 million t are from the abovementioned countries (i.e. Western Europe), leading to an estimated rate of preparing for reuse . Note that, out of the total amount that could be sent for preparing for reuse, only a portion is reused. According to stakeholders, after demolition ca. 95% is recovered and sent for preparing for reuse (due to some material breaking), and of this, about 57-65% could be reused, as suggested by Douguet & Wagner (2021) and Kancheva & Zaharieva (2023), while 15% of the material sent to preparing for reuse constitutes low-quality RA used as a replacement for natural aggregates (no structural applications) and ca. 20% is mortar, to be used as RA. Thus, the potential preparing for reuse is 59%.
Glass	6% (6%)	100% (97%)	Mulders (2013) and C. Zhang et al. (2021) report that 100% of glass was sent to recycling in selected case studies in the Netherlands.	0%	20% (20%)	Assumption based on Hartwell et al. (2021).
Excavated soil	35% (34%)	100% (97%)	For excavated soils and dredging spoils, the literature information suggests that 100% recycling is possible.	0%	100% (100%)	For excavated soils and dredging spoils, it is assumed that 100% could be sent to preparing for reuse with no losses. A demonstration experiment in Høje-Taastrup (Denmark) within the project CityLoops reports that more than 90% of excavated soil is
Dredged spoil	8% (8%)	100% (97%)		0%	100% (100%)	

[22] Denmark is grouped with Western Europe (including Austria, Belgium, Denmark, France, Germany, Ireland, Luxembourg, the Netherlands) since Danish building practice is considered similar to that of other western EU countries, rather than to that of Nordic countries, based on the observed share of wood waste in the CDW data (according to Damgaard et al. (2022)).

						prepared for reuse when developing the area (CityLoops, 2020).
Total CDW (weighted average)	37% (36%)	96% (93%)	Weighted average using data from Table 2 and data within this table.	0.1% (0.1%)	83% (82.7%)	Weighted average using data from Table 2 and data within this table.
Total CDW (weighted av. – excl. excavated soil and dredging spoil)	61% (59%)	81.5% (79%)		0.7% (0.7%)	15.5% (14%)	

Source: Own elaboration.

6.3 GHG savings and costs in the increased recycling and preparing for reuse scenarios

This section explores two scenarios that represent the maximum recycling potential (MRP) and the maximum preparing for reuse potential (MPP) at the EU level. Both scenarios build on the idea of assessing the maximum potential benefits from increased recycling and preparing for reuse at the EU level. Thus, in the MRP scenario, the highest possible rate of recycling is assumed. In the MPP scenario, the maximum potential for preparing for reuse is assumed whenever this option has been identified as possible for the selected material fraction. For those materials for which preparing for reuse is not a feasible option, it is assumed that the maximum recycling rate, as estimated for the MRP scenario, applies (Table 14). In both scenarios, landfilling (of mineral waste) and incineration (of waste with calorific value) are set to a minimum rate to close the mass balance. Backfilling operations are also reduced to the minimum possible value, acknowledging their low position in the waste hierarchy. To model recycling technologies, both scenarios assume the implementation of the best performing pathways based on the results of the LCA. Financial constraints are not considered.

The potential rates of recycling and preparing for reuse applied in the calculations are summarised in Table 14. Note that the shares refer to the input to the individual management options (e.g. % sent to preparing for reuse or % sent to recycling). These are then corrected accounting for technical inefficiencies represented by material losses at sorting and recycling stage in the LCA model. Refer to Table A17 in Annex 3 for the corrected quantities according to the recycling calculation rules reported in Caro, Albizzati, et al. (2023). The MRP and MPP scenarios are then compared to the baseline (BSL), which represents the *status quo* of the CDW management in the EU in 2020 for each material fraction (the information relative to the baseline, initially detailed in Table 9, is also reported in Table 14). On the basis of this comparison, we can quantify the impacts and costs associated with moving from BSL to MRP and from BSL to MPP, for illustrative purposes.

In order to estimate the environmental impact and cost of shifting from BSL to either MRP or MPP in the EU, the material flows reported in Table 2 are multiplied by the treatment shares reported for each fraction in the BSL, the MRP and the MPP scenarios (Table 14) and by the Climate Change impacts calculated in Section 5.2.1 and the costs calculated in Section 5.2.2 (in kg CO_2-eq. per tonne of managed waste and Euros per tonne of managed waste, respectively). A delta between the impacts or costs of BSL and MRP or MPP is then calculated. Note that for this estimation we assume that only the best performing recycling processes are employed in the MRP and MPP scenarios (for concrete, bricks, ceramics and tiles); for excavated soils and dredging spoils, the recycling process selected for the calculation is the one producing individual materials; for wood in particular, the results in the sensitivity analysis considering the cascading effect are used. Instead, in the BSL the current recycling processes aiming at producing only RA are used. Figure 11 and Figure 12 show the marginal abatement cost curve (MACC) for the MRP and MPP scenarios, respectively. Note that the baseline values are accounted for as avoided impacts and costs.

Table 14. Partitioning of the generated CDW across management pathways in the two scenarios analysed (MRP and MPP). The percentage reflects the share of the CDW generated sent to the treatment (e.g., sent to 'preparing for reuse') and not what is considered reused or recycled in practice. However, losses are duly considered in the LCA calculation.

	Baseline (BSL)					Maximum Recycling Potential (MRP)					Maximum Preparing for Reuse Potential (MPP)				
	Management pathways %					Management pathways %					Management pathways %				
Fraction	REU	REC	RBB	INC	LAN	REU	REC	RBB	INC	LAN	REU	REC	RBB	INC	LAN
CON	0	79	10	0	11	0	100	0	0	0	13	87	0	0	0
WOD	0	30	0	64	6	0	44	0	56	0	25	19	0	56	0
STE	10	84	0	0	6	10	89	0	0	1	29	70	0	0	1
ALU	10	84	0	0	6	10	89	0	0	1	50	49	0	0	1
PVC	0	30	0	12	58	0	90	0	10	0	0	90	0	10	0

EPS	0	10	0	66	24	0	27	0	73	0	0	27	0	73	0
GYP	0	10	0	0	90	0	95	0	0	5	0	95	0	0	5
C&T	0	79	10	0	11	0	100	0	0	0	10	90	0	0	0
GLW	0	2	0	0	98	0	100	0	0	0	0	100	0	0	0
STW	0	2	0	0	98	0	100	0	0	0	0	100	0	0	0
BRK	0	79	10	0	11	0	100	0	0	0	59	41	0	0	0
GLA	0	6	24	0	70	0	100	0	0	0	20	80	0	0	0
ESR	0	35	40	0	25	0	100	0	0	0	100	0	0	0	0
DDS	0	8	4	0	88	0	100	0	0	0	100	0	0	0	0
Total*	0	37	29	0	30	0	96	0	0	0	83	13	0	0	0
Total (exc. ESR & DDS)*	1	61	8	2	13	1	82	0	2	0	16	67	0	2	0

*Total calculated as weighted average based on the share of each individual material fraction in the total CDW. Note that the partitioning between the management pathways does not sum to 100% because there are some fractions of CDW that have been excluded from the analysis (mainly mixed inert, representing ca. 14%, but also minor fractions such as cardboard, electronics, paint and glue that represent ca. 1% of the total CDW).

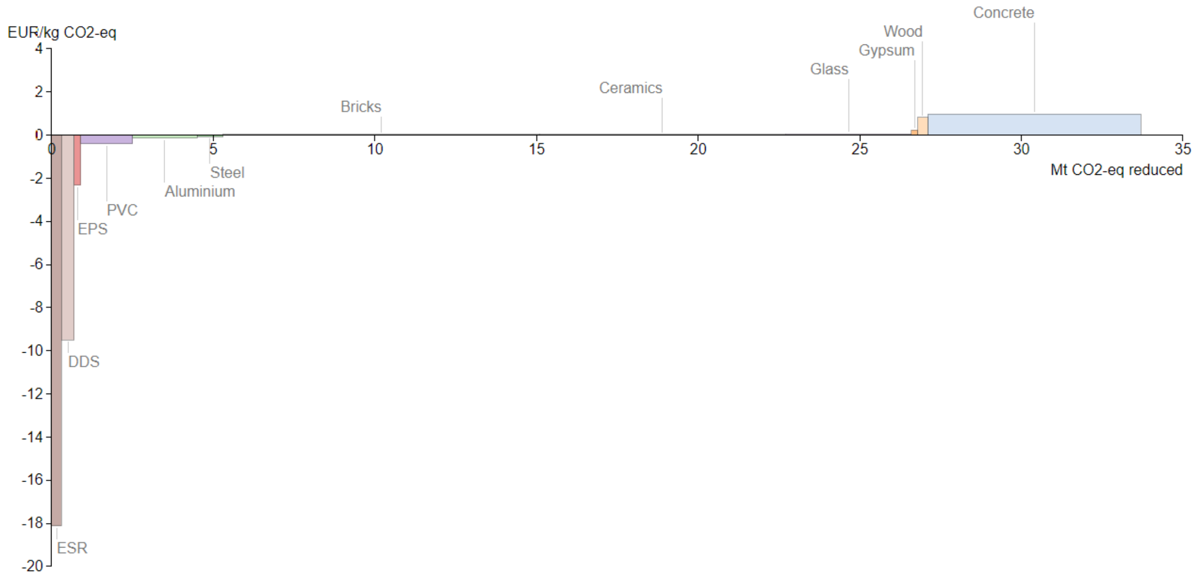
Note - CON: concrete; ALU: aluminium; BRK: bricks; C&T: ceramic and tiles; DDS: dredging spoils; EPS: expanded polystyrene; ESR: excavated soil; GLA: glass; GLW: glass wool; GYP: gypsum; INC: incineration; LN: landfilling; PVC: poly vinyl chloride; REC: recycling; RBB: recovery/backfilling. STE: steel; STW: stone wool; WOD: wood.

Source: Own elaboration.

The MACCs represent the quantity of potential GHG savings at the EU level (in Mt) for each fraction and with the marginal cost assumed to be constant (EUR per kg of CO_2 eq.). Note that selected unit-costs are negative because the cost of the baseline is avoided (thus subtracted), which leads to financial savings when shifting from the BSL to the MRP and MPP scenarios. Boxes on the x-axis represent the different fractions analysed, and the ones above the x-axis indicate that the actions have a net cost– the higher the box, the higher the cost. For instance, in the MACC of the MRP scenario for concrete (see Figure 11), a cost of EUR 1 per t of CO_2-eq. reduced is shown. On the other hand, boxes below the x-axis indicate a net saving from that action – the lower the box, the greater the saving. For instance, in the MACC of the MRP scenario for aluminium (see Figure 11), a saving of EUR 0.14 per kg of CO_2 eq. reduced is observed. This information is then combined with the width of the boxes indicating the action’s potential volume of reduction, expressed as Mt of CO_2 eq.

As shown in Figure 11 for the MRP scenario, the fraction with the highest GHG emission saving potential is the summed inert fraction (bricks, ceramics and tiles, glass, concrete) totalling 27.9 Mt of CO_2-eq. Concrete could contribute with 6.6 Mt CO_2-eq. at a total cost of EUR 6.5 billion (marginal cost of EUR 1 kg^{-1} CO_2-eq.). The lowest marginal cost (which is negative, and therefore a saving) is associated with dredging spoils and excavated soils (leading to total savings of EUR 3.8 billion and EUR 5.4 billion, respectively), which however incur limited GHG emission savings (i.e. 0.4 and 0.3 Mt of CO_2 eq. in 2020, respectively). Plastics (EPS and PVC) could contribute with 0.2 and 1.6 Mt CO_2-eq. savings respectively, and metals with 2.8 Mt CO_2-eq. (2.0 Mt CO_2-eq. for aluminium and 0.8 Mt CO_2 eq. for steel, both largely recycled already). Overall, accounting for the total (non-hazardous) CDW from buildings, soil and dredging spoil generation (131 Mt, 444 Mt, and 79 Mt, respectively; totalling 652 Mt), the MRP scenario could lead to a total reduction of ca. 34 Mt CO_2-eq. using 2020 waste generation figures and a net cost saving of ca. EUR 2.9 billion. Excluding soil and dredging spoils, for which cost savings are considered very uncertain, the GHG saving potential is ca. 33 Mt CO_2-eq. at a cost of EUR 6.3 billion (detailed information in Table 15).

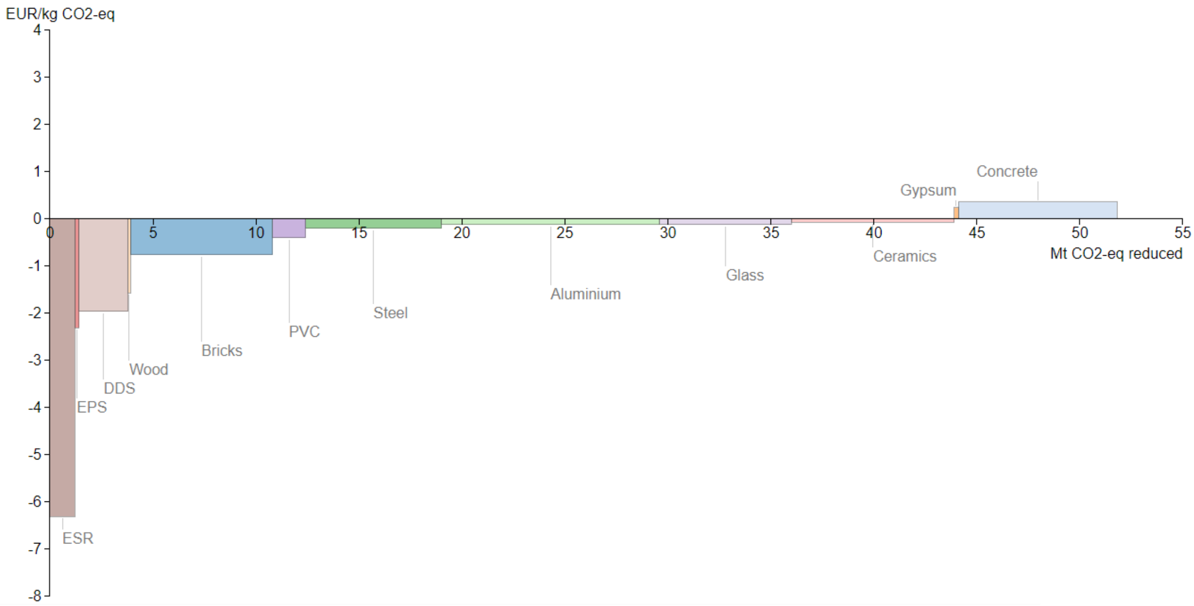
Figure 11. Marginal abatement cost curve for the maximum recycling potential (MRP) scenario. Note that the impacts and costs of the baseline are accounted for as avoided impacts and costs. DDS: dredging spoils; EPS: expanded polystyrene; ESR: excavated soil; PVC: polyvinyl chloride.



Note that for wood the results for the cascading calculation for four cycles (i.e. the results of the sensitivity analysis) are here considered for the MACC.

Source: Own elaboration.

Figure 12. Marginal abatement cost curve for the maximum preparing for reuse potential (MPP) scenario. Note that the impacts and costs of the baseline are accounted for as avoided impacts and costs. DDS: dredging spoils; EPS: expanded polystyrene; ESR: excavated soil; PVC: polyvinyl chloride.



Source: Own elaboration.

As shown in Figure 12, the fraction with the highest GHG emission saving potential in the MPP scenario is the summed inert fraction (bricks, ceramics and tiles, glass, concrete) totalling 28.9 Mt of CO_2-eq. Concrete could contribute with 7.7 Mt CO_2-eq. at a net cost of EUR 2.8 billion (marginal cost of EUR 0.37 kg^{-1} CO_2-eq.). The lowest marginal cost (negative, i.e. a saving) is associated with dredging spoils and excavated soils (leading to total savings of EUR 4.7 billion and EUR 7.6 billion, respectively), which however incur limited GHG emission saving (i.e. 2.4 and 1.2 Mt of CO_2-eq. in 2020, respectively). Plastics (EPS and PVC) could contribute with 0.2 and 1.6 Mt CO_2-eq. savings respectively, and metals with 17.2 Mt CO_2-eq. (10.6 Mt CO_2-eq. for aluminium and 6.6 Mt CO_2 eq. for steel). Overall, accounting for the total (non-hazardous) CDW from buildings, soil and dredging spoil generation (131 Mt, 444 Mt, and 79 Mt, respectively; totalling 652 Mt), the MPP scenario would lead to a total reduction at EU level of ca. 51.5 Mt CO_2-eq. using 2020 waste generation figures and a net cost saving of approximately EUR 19.5 billion. Excluding soil and dredging spoils, for which cost savings are considered very uncertain, the GHG saving potential is 48 Mt CO_2-eq., leading to savings of EUR 7.3 billion (more detailed information in Table 15).

Table 15. Marginal cost, total GHG emission reductions and total cost for the MPR and MPP scenarios by material fraction.

Fraction	MRP scenario			MPP scenario		
	Marginal cost (EUR kg^{-1} CO_2 eq.)	Total GHG reduction (Mt CO_2 eq.)	Total cost (M EUR)	Marginal cost (EUR kg^{-1} CO_2 eq.)	Total GHG reduction (Mt CO_2 eq.)	Total cost (M EUR)
CON	0.98	6.6	6495	0.37	7.7	2840
WOD	2.34	0.3	249	3.51	0.1	-157
STE	-0.08	0.8	-61	-0.20	6.6	-1297
ALU	-0.14	2.0	-286	-0.12	10.6	-1239
PVC	-0.39	1.6	-637	-0.39	1.6	-637
EPS	-2.53	0.2	-462	-2.53	0.2	-462
GYP	0.30	0.2	48	0.30	0.2	48
C&T	0.04	7.6	311	-0.06	7.9	-486
GLW	-13.00	-0.002	28	-13.00	-0.002	28
STW	-11.40	-0.002	28	-11.40	-0.002	28
BRK	0.03	9.8	341	-0.76	6.9	-5207
GLA	0.07	3.9	271	-0.11	6.4	-733
ESR	-18.10	0.3	-5434	-6.40	1.2	-7568
DDS	-9.50	0.4	-3800	-2.00	2.4	-4702
TOTAL	-	33.7	-2911	-	51.5	-19546
TOTAL excl. ESR/DDS	-	33.0	6323	-	47.9	-7276

Note: The totals are calculated assuming the total generation of building CDW (131 Mt), soil (444 Mt) and dredging spoil (79 Mt). CON: concrete; ALU: aluminium; BRK: bricks; C&T: ceramic and tiles; DDS: dredging spoils; EPS: expanded polystyrene; ESR: excavated soil; GLA: glass; GLW: glass wool; GYP: gypsum; PVC: polyvinylchloride; STE: steel; STW: stone wool; WOD: wood.

Source: Own elaboration.

7. Economic and non-economic market barriers

There are several studies in the literature documenting economic and non-economic market barriers for CDW preparing for reuse and recycling that are potential drivers of market inefficiencies (Bakas et al., 2019; Di Maria et al., 2020; Ghisellini et al., 2018; López Ruiz et al., 2020; Luciano et al., 2022; Oluleye et al., 2022; Villoria Sáez & Osmani, 2019). The literature review presented in Ghisellini et al. (2018) grouped CDW market barriers into five groups: economic; political; legislative; informative; and managerial. Bakas et al. (2019) explored the Danish market for secondary building materials. Most recently, the European Environment Agency (2022) developed a rubric for measuring well-functioning secondary material markets and applied it to specific waste fractions. These sources and others provide an overview of potential market inefficiencies impacting the private sector uptake of CDW and the expansion of secondary material markets for CDW.

In general, the literature identifies potential market inefficiencies driven by the following widely accepted principles:

- There is little or no difference between several CDW fractions and their primary material counterparts. Without differentiation, products compete on price and recycled products are often more expensive. For example, most CDW recycling results in low-value aggregates that compete with low-cost primary raw materials also used as aggregates. The next step of this argument is that prices of primary raw material products do not reflect the full life cycle costs of pollution generated due to the production and disposal phases of the same products. As recycled products include the cost of recycling, they are disadvantaged.
- However, doubts remain about the quality of recycled products in comparison to products from primary materials amongst some stakeholders (Luciano et al., 2022).
- There is often a disconnection between supply and demand in local markets for specific wastes. Due to low demand in local markets, firms find it difficult to find customers. The nature of CDW is bulky and heavy. As a result, markets for recycled construction products depend on local or regional factors such as housing and infrastructure growth and local regulation (C. Zhang et al., 2022). Regional and local factors may limit potential customer demand.
- A lack of information on building composition, meaning that reuse and recycling strategies may not be apparent until demolition is underway. Without detailed information on the components of the building it is difficult to plan for optimal removal for reuse and recycling (Lederer et al., 2020).
- A lack of financing needed to employ recycling or advanced recycling. CDW recycling capacity growth depends on high upfront investment costs. Investment in new capacity is slowed by perceived risk of demand (price signals) (Bakas et al., 2019). Investment risks stem from uncertainty of regulatory framework and uncertainty of supply.
- Cultural barriers and lack of knowledge on reuse and recycling techniques and capacity to implement (Bertozzi, 2022; European Environment Agency, 2022). Construction is a profession with entrenched practices that may not include reuse and recycling techniques for demolition. Also, knowledge of and/or perception of low quality for reused and recycling products during the design phase may limit market demand (Oyedele et al., 2014).

However, the literature does not agree on all potential market barriers. Notably, the effect of landfill taxes on CDW management has been studied extensively yet remains contested. The impact of landfill taxes is contested because some researchers find that landfill taxes are drivers of CDW treatment choices and others find that it is not a strong driver of CDW treatment. Villoria Sáez & Osmani (2019) reported that their analysis and several other studies concluded that “there was no correlation between the percentage of CDW landfilled and landfill taxation”. However, regional and municipal studies show that high landfill taxes reduce CDW sent to landfill. In general, the principle that low-cost landfilling disposal options displace recycling options holds. According to several review articles, landfill taxes are an important economic incentive to encourage recycling (Di Maria et al., 2020; López Ruiz et al., 2020; Luciano et al., 2022; Oluleye et al., 2022). Landfill taxes and disposal fees are a driver of recycling, if set appropriately, as well as a potential barrier to recycling if the cost is too low.

Markets for secondary materials depend upon the own-price elasticity of secondary materials. The literature points to low own-price elasticity of demand for many CDW fractions. For example, if own-price elasticity is low, when the price of aluminium goes up the quantity of aluminium would not immediately go down because it takes time for consumers to find alternatives. In general, in the short run, firms could pass on the cost of investments to consumers.

Is the CDW market well-functioning? It is important to remember that each fraction of CDW has its own related but differentiated market in Europe. These markets have different actors, technologies, barriers and histories. Therefore, it is difficult to generalise findings to all CDW fractions. Longstanding industries such as scrap metal have built-up networks, and management over time that have created a well-functioning market. The EEA's 2022 report, "Investigating Europe's secondary raw material (SRM) markets" illustrates the variation in CDW recycling markets.

> "SRM markets exist for many materials (metals, paper, wood, plastics, construction and demolition materials, biomaterials, etc.). Each of these markets is different in terms of its operational characteristics, historical and current developments, degree of closure of its material cycle, business and economic importance. While some markets have been well-established for a long time and are rather successful in providing a stable and relevant contribution to the circular economy, others still suffer from barriers to their further development. This remains the case even when they are targeted by strong waste and recycling policies at the EU and national levels."
>
> (European Environment Agency, 2022)

The EEA's report (European Environment Agency, 2022) establishes that CDW markets have diverse levels of success. For example, aluminium, wood, and aggregates:

- According to the EEA, aluminium has a well-functioning, mature market in Europe today. "Aluminium recycling rates are among the highest compared with those of other materials: in Europe, recycling rates are over 90% in the automotive and building sectors, and 75% for aluminium cans".
- At the same time, wood and aggregates do not currently have well-functioning markets. EEA concludes that the market for wood, "has the potential to be well-functioning. However, wood waste for recycling does not fully meet the criteria for a well-functioning SRM market in terms of the quantities (e.g. the share of SRM with respect to the total market) and the industrial capacity for producing SRM".
- EEA concludes that "the market for aggregate from CDW generally does not meet the criteria to be well-functioning. The markets for aggregates exist, the recycling is under-used and varies among Member States".

The results of the recent JRC stakeholder survey by Pacheco et al. (2023) reinforces two themes described in the literature. First, local/regional aspect of CDW markets is relevant to outcomes. Second, stakeholders' perception of quality (i.e. fit for use) will influence market outcomes. Pacheco et al. (2023) surveyed stakeholders on the use of recycled aggregates in concrete. They found that stakeholders "strongly support the idea that a major obstacle to the increased market uptake of recycled aggregate concrete is the <u>scarce availability of recycled aggregates fit for use</u> <u>in concrete</u>" *(emphasis added)* (Pacheco et al., 2023). The authors state that, "In countries where recycled aggregate concrete is not produced, the procurement of recycled aggregates adequate for concrete will encounter strong difficulties. In countries where recycled aggregate concrete is produced (sometimes): attempts to increase the uptake of recycled aggregates by the concrete industry will also face strong challenges in procurement." (Pacheco et al., 2023). The results emphasise the diversity of stakeholder views on a single waste fraction's non-economic market barriers. We may draw from these results that stakeholders of other waste fractions may be similarly diverse.

In summary, enabling measures are important to foster reuse and prevention and improving collection for recycling. Not all CDW fractions currently have well-functioning markets. The economic barriers include inefficient pricing that does not include environmental costs as shown in the ELCC above. The non-economic barriers include the cultural bias against secondary materials and lack of information about the technical qualities that slows down demand for secondary materials. As it is described in the potential for preparing for reuse and recycling in Section 6, both scenarios, MRP and MPP, have the potential to reduce the marginal abatement cost of CO_2 (EUR per kg of CO_2 eq.). The potential environmental benefits cannot be achieved without addressing the economic and non-economic barriers that continue to prevent CDW secondary material markets from becoming well-functioning and scaling up.

8. Limitations of the study and further research

8.1 Limitations related to the quantification of CDW generation and treatment

This study is based on the quantities reported to Eurostat complemented with literature, as detailed in the background study by Damgaard et al. (2022). There are some reporting issues concerning waste generation, notably for soil waste. There are also some reporting issues concerning treatment categories due to the different interpretation of backfilling among Member States. There is need for clarity on basic definitions such as backfilling (see inconsistencies in reporting by the same member state over the years, Box 1). Even if preparing for reuse is a recognised waste treatment category, there is no data within Eurostat. Specific data on the amounts sent for preparing for reuse is needed to compare the *status quo* of this management option against its full potential.

8.2 Limitations related to excavated soils and dredging spoils

Excavated soils and dredging spoils are usually excluded from CDW studies since they do not count for the recovery rate imposed on the WFD. One of the main limitations of this study is the lack of data and the uncertainty associated with these two fractions. Concerning the modelling, the composition of natural constituents and possible contaminants of excavated soil waste and dredging spoils is variable and case dependent. General values have been used within this study and the uncertainty has been solved herein through uncertainty analysis to account for it, but it should be considered case-by-case. For soils, at European level, the European Soil Data Centre (ESDAC) hosted by the European Commission's Joint Research Centre (JRC), is the focal point for soil data, and within the Land Use and Cover Survey (LUCAS) soil module datasets are available including particle size distribution at EU level, as well as coarse fragments (Panagos et al., 2022). Thus, soil coarse-, sand-, silt- and clay content is measured (in %) in the samples of topsoil (0-20 cm) of the LUCAS database in 2009 for 23 Member States (one of them is the UK, no longer a Member State) and then extrapolated to the full extent of the EU. Samples from Bulgaria and Romania were in 2012 sampled and included in the database (Tóth et al., 2013). In this study it is assumed that the subsoil has the same characteristics as the topsoil. A possible option to estimate data for the subsoil would be identify the type of soil (e.g. Cambisol, Histosol) from LUCAS and estimate the grain distribution based on that, as well as for the bulk density. For dredging spoils, there is no data at EU level of material composition that can differ from soils being usually dominated by silt and clay fractions (accounting for 60-90% of the solid content). In this study data from a case study in Sweden (Ferrans et al., 2019) is used to characterise EU dredging spoils.

Possible storage, as well as administrative and operational cost for material analysis has not been considered herein, thus maybe underestimating the real cost of management options. For all soil waste removals, a soil classification test to accurately assess contamination levels is needed. If the management option selected is landfill, probably a Waste Acceptance Criteria test is carried out to determine which type of landfill. On the other hand, for reuse, recovery and recycle options to achieve the EoW criteria, different analysis on the technical criteria of the final use must be conducted.

The savings and burdens associated with the recycling through stabilisation for soils and dredging spoils (either using cement or lime) are based on recipes from literature that calculates the quantity of binder depending on the characteristics of the soil or dredged spoils. As mentioned already in this section, this is very uncertain and has been somehow captured with the uncertainty analysis performed, but due to the great variability of the results, those options have not been considered in the calculation of the potential at EU level. More clarity is needed concerning this management pathway since it is not clear if it is commonly applied (a specific question on the survey to stakeholders was done concerning that and half of the respondents said that it is not commonly applied), as well as on how to report these quantities of treated soil to Eurostat. Furthermore, some stakeholders rose the attention to a possible long-term impact on the soil that should be further assessed.

For the potential for preparing for reuse and recycling in the EU, since no data for excavated soils and dredging spoils was available, it is assumed 100% potential in both cases, something not fully realistic but somehow showing the full potential of these two, usually, undervalued fractions. The recycling option to individual fractions is the selected option for the potential scenarios, even if as stated by the stakeholders this option is rarely done due to economic constraints. Within the scenarios, small increases in quantities treated through preparing for reuse and recycling lead to big differences in the marginal costs (acknowledging the probable underestimation of those cost treatments comparing to landfilling as mentioned before) and thus to great savings in the total costs. For that reason, the total numbers for both scenarios MRP and MPP concerning excavated soils and dredging spoils should therefore be taken with care.

8.3 Limitations related to the environmental and economic assessment

The savings associated with the recycling of concrete into RA of high-quality (e.g. for use in structural concrete) and of low-quality (e.g. for road sub-base or backfilling) are the same as in both cases we assume the replacement of natural gravel (no specific data available). While this should be improved, it is anticipated not to change significantly the overall environmental savings due to the very low impact of the extraction and processing of natural aggregates.

The savings associated with the recycling of bricks and ceramics/tiles into cementitious materials (scenarios named REC-CEM for each fraction) are uncertain as they are based on the process described in a single publication (Fořt & Černý, 2020). We could not prove the existence of a proven technology and market for this pathway. The related environmental savings of this recycling pathway are based on the assumptions taken following the study abovementioned and should therefore be taken with care.

The estimation of the potential share of waste material that could be subject to 'preparing for reuse' is highly uncertain (Table 13) as it is based on few sources, notably case studies, or authors' estimates. The same likewise applies to the environmental and economic savings associated with 'preparing for reuse' scenarios. The 'preparing for reuse' scenarios that were included in this study following a specific request of the stakeholders' consultation, are to be considered as illustrative and should be used with care. The only preparing for reuse scenarios for which the authors could prove the existence of established processing technologies and market is the case of metals (stakeholders suggested that 5-15% of the waste generated is currently sent to preparing for reuse) and solid bricks in Denmark. For the latter, ca. 3 million bricks are estimated to be currently processed annually for reuse while the total potential for reuse was estimated at 47 million bricks in previous studies[23].

The data used to represent the costs of recycling and the price of recycled materials were taken from literature, except for the case of reused bricks, and are thus subject to significant uncertainties. While we transparently reported sources (see Supplementary Information of Caro et al. (2024) for details) and strived to address uncertainties using state-of-the-art techniques, the specific magnitude of the ELCC results should be used with care. Nevertheless, we believe that the resulting costs of the scenarios investigated are robust enough to show tendencies and ranking between the different management scenarios (incineration, landfilling and the multiple recycling pathways assessed), e.g. to flag whether a selected recycling pathway is not competitive economically under given costs and prices.

[23] Based on feedback from the stakeholder consultations (specific meetings with Danish companies operating in the business of solid bricks reuse were held to get further insights into the reuse operations and market in Denmark and the potential at the EU level).

9. Stakeholder consultation

This section provides an overview of the stakeholder consultation process and results. In the course of preparation of this report, JRC actively sought stakeholder feedback through three channels:

1) Stakeholder review of the draft (and also the final) report;

2) Stakeholder workshop on the techno-economic and environmental assessment of construction and demolition waste management on July 21, 2023; and

3) Stakeholder survey, open for more than thirty days, until July 28, 2023.

The stakeholders were invited to participate in these communication channels via direct emails based on a list of stakeholders maintained by the JRC Circular Economy and Industrial Leadership Unit.

The following sections summarise the results and impact of the stakeholder consultation.

9.1 Dialogue on the techno-economic and environmental assessment report and workshop with stakeholders

Self-declared stakeholders were provided a draft of the report and a new survey before the workshop. Stakeholders had the opportunity to comment on the report directly through the survey app, during the workshop, or through direct email. Several stakeholders flagged new data sources and provided their own data to the Commission.

Also, the literature review by Damgaard et al. (2022), which informed the characterisation of CDW in the report, includes country-specific data from a separate stakeholder consultation.

The workshop with stakeholders on June 21, 2023, attracted approximately 60 stakeholders. Stakeholders who participated in the workshop called for coordination between Commission initiatives, including waste prevention, end of waste, and the EU Taxonomy. In addition, clarification was sought on specific scenarios, for example EoL scenarios (notably stakeholders asked to add 'preparing for reuse' for selected fractions), increasing demand for low carbon products and closed-loop recycling options rather than downcycling. Further, questions were asked about the greenhouse gas emissions calculated for wood. There were three overall threads that ran through the stakeholder's discussion:

- The lack of well-functioning markets for CDW products at prices that support uptake of CDW at scale for all fractions. The need to address economic and non-economic market barriers. Consumer information that highlights specifications and benefits of recycled and primary material products was noted by stakeholders.
- The need to distinguish between quality of CDW for recycling as high quality (with the meaning of meeting specifications of non-recycled products to be substituted) versus low quality (with the meaning of downcycling) in future work.
- The need for regulatory coordination at the Member State and EU level was mentioned, with examples of new Member State initiatives on CDW in Denmark, Spain, and Austria.

Finally, stakeholders linked the waste discussion to products from primary and secondary resources. In summary, stakeholders would like to see the Commission present a holistic perspective across the value chain that incorporates product design of the products on the market such that secondary materials and primary materials are on an equal footing.

At the end of the review process, stakeholders were given the opportunity to review again the report and send a final round of comments and feedback that were acknowledged and properly addressed.

9.2 Stakeholder survey results

The survey received 24 respondents of whom 88% were industry organisations, 4% were non-governmental organisations, 4% were research organisations, and 4% non-profit organisations.

The survey comprised five yes/no questions on the techno-economic assessment (see Table 16). The questions focused on topics that were not fully described in the literature, or the authors suspect that current practice is different than the literature indicated. Specifically, the questions concerned the composition of excavated soils, dredging spoils, and infrastructure waste, recycling technologies for CDW, relationships between excavated soils/dredging spoils, their recovery via stabilisation, and their use in agriculture, environmental impacts from the recycling life cycle stage. The average percentage of respondents that answered individual questions was 69%.

Although the percentage of respondents for every question was not above 50%, the survey tool was successful in eliciting additional data. Stakeholders provided additional data for four out of five questions. Question number 2, "Is the description in Section 4 of the background document covering the main recycling technologies for CDW or are there any additional innovative or emerging technologies that could play a role in the near future (2023-2035)?" elicited the most feedback. Stakeholders contributed data for recycling technologies of concrete, bricks, ceramics, mineral wool, glass, gypsum, metals, and plastics and the preparing for reuse of CDW.

Most survey respondents (100% response) were interested in the generation and composition of excavated soils, dredging spoils and infrastructure waste while the relationship between agriculture and excavated and dredging spoil was the topic with less answers (33%) since respondents focused on areas where they felt most comfortable with their knowledge.

The survey results are informative for the current study; however, given the small number of survey respondents, the results are not statistically significant for the entire EU. Rather, the results reflect expert industry and municipal knowledge on the topic.

Table 16. Template for questions within the survey and the statistics of the results obtained.

Questions	Percentage of Survey Respondents that answered the question	Percentage of Question Respondents that answered Yes	Percentage of Question Respondents that answered No	Did Respondents provide additional data? Yes or No
1. Are you aware of any (additional) data on the generation and composition of excavated soils, dredging spoils and infrastructure waste	100%	17%	83%	Y
2. Is the description in Section 4 of the background document covering the main recycling technologies for CDW or are there any additional innovative or emerging technologies that could play a role in the near future (2023-2035)?	88%	48%	52%	Y
3. Based on your knowledge, is recovery via stabilisation (with lime or cement) a commonly applied pathway for excavated soils and dredging spoils?	38%	55%	45%	N
4. Based on your knowledge, is use in agriculture a commonly applied pathway for excavated soils and dredging spoil?	33%	50%	50%	Y
5. Would you be able to provide additional data to update the environmental impacts from the recycling stage in the Life Cycle Assessment models used, and if so, would you be available to be contacted	88%	52%	48%	Y

by JRC to form part of a technical working subgroup on LCA/ELCC?				

Source: Own elaboration.

9.3 Stakeholder impact on final report

As a result of the workshop and survey, additional dialogues were initiated between the authors and stakeholders with expertise to share with JRC (see Table 17). These meetings were held on a rolling basis from spring 2023 until closing the final report. Follow-up actions resulted in changes in the report, particularly for the waste characterisation and life cycle inventory data. JRC follow ups to stakeholders led to the reviewing of Sections 2, 3 and 5. The impact of stakeholder's comments is summarised in Table 17. The authors are thankful for the data and information provided by the stakeholders that improved the final report.

Table 17. Relevant bilateral dialogues between JRC and stakeholders.

Date of Meeting / Contact / Visit	**Stakeholder's** Name and Organisation	Topic	Did follow-up meeting result in changes in the report (data, or facts)?
07/07/2023 (Additional visit to their premises on November 13, 2023)	HERCAL	CDW – mineral fraction. Treatment routes focused on concrete recycling and products obtained.	Refinement of information on the technologies and processes for concrete waste management.
04/08/2023	Concrete Europe	Infrastructure waste	Refinement of information on the generation and composition of infrastructure waste.
07/09/2023	DTI	Preparing for reuse of construction products.	Provided an overview of preparing for reuse of CDW materials and provided a contact with companies preparing for reuse bricks within Denmark. Expanded the reuse section in the final report.
17/09/2023	ECOS	CDW – insulation materials. Recycling quantities.	Refinement of information concerning the management of insulation material at EU level.
20/09/2023	Euro Panels	CDW - wood fraction. Carbon neutrality assumption and cascading principle.	Inclusion of sensitivity analyses on wood CDW accounting for the cascade cycle and future cleaner energy mix.
29/09/2023	Gamle Musler	CDW – brick fraction. Preparing for reuse of bricks in Denmark.	Refinement of information on the technologies and processes for bricks waste management and the reuse potential.

Source: Own elaboration.

10. Conclusions

EU waste legislation is driving improvements towards environmentally sound management of waste and spurring its contribution to the circular economy. As CDW represents almost 40% of the waste generated in the EU, this study focuses on the current management and related environmental and economic implications of CDW. We do this by identifying current and potential management options for all individual fractions of CDW (preparing for reuse, recycling, recovery, incineration and landfilling) and performing an in-depth techno-economic and environmental assessment of these management options. The study further provides an overview of the potential for preparing for reuse and recycling at the material fraction level and summarises the main economic and market barriers based on the available literature. In conclusion, this study lays the scientific groundwork for the European Commission's consideration of possible objectives and other measures designed to improve the management of CDW in the EU.

The Life Cycle Assessment results for the impact category Climate Change indicate that preparing for reuse and recycling are the options incurring the highest GHG savings, when assuming the use of best-performing recycling technologies. The highest GHG savings are achieved for preparing for reuse and recycling metals. Landfilling (or incineration when applicable) incurs the highest GHG burdens for all individual material fractions except for wood and mineral wool waste. When assuming the use of recycling processes that produce only recycled aggregates, savings from recycling are often comparable to (or only slightly better than) landfilling. The reason is that the GHG savings connected to avoiding natural material extraction and processing (gravel, sand) are limited. For the material fractions, mineral wool and wood waste, the results are different. Specifically, for glass wool and stone wool waste, landfilling performs comparably to recycling due to the limited GHG savings associated with the replacement of natural materials otherwise used in the mineral wool production process. For wood waste, incineration incurs higher GHG savings than recycling when cascading cycles are not accounted for and when considering the current EU energy mix for the calculations. These results are heavily affected by the following: i) biogenic CO_2 released from wood waste incineration is considered neutral with respect to Climate Change (i.e. no impact); ii) we only accounted for a single life cycle (i.e. no cascading cycles); and iii) we assumed the current EU energy mix. A sensitivity analysis on wood including cascading cycles shows that wood waste recycling (same for reuse) is preferable to incineration when a second cascading loop is included. Also, when accounting for a future cleaner EU energy mix, this gap further increases in favour of recycling (or reuse). These results highlight the importance of considering multiple cycles for the case of wood and the future EU energy mix. While one could also take a detailed look at the benefits from biogenic carbon storage (delayed emissions) and avoiding land use changes connected with the timber demand, this was out of the scope of the present analysis. It is understood that more data on biogenic carbon storage and land use change would further favour wood recycling over energy recovery options. For plastic waste (PVC and EPS), while recycling is the preferred option from a GHG reduction perspective, landfilling performs better than incineration due to the release of GHG upon incinerating the plastic, which is not compensated by the GHG savings from energy recovery[24]. For excavated soils and dredging spoils, preparing for reuse, recycling and recovery (backfilling) management options perform better than landfilling although overall GHG savings are limited for the same reasons as explained earlier for natural aggregates replacement. Note that the remaining impact categories generally follow a similar trend to that of Climate Change with respect to the ranking of the management scenarios.

The Environmental Life Cycle Costing results indicate that landfilling, when including an average EU landfill tax, is the worst economic option for half of the waste material fractions considered. For the other half, the cost of landfilling is simply cheaper. More advanced recycling pathways for concrete, ceramic and tiles, and bricks (to cement and aggregates) are (with data currently available) clearly more expensive than landfilling, mainly due to the processing costs although the cost increase due to selective demolition is also relevant. However, even simpler recycling processes producing only recycled aggregates appear to have comparable costs to landfilling overall and are thus in close competition with landfilling economically. Incineration, whenever applicable, is the most convenient option economically (with net income) due to the significant revenues from energy recovery. This is the case for plastic and wood waste owing to the high calorific value. For metals, preparing for reuse and recycling are clearly the most profitable options. It is important to note that, when external costs are accounted for via Societal Life Cycle Costing, recycling pathways significantly reduce the societal costs relative to landfilling and incineration. In conclusion, although the cost of recycling is higher than

[24] Given the assumptions in terms of average electricity and heat recovery in EU incinerators and without considering Carbon Capture and Storage.

landfilling for concrete, gypsum, ceramic and tiles, glass wool, stone wool, brick and glass and higher than incineration for wood, PVC, and EPS, the societal cost is lower when including externalities (for plastic waste always; for wood waste when cascading uses are considered). This means that landfilling and incineration are more costly options for society and should be further discouraged. It should be noted that our results in terms of costs are based on average figures and, therefore, should not be considered representative of the situation in all Member States. In particular, we apply a landfill tax for inert waste of EUR 19 t^{-1}, the average for the EU based on the data available from a current report of the European Environment Agency. However, some Member States apply much higher rates (up to EUR 100 t^{-1} or more, e.g. in the Netherlands) and this has been proven to have important implications in terms of waste diversion from landfill and increased recycling.

The study further explores the potential for recycling and preparing for reuse for each individual material fraction of CDW. The figures are based on a review of the available literature complemented with information collected from stakeholders. Overall, we found that the preparing for reuse and recycling rate potential could be in the range of 27-100% across the individual material fractions of CDW investigated, averaging 83% for CDW as a whole (excluding excavated soils and dredging spoils) (the values should be interpreted as the proportion of waste that can be 'sent for preparing for reuse and recycling', i.e. without considering the losses within the recycling or reuse process; the value drops to 79% when considering losses). Note that this figure is calculated excluding the mixed inert waste fraction (ca. 14%) and soil and dredging spoils (excluded from the recovery rate target of the EU Waste Framework Directive). When considering the recovery of the mixed inert fraction as recycling in the equation, the total recycling rate of CDW would rise to as much as 97%[25]. As for the potential for preparing for reuse alone, we estimated that this could vary between 0% and 50% depending on the material fraction (excluding excavated soils and dredging spoils), averaging 16% for CDW as a whole (to be interpreted as 'sent for preparing reuse'; the value drops to 14% when considering losses). These should be considered as preliminary estimates, based on literature and specific case studies, especially for the case of preparing for reuse. Based on these figures, two scenarios are analysed reflecting a maximum recycling rate scenario and a maximum preparing for reuse and recycling rate scenario. The first scenario is reducing landfilling and incineration to the minimum and assumes implementation of the best performing recycling processes following a best available technologies approach; the reuse rate is kept at the same levels as today. The second scenario follows the exact same assumptions as the first but prioritises preparing for reuse whenever applicable and to the maximum extent technically possible. The cost savings in the latter simulation are due to the theoretical savings gained by reusing rather than processing the waste via incineration, landfilling and recycling (as in the baseline for the year 2020) and should be considered as a theoretical tendency rather than an accurate cost estimate. The results of the assessment show that, compared to the baseline (*status quo* management of CDW in the EU; using 2020 waste generation figures) and excluding soil and dredging spoils, a total annual reduction of ca. 33 Mt CO_2 eq. at a net cost of approximately EUR 6.3 billion would be achieved with the maximum recycling potential scenario (up to 34 Mt CO_2 eq. savings at a net saving of approximately EUR 2.9 billion when including excavated soils and dredging spoils). In the same line, with the maximum preparing for reuse and recycling scenario a total reduction of ca. 48 Mt CO_2 eq. with a net cost saving of approximately EUR 7.3 billion would be achieved (up to 51.5 Mt CO_2 eq. savings at a net saving of approximately EUR 19.5 billion when including excavated soils and dredging spoils). Thus, preparing for reuse should be promoted along with recycling to maximise potential environmental and economic benefits.

Remarkably, using a Marginal Cost Abatement Curve, the study shows that all material fractions contained in CDW truly have non-negligible potential contributions to GHG reductions and environmental savings at below the current CO_2 price, except for wood, gypsum and concrete waste.

[25] Our estimate of 83% is thus not directly comparable with the current EU recovery rate of CDW of 89%, as estimated by Eurostat because i) Eurostat's 89% is a recovery rate (not a recycling rate) and ii) Eurostat's 89% includes mixed inert waste (which we did not include in the calculation leading to our 83%). If a comparison should be made, our value of 97% should be used instead, which is derived assuming that all of the 'mixed inert waste' fraction can be recycled or recovered to some form of recycled aggregates.

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List of main abbreviations and definitions

CAPEX	Capital Expenditure
CC	Climate Change
CD	Conventional Demolition
CDW	Construction and Demolition Waste
CRCA	Coarse Recycled Concrete Aggregate
DfD	Design for Deconstruction
EoL	End-of-Life
EoW	End-of-Waste
EPS	Expanded Polystyrene
EU	European Union
EWC	European Waste Code
FU	Functional Unit
GHG	Greenhouse Gas
LCA	Life Cycle Assessment
ELCC	Environmental Life Cycle Costing
LoW	List of Waste
MACC	Marginal Abatement Cost Curves
MFA	Material Flow Analysis
OPEX	Operational Expenditure
PVC	Polyvinylchloride
RA	Recycled Aggregates
RAC	Recycled Aggregate Concrete
RCA	Recycled Concrete Aggregate
SD	Selective Demolition
SLCC	Societal Life Cycle Costing
SRM	Secondary Raw Material
TRL	Technology Readiness Level
WFD	Waste Framework Directive

Definitions

Backfilling	is defined in Art. 3(17a) as "any recovery operation where suitable non-hazardous waste is used for purposes of reclamation in excavated areas or for engineering purposes in landscaping. Waste used for backfilling must substitute non-waste materials, be suitable for the aforementioned purposes, and be limited to the amount strictly necessary to achieve those purposes."
Material recovery	is defined in Art. 3(15a) of Directive 2008/98/EC as "any recovery operation, other than energy recovery and the reprocessing into materials that are to be used as fuels or other means to generate energy. It includes, inter alia, preparing for reuse, recycling and backfilling".
Preparing for reuse	means checking, cleaning or repairing recovery operations, by which products or components of products that have become waste are prepared so that they can be reused without any other pre-processing" (Directive 2008/98/EC, Art. 3(16)).
Recycling	means any recovery operation by which waste materials are reprocessed into products, materials or substances whether for the original or other purposes. It (...) does not include energy recovery and the reprocessing into materials that are to be used as fuels or for backfilling operations" (Directive 2008/98/EC, Art. 3(17)).

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Annexes

Annex 1. Remaining impact categories results

This annex reports the remaining impact category results for the analysis in 2020. The abbreviations in the following tables correspond to: OD – Ozone depletion; HT_CAN – Human toxicity, cancer; HT_NOCAR – Human toxicity, non-carcinogenic; PM – Particulate matter; IR – Ionising radiation; POF – Photochemical ozone formation; ACI – Acidification; EU_TER – Eutrophication, terrestrial; EU_FRE – Eutrophication, freshwater; EU_MAR – Eutrophication, marine; RU_EN – Resource use, energy carrier; RU_MIN – Resource use, minerals and metals; WU – Water use; LU – Land use; ECOTOX – Ecotoxicity, freshwater; SC – Societal costs.

Table A1. Remaining environmental impact categories for concrete

Management option	OD	HT_CAN	HT_NOCAR	PM	IR	POF	ACI	EU_TER	EU_FRE	EU_MAR	RU_EN	RU_MIN	WU	LU	ECOTOX	SC
	kg CFC-11 eq	CTUh	CTUh	Disease incidences	kBq U-235 eq.	mol H+ eq	mol N eq	kg N eq.	kg P eq.	kg N eq	MJ	kg SB eq	m3 water eq	-	CTUe	EUR
REU	9.6E-07	-9.7E-09	-4.3E-07	5.6E-07	-1.3E+00	-7.1E-02	-1.1E-01	-3.4E-01	-1.2E-03	-2.6E-02	-1.8E+02	-2.3E-04	-6.2E+01	-1.6E+02	-6.7E+02	-8.1E+01
REC-CEM	4.4E-06	2.2E-09	-9.6E-08	3.6E-06	1.9E-01	1.1E-01	5.2E-02	3.4E-01	-6.5E-04	3.3E-02	1.3E+02	-2.4E-05	-5.0E+01	2.5E+01	-1.9E+02	7.6E+01
REC-RA	2.1E-06	1.0E-09	7.3E-08	1.3E-06	4.1E-01	5.4E-02	4.5E-02	1.8E-01	2.2E-05	1.7E-02	1.3E+02	-2.5E-06	-5.0E+01	2.0E+01	7.9E+01	4.4E+01
LAN	3.28E-06	5.54E-09	1.30E-07	2.50E-06	9.42E-01	1.11E-01	9.14E-02	3.8E-01	8.34E-05	3.49E-02	2.15E+02	3.07E-05	7.42E-01	1.48E+02	1.53E+02	4.8E+01

Table A2. Remaining environmental impact categories for wood

Management option	OD	HT_CAN	HT_NOCAR	PM	IR	POF	ACI	EU_TER	EU_FRE	EU_MAR	RU_EN	RU_MIN	WU	LU	ECOTOX	SC
	kg CFC-11 eq	CTUh	CTUh	Disease incidences	kBq U-235 eq.	mol H+ eq	mol N eq	kg N eq.	kg P eq.	kg N eq	MJ	kg SB eq	m3 water eq	-	CTUe	EUR
REU	2.7E-06	-4.2E-08	-3.7E-07	9.5E-07	1.7E+00	-2.3E-01	-3.9E-01	-1.6E+00	-5.1E-02	-1.1E+00	4.5E+02	7.4E-05	-1.3E+01	-4.7E+04	-4.8E+03	-1.5E+02
REC-PBD	-1.3E-05	8.1E-08	-9.5E-07	-1.1E-05	-7.1E+00	-8.5E-01	-6.5E-01	-2.5E+00	-6.0E-03	-2.3E-01	-1.4E+03	1.0E-04	7.6E+00	-2.3E+05	-1.2E+03	-4.1E+00
LAN	3.28E-06	5.54E-09	1.30E-07	2.50E-06	9.42E-01	1.11E-01	9.14E-02	3.84E-01	8.34E-05	3.49E-02	2.15E+02	3.07E-05	7.42E-01	1.48E+02	1.53E+02	2.9E+01
INC	-5.0E-05	-1.6E-07	-3.3E-06	-1.5E-05	-5.5E+01	-1.4E+00	-3.0E+00	-5.8E+00	-1.4E-02	-4.7E-01	-1.2E+04	-1.2E-03	-1.2E+02	-1.8E+04	-9.3E+03	-4.4E+02

Table A3. Remaining environmental impact categories for steel

Management option	OD	HT_CAN	HT_NOCAR	PM	IR	POF	ACI	EU_TER	EU_FRE	EU_MAR	RU_EN	RU_MIN	WU	LU	ECOTOX	SC
	kg CFC-11 eq	CTUh	CTUh	Disease incidences	kBq U-235 eq.	mol H+ eq	mol N eq	kg N eq.	kg P eq.	kg N eq	MJ	kg SB eq	m3 water eq	-	CTUe	EUR
REU	-8.7E-05	-1.1E-05	-5.1E-05	-1.9E-04	-4.1E+01	-8.4E+00	-8.9E+00	-2.0E+01	-8.9E-02	-1.8E+00	-2.7E+04	-3.4E-02	-4.2E+02	-1.3E+04	-5.8E+04	-9.2E+02
REC-STE	-3.8E-05	-1.8E-06	5.3E-04	-7.0E-05	4.6E+01	-3.8E+00	-2.4E+00	-7.4E+00	-4.0E-02	-6.7E-01	-5.9E+03	-2.0E-02	-8.8E+01	-3.2E+03	-2.8E+04	-2.1E+02
LAN	3.28E-06	5.54E-09	1.30E-07	2.50E-06	9.42E-01	1.11E-01	9.14E-02	3.84E-01	8.34E-05	3.49E-02	2.15E+02	3.07E-05	7.42E-01	1.48E+02	1.53E+02	2.9E+01

Table A4. Remaining environmental impact categories for aluminium

Management option	OD	HT_CAN	HT_NOCAR	PM	IR	POF	ACI	EU_TER	EU_FRE	EU_MAR	RU_EN	RU_MIN	WU	LU	ECOTOX	SC
	kg CFC-11 eq	CTUh	CTUh	Disease incidences	kBq U-235 eq.	mol H+ eq	mol N eq	kg N eq.	kg P eq.	kg N eq	MJ	kg SB eq	m3 water eq	-	CTUe	EUR
REU	-7.5E-04	-2.4E-05	-3.6E-04	-7.5E-04	-5.2E+02	-3.2E+01	-7.1E+01	-9.6E+01	-4.0E-01	-8.6E+00	-1.5E+05	-1.7E-02	-9.5E+03	-2.0E+04	-2.2E+05	-3.3E+03
REC-ALU	-5.3E-04	-1.6E-05	-2.7E-04	-5.3E-04	-1.7E+02	-2.0E+01	-4.9E+01	-6.1E+01	-2.9E-01	-5.6E+00	-8.9E+04	8.2E-03	-6.8E+03	3.4E+04	-1.5E+05	-2.3E+03
LAN	3.28E-06	5.54E-09	1.30E-07	2.50E-06	9.42E-01	1.11E-01	9.14E-02	3.84E-01	8.34E-05	3.49E-02	2.15E+02	3.07E-05	7.42E-01	1.48E+02	1.53E+02	2.9E+01

Table A5. Remaining environmental impact categories for PVC

Management option	OD	HT_CAN	HT_NOCAR	PM	IR	POF	ACI	EU_TER	EU_FRE	EU_MAR	RU_EN	RU_MIN	WU	LU	ECOTOX	SC
	kg CFC-11 eq	CTUh	CTUh	Disease incidences	kBq U-235 eq.	mol H+ eq	mol N eq	kg N eq.	kg P eq.	kg N eq	MJ	kg SB eq	m3 water eq	-	CTUe	EUR
REC-PVC	-6.8E-04	-7.7E-07	-1.8E-05	-5.4E-05	-8.6E+00	-4.1E+00	-6.0E+00	-1.3E+01	-4.8E-02	-1.2E+00	-3.4E+04	-1.7E-02	-3.3E+02	-3.0E+03	-2.7E+04	-6.5E+02
LAN	3.28E-06	5.54E-09	1.30E-07	2.50E-06	9.42E-01	1.11E-01	9.14E-02	3.84E-01	8.34E-05	3.49E-02	2.15E+02	3.07E-05	7.42E-01	1.48E+02	1.53E+02	2.9E+01
INC	-1.3E-04	-3.9E-07	-9.1E-06	-3.9E-05	-1.2E+02	-3.6E+00	-7.2E+00	-1.5E+01	-3.2E-02	-1.2E+00	-2.9E+04	-2.8E-03	-2.7E+02	-4.5E+04	-2.3E+04	-7.9E+02

Table A6. Remaining environmental impact categories for EPS

Management option	OD	HT_CAN	HT_NOCAR	PM	IR	POF	ACI	EU_TER	EU_FRE	EU_MAR	RU_EN	RU_MIN	WU	LU	ECOTOX	SC
	kg CFC-11 eq	CTUh	CTUh	Disease incidences	kBq U-235 eq.	mol H+ eq	mol N eq	kg N eq.	kg P eq.	kg N eq	MJ	kg SB eq	m3 water eq	-	CTUe	EUR
REC-EPS	-4.3E-05	-3.6E-07	-6.7E-06	-7.5E-05	-3.7E+01	-5.4E+00	-8.4E+00	-1.5E+01	-1.1E-02	-1.3E+00	-4.7E+04	-1.0E-03	-1.1E+03	-1.5E+04	-9.0E+03	-8.7E+02
LAN	3.28E-06	5.54E-09	1.30E-07	2.50E-06	9.42E-01	1.11E-01	9.14E-02	3.84E-01	8.34E-05	3.49E-02	2.15E+02	3.07E-05	7.42E-01	1.48E+02	1.53E+02	2.9E+01
INC	-1.4E-04	-4.3E-07	-1.0E-05	-4.1E-05	-1.5E+02	-4.0E+00	-8.3E+00	-1.6E+01	-3.8E-02	-1.3E+00	-3.3E+04	-3.4E-03	-3.3E+02	-4.7E+04	-2.5E+04	-9.4E+02

Table A7. Remaining environmental impact categories for Gypsum

Management option	OD	HT_CAN	HT_NOCAR	PM	IR	POF	ACI	EU_TER	EU_FRE	EU_MAR	RU_EN	RU_MIN	WU	LU	ECOTOX	SC
	kg CFC-11 eq	CTUh	CTUh	Disease incidences	kBq U-235 eq.	mol H+ eq	mol N eq	kg N eq.	kg P eq.	kg N eq	MJ	kg SB eq	m3 water eq	-	CTUe	EUR
REC-GYP	-3.3E-05	-8.5E-08	-3.7E-07	-6.5E-06	-7.7E+00	-3.1E-01	-5.3E-01	-1.6E+00	-6.1E-03	-2.1E-01	-1.6E+03	-8.9E-04	-1.2E+02	-8.8E+03	-4.3E+03	4.8E+01
LAN	3.28E-06	5.54E-09	1.30E-07	2.50E-06	9.42E-01	1.11E-01	9.14E-02	3.84E-01	8.34E-05	3.49E-02	2.15E+02	3.07E-05	7.42E-01	1.48E+02	1.53E+02	2.9E+01

Table A8. Remaining environmental impact categories for ceramic & tiles

Management option	OD	HT_CAN	HT_NOCAR	PM	IR	POF	ACI	EU_TER	EU_FRE	EU_MAR	RU_EN	RU_MIN	WU	LU	ECOTOX	SC
	kg CFC-11 eq	CTUh	CTUh	Disease incidences	kBq U-235 eq.	mol H+ eq	mol N eq	kg N eq.	kg P eq.	kg N eq	MJ	kg SB eq	m3 water eq	-	CTUe	EUR
REU	-4.8E-05	-4.5E-07	-1.4E-05	-1.5E-03	-1.8E+01	-1.6E+00	-2.8E+00	-6.1E+00	-1.8E-02	-5.0E-01	-8.7E+03	-2.0E-02	-2.0E+02	-7.1E+03	-1.6E+04	-3.8E+02
REC-CEM	-6.4E-07	-2.0E-08	-2.1E-06	-1.7E-06	-6.2E-01	-4.9E-01	-5.0E-01	-2.1E+00	-5.1E-03	-1.7E-01	-2.9E+02	-1.6E-04	-3.0E+01	5.7E+01	-2.9E+03	3.2E+01
REC-RA	3.7E-06	4.3E-09	1.4E-07	2.8E-06	1.2E+00	1.3E-01	1.2E-01	4.5E-01	1.4E-04	4.1E-02	2.8E+02	1.9E-05	-4.5E+01	8.9E+01	1.7E+02	7.1E+01
LAN	3.64E-06	6.29E-09	1.44E-07	2.95E-06	1.18E+00	1.35E-01	1.13E-01	4.71E-01	1.22E-04	4.29E-02	2.58E+02	3.51E-05	1.15E+00	1.61E+02	1.76E+02	3.7E+01

Table A9. Remaining environmental impact categories for glass wool

Management option	OD	HT_CAN	HT_NOCAR	PM	IR	POF	ACI	EU_TER	EU_FRE	EU_MAR	RU_EN	RU_MIN	WU	LU	ECOTOX	SC
	kg CFC-11 eq	CTUh	CTUh	Disease incidences	kBq U-235 eq.	mol H+ eq	mol N eq	kg N eq.	kg P eq.	kg N eq	MJ	kg SB eq	m3 water eq	-	CTUe	EUR
REC-GLW	3.6E-06	6.1E-09	1.5E-07	1.9E-06	1.3E+00	6.3E-02	5.6E-02	1.4E-01	1.5E-04	1.8E-02	2.7E+02	4.1E-05	6.0E-01	1.5E+02	-1.9E+03	7.1E+01
LAN	3.28E-06	5.54E-09	1.30E-07	2.50E-06	9.42E-01	1.11E-01	9.14E-02	3.84E-01	8.34E-05	3.49E-02	2.15E+02	3.07E-05	7.42E-01	1.48E+02	1.53E+02	2.9E+01

Table A10. Remaining environmental impact categories for stone wool

Management option	OD	HT_CAN	HT_NOCAR	PM	IR	POF	ACI	EU_TER	EU_FRE	EU_MAR	RU_EN	RU_MIN	WU	LU	ECOTOX	SC
	kg CFC-11 eq	CTUh	CTUh	Disease incidences	kBq U-235 eq.	mol H+ eq	mol N eq	kg N eq.	kg P eq.	kg N eq	MJ	kg SB eq	m3 water eq	-	CTUe	EUR
REC-STW	3.5E-06	3.4E-09	1.5E-07	2.1E-06	1.0E+00	9.2E-02	8.5E-02	3.1E-01	1.1E-04	2.8E-02	2.6E+02	1.7E-05	-5.4E+01	8.8E+01	1.6E+02	7.3E+01
LAN	3.28E-06	5.54E-09	1.30E-07	2.50E-06	9.42E-01	1.11E-01	9.14E-02	3.84E-01	8.34E-05	3.49E-02	2.15E+02	3.07E-05	7.42E-01	1.48E+02	1.53E+02	2.9E+01

Table A11. Remaining environmental impact categories for bricks

Management option	OD	HT_CAN	HT_NOCAR	PM	IR	POF	ACI	EU_TER	EU_FRE	EU_MAR	RU_EN	RU_MIN	WU	LU	ECOTOX	SC
	kg CFC-11 eq	CTUh	CTUh	Disease incidences	kBq U-235 eq.	mol H+ eq	mol N eq	kg N eq.	kg P eq.	kg N eq	MJ	kg SB eq	m3 water eq	-	CTUe	EUR
REU	-1.8E-05	-2.5E-07	-1.5E-06	-8.4E-06	-4.5E+00	-6.2E-01	-6.4E-01	-2.0E+00	-2.5E-03	-1.8E-01	-2.2E+03	-1.1E-03	-3.4E+01	-1.1E+03	-1.9E+03	-3.7E+02
REC-CEM	-1.4E-06	-2.4E-08	-2.4E-06	-2.3E-06	-1.6E+00	-5.6E-01	-5.9E-01	-2.4E+00	-5.7E-03	-2.0E-01	-4.7E+02	-2.0E-04	-3.5E+01	-1.3E+01	-3.3E+03	2.1E+01
REC-CON	-1.5E-05	-2.1E-07	-1.1E-06	-7.1E-06	-6.1E-01	-5.0E-01	-4.4E-01	-1.6E+00	-1.5E-03	-1.4E-01	-1.4E+03	-8.4E-04	-4.9E+00	-6.9E+02	-1.4E+03	5.7E+01
REC-RA	3.4E-06	4.5E-09	1.5E-07	2.3E-06	1.9E+00	1.2E-01	1.3E-01	4.0E-01	3.0E-04	3.7E-02	3.7E+02	3.6E-05	-4.3E+01	1.4E+02	2.1E+02	7.5E+01
LAN	3.28E-06	5.54E-09	1.30E-07	2.50E-06	9.42E-01	1.11E-01	9.14E-02	3.84E-01	8.34E-05	3.49E-02	2.15E+02	3.07E-05	7.42E-01	1.48E+02	1.53E+02	2.9E+01

Table A12. Remaining environmental impact categories for glass

Management option	OD	HT_CAN	HT_NOCAR	PM	IR	POF	ACI	EU_TER	EU_FRE	EU_MAR	RU_EN	RU_MIN	WU	LU	ECOTOX	SC
	kg CFC-11 eq	CTUh	CTUh	Disease incidences	kBq U-235 eq.	mol H+ eq	mol N eq	kg N eq.	kg P eq.	kg N eq	MJ	kg SB eq	m3 water eq	-	CTUe	EUR
REU	-1.4E-04	-3.5E-07	-8.1E-06	-1.3E-04	-4.3E+01	-5.8E+00	-1.2E+01	-2.4E+01	-2.4E-02	-2.0E+00	-1.5E+04	-1.6E-02	-3.2E+02	-9.9E+03	-2.6E+04	-5.8E+02
REC-GLA	-1.02E-04	1.55E-07	2.07E-05	-9.23E-06	-2.50E+01	-1.6E+00	-3.9E+00	-8.4E+00	-1.06E-02	-5.41E-01	-8.1E+03	-1.21E-02	-1.9E+02	-7.4E+03	-1.8E+04	1.2E+01
REC-RA	4.2E-06	5.6E-09	1.8E-07	2.9E-06	1.9E+00	1.4E-01	1.4E-01	4.7E-01	2.9E-04	4.3E-02	4.0E+02	4.0E-05	-4.6E+01	1.5E+02	2.4E+02	7.6E+01
LAN	3.28E-06	5.54E-09	1.30E-07	2.50E-06	9.42E-01	1.11E-01	9.14E-02	3.84E-01	8.34E-05	3.49E-02	2.15E+02	3.07E-05	7.42E-01	1.48E+02	1.53E+02	2.9E+01

Table A13. Remaining environmental impact categories for excavated soils

Management option	OD	HT_CAN	HT_NOCAR	PM	IR	POF	ACI	EU_TER	EU_FRE	EU_MAR	RU_EN	RU_MIN	WU	LU	ECOTOX	SC
	kg CFC-11 eq	CTUh	CTUh	Disease incidences	kBq U-235 eq.	mol H+ eq	mol N eq	kg N eq.	kg P eq.	kg N eq	MJ	kg SB eq	m3 water eq	-	CTUe	EUR
REU-RCB	-1.2E-07	-9.7E-11	1.3E-09	-2.4E-07	-3.9E-01	1.0E-02	1.0E-02	5.0E-02	1.6E-05	4.7E-03	-2.5E+01	-2.1E-05	-3.9E+01	1.0E+01	1.0E+01	4.0E+00
REC-LIM	1.5E-06	-2.6E-08	-3.6E-07	-3.3E-06	-6.2E+00	-1.6E-01	-2.2E-01	-7.8E-01	-2.0E-02	-7.0E-02	-4.4E+02	-5.2E-04	-1.0E+02	3.9E+02	-7.4E+02	-1.9E+01
REC-CEM	4.0E-08	1.3E-08	5.0E-07	5.6E-07	2.9E+00	9.0E-02	1.3E-01	3.8E-01	1.0E-02	3.0E-02	2.6E+02	-2.2E-04	6.5E+02	-1.8E+02	3.9E+02	-3.7E+00
REC-IND	-7.0E-07	-1.8E-08	-1.0E-07	-9.5E-07	3.4E+00	-1.0E-02	2.0E-02	-5.0E-02	6.2E-04	-2.6E-03	8.2E+01	-2.6E-04	-1.6E+01	1.1E+01	-6.5E+01	7.6E-01
LAN	3.3E-07	4.2E-09	4.5E-08	1.4E-07	2.4E-01	6.0E-02	6.0E-02	2.5E-01	7.0E-04	2.0E-02	0.0E+00	0.0E+00	7.6E+00	3.8E+02	1.0E+02	5.7E+01

Table A14. Remaining environmental impact categories for dredging spoils

Management option	OD	HT_CAN	HT_NOCAR	PM	IR	POF	ACI	EU_TER	EU_FRE	EU_MAR	RU_EN	RU_MIN	WU	LU	ECOTOX	SC
	kg CFC-11 eq	CTUh	CTUh	Disease incidences	kBq U-235 eq.	mol H+ eq	mol N eq	kg N eq.	kg P eq.	kg N eq	MJ	kg SB eq	m3 water eq	-	CTUe	EUR
REU	3.1E-07	-2.2E-10	5.2E-09	3.3E-07	-3.5E-01	3.0E-02	2.0E-02	1.0E-01	-3.9E-05	1.0E-02	1.5E+00	-2.3E-05	-4.5E+01	1.7E+00	1.6E+01	-5.3E+00
REC-RCB	1.1E-06	8.6E-09	1.4E-07	1.4E-06	1.0E+00	8.0E-02	1.5E-01	4.8E-01	4.2E-03	8.0E-02	4.9E+02	1.5E-04	-3.5E+01	8.2E+01	5.1E+02	6.9E-01
REC-CEM	-1.7E-06	9.6E-09	-3.6E-07	-3.4E-07	-5.6E+00	-1.3E-01	-1.1E-01	-4.2E-01	-1.0E-02	-3.0E-03	1.3E+01	2.6E-04	-9.3E+01	-4.0E+02	-2.5E+02	2.9E+01
REC-IND	4.1E-07	-1.5E-08	3.1E-09	5.5E-07	6.0E+00	6.0E-02	1.6E-01	3.7E-01	4.9E-03	7.0E-02	6.4E+02	-1.6E-04	6.7E+00	8.9E+01	4.1E+02	9.1E+00
LAN	1.6E-06	1.3E-08	1.9E-07	1.8E-06	1.7E+00	1.4E-01	2.0E-01	7.0E-01	4.9E-03	1.0E-01	5.2E+02	1.8E-04	1.8E+01	4.6E+02	6.1E+02	6.1E+01

Annex 2. Sensitivity results for wood fraction including the cascading use and greener energy use

Table A15. Data and results of the sensitivity analysis for wood fraction including four life cycles in the cascading use with current energy mix (2020). Impacts on Climate Change.

Cascading calculation - Current energy mix					
Particleboard/waste input	%		59	56	54
		1st life	2nd life	3rd life	4th life
Recycling	kg	1000	592	332	180
Incineration	kg	1000	592	332	180

		1st life	2nd life	3rd life	4th life	Total 2 cycles	Total 4 cycles
Reuse	kg CO_2-eq.	-52	-52	-52	-52	-103	-206
Recycling	kg CO_2-eq.	-69	-41	-23	-12	-110	-145
Incineration	kg CO_2-eq.	-265	-157	-149	-144	-423	-716
Production	kg CO_2-eq.		374	210	114	374	697
Incineration + Production	kg CO_2-eq.	-265	217	61	-30	-49	-18

Virgin Production impact	kg CO_2-eq./t particleboard	631

Source: Own elaboration

Table A16. Data and results of the sensitivity analysis for wood fraction including four life cycles in the cascading use with future energy mix (2050). Impacts on Climate Change.

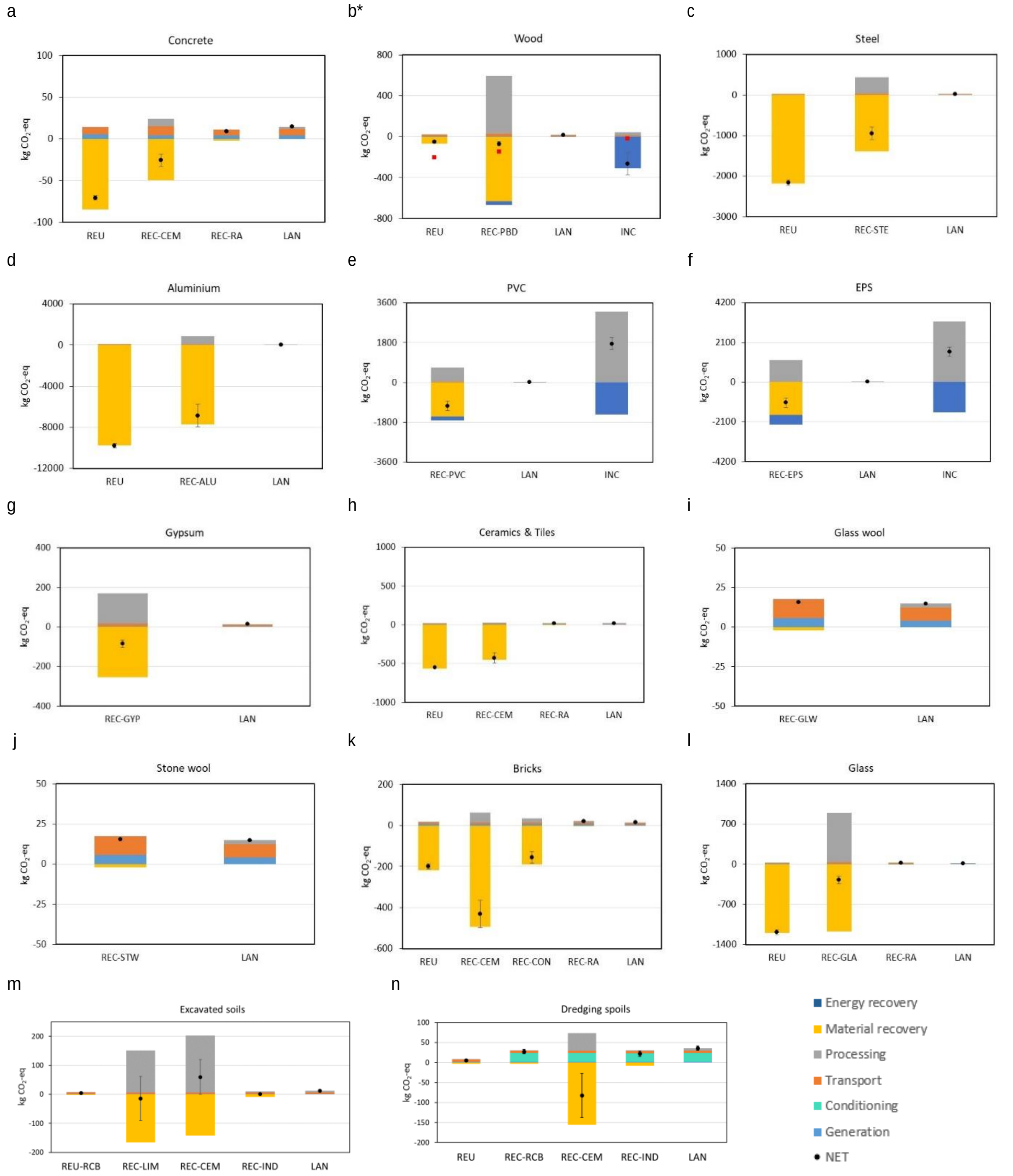
Cascading calculation - Future energy mix					
Particleboard/waste input	%		59	56	54
		1st life	2nd life	3rd life	4th life
Recycling	kg	1000	592	332	180
Incineration	kg	1000	592	332	180

		1st life	2nd life	3rd life	4th life	Total 2 cycles	Total 4 cycles
Reuse	kg CO_2-eq.	-52	-52	-52	-52	-103	-206
Recycling	kg CO_2-eq.	-69	-41	-23	-12	-110	-145
Incineration	kg CO_2-eq.	-100	-59	-33	-18	-159	-211
Production	kg CO_2-eq.		374	210	114	374	697
Incineration + Production	kg CO_2-eq.	-100	315	176	96	215	487

Virgin Production impact	kg CO_2-eq./t particleboard	631

Source: Own elaboration

Figure A1. Characterised Climate Change results per tonne of CDW fraction managed with breakdown of the contributions in 2050. Values above zero represent burdens, while values below zero represent savings. The final net impact, per each individual category, is the sum of burdens and savings and is represented with a black dot. The error bars represent the standard deviation around the net result. For the abbreviations used please refer to Table 11.



*Note that for wood, the red square (■) indicates the result for the cascading calculation for four cycles (sensitivity analysis).

Source: Own elaboration

Annex 3. Share quantities within the scenarios of management pathways in practice according to the calculation rules from the EC

Table A17. Partitioning of the generated CDW across management pathways in the two scenarios analysed (MRP and MPP). The percentage reflects the share of the CDW generated reused or recycled in practice in line with the updated calculation rules proposed by the European Commission. Note that losses in the recycling process from WOD, PVC and EPS would go to incineration, losses from STE, ALU, GLW and STW would go to landfill, and losses from C&T, BRK, GLA, ESR and DDS would go to backfilling.

	Baseline (BSL)					Maximum Recycling Potential (MRP)					Maximum Preparing for Reuse Potential (MPP)				
	Management pathways %					Management pathways %					Management pathways %				
Fraction	REU	REC	RBB	INC	LAN	REU	REC	RBB	INC	LAN	REU	REC	RBB	INC	LAN
CON	0	79	10	0	11	0	100	0	0	0	13	87	0	0	0
WOD	0	25	0	69	6	0	37	0	63	0	25	16	0	59	0
STE	10	70	0	0	20	10	75	0	0	15	29	59	0	0	12
ALU	10	77	0	0	13	10	82	0	0	8	50	45	0	0	5
PVC	0	26	0	16	58	0	78	0	22	0	0	78	0	22	0
EPS	0	7	0	69	24	0	19	0	81	0	0	19	0	81	0
GYP	0	10	0	0	90	0	94	0	0	6	0	94	0	0	6
C&T	0	73	16	0	11	0	92	8	0	0	7	83	10	0	0
GLW	0	2	0	0	98	0	80	0	0	20	0	80	0	0	20
STW	0	2	0	0	98	0	80	0	0	20	0	80	0	0	20
BRK	0	73	16	0	11	0	93	7	0	0	40	38	22	0	0
GLA	0	6	24	0	70	0	97	3	0	0	20	78	2	0	0
ESR	0	34	41	0	25	0	97	3	0	0	100	0	0	0	0
DDS	0	8	4	0	88	0	97	3	0	0	100	0	0	0	0
Total*	0	36	30	1	30	0	93	3	1	0	83	13	0	0	0
Total (exc. ESR & DDS)*	1	59	9	3	13	1	79	1	3	1	14	65	2	2	1

*Total calculated as weighted average based on the share of each individual material fraction in the total CDW. Note that the partitioning between the management pathways does not sum to 100% because there are some fractions of CDW that have been excluded from the analysis (mainly mixed inert, representing ca. 14%, but also minor fractions such as cardboard, electronics, paint and glue that represent ca. 1% of the total CDW).

Note - CON: concrete; ALU: aluminium; BRK: bricks; C&T: ceramic and tiles; DDS: dredging spoils; EPS: expanded polystyrene; ESR: excavated soil; GLA: glass; GLW: glass wool; GYP: gypsum; INC: incineration; LN: landfilling; PVC: poly vinyl chloride; REC: recycling; RBB: recovery/backfilling. STE: steel; STW: stone wool; WOD: wood.

Source: Own elaboration

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