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Cost–Benefit Analysis for End-of-Life Scenarios: A Case Study of an Electric Moped

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Abstract

This study presents an economic analysis of different end-of-life (EoL) strategies for light electric vehicles (LEVs). Utilising the case of a shared electric moped scooter, a cost–benefit analysis evaluates the profitability of three EoL scenarios. These scenarios encompass different combinations of R-strategies (reuse, repurpose, recycle), all of which have been shown to offer ecological saving potential in previous research. The net present value (NPV) of the current EoL treatment in Germany amounts to EUR 75.81 per e-moped, while alternative treatments which focus on repurposing the battery and increasing the number of components for reuse account for EUR 300.87 and EUR 379.01 per e-moped, respectively. In addition to providing in-depth insights into key cost factors (e.g., labour costs for disassembly) and benefits (e.g., sale of used components), this study includes sensitivity analyses. The scenarios differ in their sensitivity to changes in disassembly labour costs, spare parts revenue, and the social cost of carbon. Among all tested parameters, variations in the sale prices of components destined for reuse in the current EoL treatment scenario exhibit the highest influence on NPV, with a sensitivity coefficient of 1.43. Overall, component reuse emerges as a profitable EoL strategy, while battery repurposing appears promising for the future. Assuming a generally positive ecological impact of circular product systems for LEVs, this study also confirms their economic viability. From both economic and environmental perspectives, the findings of this study serve as a valuable catalyst for advancing circular product design, thereby facilitating the implementation of EoL strategies for LEVs.

Keywords: cost–benefit analysis; circular design; end-of-life; r-strategies; light electric vehicles

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1. Introduction

To address the increasing scarcity of resources and to contribute to achieving global climate protection goals by reducing waste and emissions, a shift towards a Circular Economy (CE) is needed [1]. The CE stands in opposition to a linear economy and aligns with Sustainable Development Goal 12 (Sustainable Consumption and Production) of the United Nations. The way products, services, systems, and infrastructures are designed has been recognized as a catalyst to move away from linear economy models. Without a systemic change, in the way we design them, the potential of the CE will not be reached [2]. This systemic design approach is known as Circular Design (CD) [2]. The objectives of CD further encompass the utilization of renewable energy sources, the avoidance of toxic

chemicals and materials that impede potential product reuse, and the elimination of waste. These aims are pursued through the deliberate selection and design of materials, products, systems, and business models in support of a CE [1,2]. Key implementation measures of the concepts of CE and CD are the so-called R-strategies. R-strategies such as *refuse*, *rethink*, *reduce* aim to narrow the loop at the production phase by reducing the input and/or output per unit or the units per use. Slowing the loop occurs mainly within the use phase, where R-strategies such as *reuse*, *repair*, *refurbish*, *remanufacture*, *repurpose* tackle this by extending the lifespan of the product and its components. *Recycle* and *recover* aim to close the loop at the EoL phase increasing the circulation rate [3].

The implementation of such strategies is crucial to reduce global greenhouse gas (GHG) emissions in sectors with high GHG emissions contributions [3]. Urban transportation in the EU for instance, accounts for 23% of all European transport GHG emissions [4]. To reduce environmental emissions, robust regulations, financial incentives, and significant investments in infrastructure supporting low- and zero-emission vehicle use are essential [5]. In general, LEVs are anticipated to play a central role in de-carbonizing the transport sector, which is a significant contributor to global GHG production [5]. Specifically, they are promising for establishing environmentally friendly micromobility solutions [6–9]. While the potential of electric vehicles (EVs) and LEVs to reduce GHG emissions compared to combustion-powered vehicles has been demonstrated by numerous LCA studies [10–14], most research examines both the manufacturing and usage phases in detail, including different charging scenarios (e.g., source of energy, ambient conditions, lifespan, etc.). Nevertheless, different EoL strategies can indirectly influence the production and use phases of product service systems already shown on different product service systems, such as coffee machines, batteries or smartphones [9,15,16]. With the global market for EVs continuously growing [8] and the market for micromobility and LEVs also expanding [17], it is crucial to establish and implement well-defined, innovative EoL strategies to ensure economically and environmentally sustainable integration and management of sustainable mobility.

Despite the increasing relevance of CE approaches in the context of sustainable mobility, significant research gaps remain regarding the environmental and economic evaluation of EoL scenarios for LEVs. Recent efforts include both ecological and economic studies; nevertheless, these primarily focus on individual components, on isolated process steps, or assess EoL strategies in a more boarder sense. For example, the economic feasibility of establishing dedicated LIB recycling facilities has been investigated in Brazil [18], while other studies have examined the technical, ecological, and economic potential of stationary reuse through battery repurposing [19]. Further analyses concentrated on downstream applications of retired EV batteries in Germany, addressing dismantling processes but limiting the scope to the battery alone [20]. Additional research has characterised EoL strategies within the CE framework [21], though often restricted to single aspects such as glass recovery in France [22], manual dismantling in Romania [23], or the recycling of plastic components in the EU [24].

In-depth evaluations of LEVs in their EoL phase considering multiple EoL strategies have rarely been comprehensively carried out. Notably, existing studies also tend to exclude the explicit monetization of ecological benefits—such as avoided CO_2 emissions or reduced primary energy consumption—thereby limiting their ability to fully capture the sustainability impacts within economic decision-making frameworks. This study addresses those gaps by explicitly integrating monetized ecological benefits into the economic evaluation of EoL strategies.

From a comprehensive point of view, the concept of sustainability cannot be reduced to the ecological dimension but includes social and economic dimensions as well [25]. In this context, this study expands on the ecological LCA of EoL scenarios for LEVs by [26,27],

who evaluated different R-strategies, by incorporating an economic assessment of the same scenarios. Based on their case study of a shared e-moped, the authors pointed out that component reuse, recycling materials, and, in particular, repurposing batteries for stationary energy storage solutions have the potential to reduce global warming potential (GWP_{100}) significantly. With this result, they contribute to the discourse concerning the implementation of EoL strategies in the sector of LEVs. Building on this ecological assessment, we now address the following questions with a focus on economic analyses: RQ1: To what extent are EoL alternatives that demonstrate environmental benefits also economically feasible for a shared e-moped treated in Germany? RQ2: How can different EoL scenarios for LEVs be assessed in terms of a cost–benefit analysis? By answering these questions, we, firstly, contribute to the scientific discourse concerning the circularity of LEVs and, secondly, provide relevant information for strategical–economic considerations of relevant stakeholders in a CE of LEVs.

1.1. Definitions of Concepts

This section provides definitions of central terms and concepts.

According to [28], the EoL of a product can be understood as the point in its life cycle, where the product is no longer used or needed by the initial user. In terms of waste prevention, different EoL strategies can be applied to a product that has reached its EoL [29].

This study employs the R-strategies outlined by Potting et al., who distinguish nine strategies ranging from Refuse (R0) to Recover (R9), which are defined in Table 1, as seen in [3]. In general, the application of R-strategies increases the circularity of product systems and enables the transition from a linear to a circular economy [3]. However, it is important to note that Potting's classification ranks the strategies according to their application priority. This means that the R0 strategy should, whenever possible, be implemented before the R1 strategy. Consequently, the Recover (R9) strategy should be applied only after all other R-strategies have been considered, as it represents one of the least preferred options in terms of circularity.

Table 1. Definition of R-strategies according to [3].

R-Strategy	Definition
Refuse (R0)	Make product redundant by abandoning its function or by offering the same function with a radically different product.
Rethink (R1)	Make product use more intensive (e.g., through sharing products or by putting multi-functional products on the market).
Reduce (R2)	Increase efficiency in product manufacture or use by consuming fewer natural resources and materials.
Reuse (R3)	Reuse of a discarded product that remains in good condition and fulfils its original function.
Repair (R4)	Repair and maintain a defective product to be used for its original function.
Refurbish (R5)	Restore an old product and update it.
Remanufacture (R6)	Use parts of a discarded product in a new product with the same function.
Repurpose (R7)	Use a discarded product or part of it in a new product with a different function.
Recycle (R8)	Process materials to obtain the same (high-grade) or lower (low-grade) quality.
Recover (R9)	Incinerate materials with energy recovery.

Additionally, regarding the definition of central terms in the context of CE and EoL strategies, defining cost–benefit analysis (CBA) is necessary for this study. Referring to [30], a CBA can be described as an assessment method to quantify in monetary terms the total value of a policy or a project, including all consequences to all members of society. Compared to a revenue–cost analysis or a life cycle costing assessment, a CBA includes market and non-market values, such as life quality, health benefits, or environmental emissions [30,31]. The methodological steps of conducting a CBA are presented in Section 2.

1.2. Case Study

This paper applies an economic assessment to current and alternative EoL scenarios for a shared e-moped in Germany, building on the case framework described in [26,27]. In order to provide a basis for a comparative assessment of the ecological and economic aspects, the system boundaries, the functional unit (FU), as well as all assumptions and scenarios have been kept consistent. The FU considered is one shared e-moped with a lifetime driving distance of 50,000 km [10]. The study focuses on the Kumpan 1954Ri model, with technical data provided by the OEM e-bility GmbH (Remagen, Germany). In line with EU regulations, e-mopeds are defined as two-wheeled motor vehicles with a maximum design speed of 45 km/h and a rated power output of up to 4 kW for electric motors [32]. Shared e-mopeds operate through smartphone-based rental systems, enabling customers to access vehicles for short-term use [10]. According to empirical usage data, the average operation consists of 4.9 km per ride, 3.7 trips per day, a daily driving distance of 18.1 km, and an average ride time of 16.7 min per trip [10]. These operational parameters are economically significant as they directly determine maintenance frequency, battery handling, and component replacements.

A key modelling distinction between shared and privately owned e-mopeds lies in the treatment of battery replacements. For the 50,000 km lifespan of a shared e-moped, 1.25 batteries are required [10]. This fractional value is feasible in a sharing context, since impacts and benefits arising from the remaining 0.75 battery can be allocated to another vehicle within the fleet [27]. In contrast, for a privately owned e-moped, two full batteries would need to be modelled, leading to higher costs. Moreover, the collection phase of the analysed e-moped was modelled based on the practices of German sharing operators, while data from privately owned vehicles were not considered, as outlined in [27].

The assessed e-moped is legally classified under the L1e-B category, subject to EU type-approval requirements [33]. Unlike smaller LEVs such as e-scooters and e-bikes, it does not fall under the EU WEEE Directive [34] or the German Electrical and Electronic Equipment Act [35]. In Germany, EoL vehicles are typically processed by certified dismantling companies, which remove selected components for reuse or recycling before sending the remaining structure to a shredding facility [36,37]. Although no specific EoL regulation exists for e-mopeds at German or EU level [36,38,39], German e-moped sharing operator [40], which formerly used the e-moped model analysed in this study, applied practices aligned with broader EoL vehicle regulations. The absence of dedicated regulation highlights the need to identify economically viable treatment pathways, making this case especially relevant for exploring the integration of circular economy strategies in shared LEV systems. Table 2 shows the different treatment pathways for the vehicle and the battery in weight percentage according to different regulations in comparison to the assessed scenarios of this study. In the case of treatment through recycling, the percentages given in Table 2 indicate how much weight in percent was selected to be recycled and not the percentage of effectively recycled material after EoL.

Table 2. Overview of treatment pathways per weight percentage according to different regulations and defined scenarios.

	EoL Vehicles [36,39]	**EoL Electrical/Electronic Products [34,35]**	**EoL Batteries [41,42]**	**Scenario 1 [26,27]**	**Scenario 2 [26,27]**	**Scenario 3 [26,27]**
Scope	EU [39]; Germany [36]; EoL vehicles: categories M1 and N1; excludes L1e-B category.	EU [34]; Germany [35]; all electrical and electronic equipment; excludes vehicles that require EU type-approval.	EU [42]; Germany [41]; all batteries and accumulators, including portable, automotive, and industrial types.	Germany; e-moped Kumpan 1954Ri (L1e-B category).		
Treatment specifications						
Battery	Batteries must be removed prior to treatment and managed in accordance with the German Battery Act [41], which implements the EU Battery Regulation [42].		Landfill or incineration of batteries is prohibited; all collected batteries must be sent for recycling [41,42].	Treatment according to German Battery Act [41].	75% of the battery's weight is sent for recycling; 25% of the battery's weight is reused [26,27].	67% of the battery's weight is sent for recycling; 33% of the battery's weight is repurposed [26,27].
Vehicle	At least 85% of the vehicle's weight must be reused or recycled, 95% recovered overall (incl. energy recovery), and up to 5% may be disposed of, such as by landfill [36,39].	At least 80% of product's weight reused or recycled, 85% total recovery (incl. energy recovery), and up to 15% may be disposed of, such as by landfill [34,35] [1].	N/A [2]	4.3% of vehicle's weight reused, 81.1% recycled, 10.3% energy recovered (incineration), and 4.3% disposed of (landfill) [26,27].	18.8% of vehicle's weight reused, 66.6% recycled, 10.3% energy recovered (incineration), and 4.3% disposed of (landfill) [26,27].	4.3% of vehicle's weight reused, 81.1% recycled, 10.3% energy recovered (incineration), and 4.3% disposed of (landfill) [26,27].

[1] Treatment regulations for product category 4: large electrical equipment, ≥50 cm; [2] N/A = not available.

In line with [27], this study excludes R-strategies R0–R2, as they are primarily associated with the product's manufacturing and use phases rather than the EoL stage. Strategies R3–R6, by contrast, focus on the reuse of product components to varying extents while still maintaining their original function [3]. Following the circularity hierarchy outlined in [3], Strategy R3—also adopted in [3]—is applied here. To ensure alignment with the EU Battery Regulation [42,43], Strategy R7 is likewise implemented in both [27] and this work. Finally, R-strategies R8 and R9, which reflect standard practices in many existing EoL treatments, are also included in the analysis. Figure 1 gives a comprehensive overview of the R-strategies applied in this case study, based on [3].

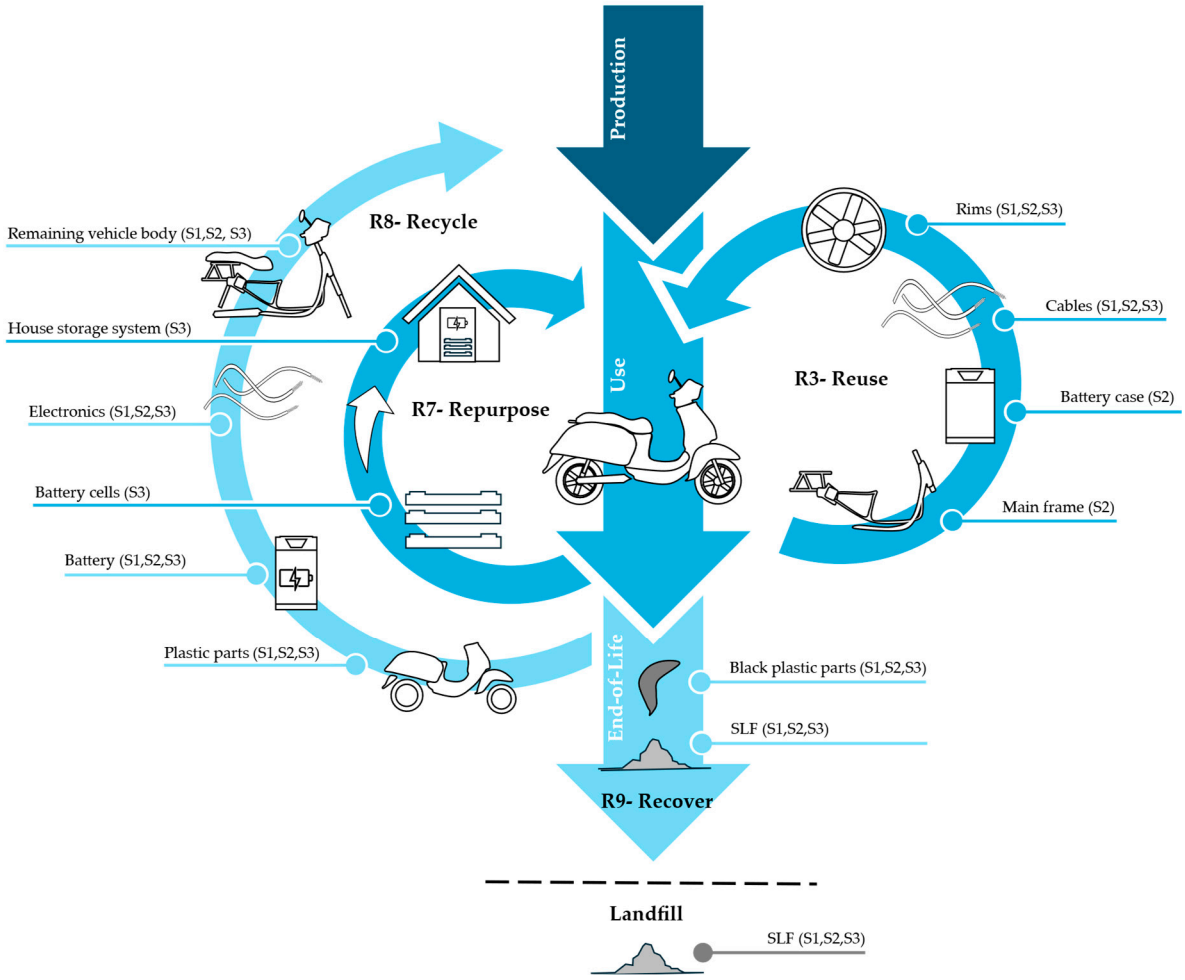


Figure 1. Framework of applied R-strategies, based on [3], for the case study examining end-of-life treatment scenarios for e-mopeds, in accordance with [27]. S1 = Scenario 1 (current treatment); S2 = Scenario 2 (component reuse); S3 = (battery repurpose); SLF = shredder light fraction.

2. Materials and Methods

The investigation of the economic dimension of different EoL scenarios, as defined by [27], were evaluated in this study through a CBA. This section describes all implemented stages of the CBA, first in a general sense and subsequently in detail.

The CBA of this work is structured according to the ten-step guidelines established by [30].

- Step 1: Explain the purpose of the CBA
- Step 2: Specify the alternative projects
- Step 3: Specify the standing
- Step 4: Identify and characterize impact categories
- Step 5: Predict impacts quantitatively over the project's life
- Step 6: Monetize all impacts
- Step 7: Discount benefits and costs
- Step 8: Compute the net present value of each alternative
- Step 9: Perform sensitivity analysis
- Step 10: Make a recommendation

The first step is to clearly define the objective of the analysis. This involves specifying the decision context and determining the goal, such as assessing the feasibility, comparing alternatives, or prioritizing investments. This is followed by the identification of mutu-

ally exclusive or independent alternatives that must be assessed, as well as the decision regarding which individuals, groups, or regions should be included when measuring and valuing costs and benefits. In the next step, all relevant impact categories associated with each project alternative, including direct, indirect, tangible, and intangible effects must be identified. For each category, appropriate metrics should be selected to enable quantification and comparison. Based on this and with the help of empirical data and models, future costs and benefits in these categories are predicted and expressed in monetary terms using valuation methods. Future costs and benefits must be converted into present values using an appropriate discount rate, thus allowing consistent comparison. Building on the previous steps, the benefit and cost values are aggregated to calculate the net present value (NPV) for each alternative. A positive NPV indicates that the benefits exceed the costs, providing a basis for ranking options. In a sensitivity analysis, key assumptions can be varied to test the robustness of results, assessing uncertainty and identifying parameters that significantly influence outcomes. Finally, a recommendation is made regarding which alternative offers the greatest net benefit.

Following this general depiction of typical steps of a CBA according to [30], the application of this method for the purpose of this paper is presented in detail.

Step 1: The CBA of this study aims to assess the economic potential of the different EoL scenarios for shared e-mopeds in Germany, including the ecological assessment in [26,27], allowing policy-making recommendations based on an ecological and economical perspective. For this purpose, an ex ante CBA was conducted to assess the economic dimension of the status quo as well as different possible EoL scenarios for e-mopeds.

Step 2: Three distinct project alternatives have been identified for the treatment of e-mopeds in Germany based on the EoL scenarios, as defined in [26,27].

The first alternative, referred to as Scenario 1 (current treatment), represents the current standard practice for EoL treatment in accordance with EU and German regulations [36,41]. In this scenario, a minimum of 85% of the vehicle's total weight must be either reused or recycled, while 95% must undergo some form of final treatment, such as energy recovery, excluding landfill disposal [36]. The battery, on the other hand, has to be fully recycled, in line with European and German directives [41,42]. Components such as tires and large plastic parts are dismantled for recycling, while certain parts, like some cables or front rims, are reused for two life cycles. After disassembly, the remaining vehicle body (RVB) is sent to a shredding facility. There, the material is separated into steel, non-ferrous metals, and the shredder light fraction (SLF), the latter of which is partially incinerated and partially landfilled. Plastic sorting technology is used during this process, but due to technical limitations, black plastics are not properly detected and are subsequently incinerated [37]. The battery cells themselves undergo a combined pyrometallurgical and hydrometallurgical treatment, representing a common battery recycling pathway in Europe [44].

The second alternative, referred to as Scenario 2 (component reuse), expands upon Scenario 1 (current treatment) by increasing the number of vehicle and battery components that are reused. In this scenario, the e-moped's main structural frame is considered for reuse due to its high durability and relative protection from wear and tear. Additionally, all parts of the battery housing are assumed in [27] to be suitable for reuse for two life cycles, given their protected location inside the vehicle. The logistical and treatment processes remain consistent with those described in Scenario 1, but the transported mass and disassembly time is adjusted to account for the reused components. In total, the reused elements comprise approximately 19% of the RVB weight and 25% of the battery weight. This scenario represents a moderate shift toward circularity by extending the service life of selected components through reuse rather than immediate recycling or disposal.

The third alternative, identified as Scenario 3 (battery repurpose), focuses specifically on the battery, which is known to contribute significantly to the overall environmental burden of LEV production. In this project, while the vehicle is treated in the same manner as in Scenario 1, a new pathway is introduced for the battery cells. Instead of full recycling, 50% of the cells are recovered and repurposed for use in stationary energy storage applications, such as a home storage system. This repurposing is based on the understanding that battery cells typically reach EoL in automotive use when their state of health (SoH) falls to around 80%. While such batteries may no longer be suitable for use in vehicles, they can still provide functional storage capacity for less demanding, stationary applications. With an assumed SoH of 80% and a 50% recovery rate after [9,45], the 1.25 batteries can contribute 0.7485 kWh of repurposed capacity. This value is calculated by [27] scaling the original capacity per battery pack (1.497 kWh) to the necessary 1.25 packs per FU (1.87125 kWh), assuming 80% SoH (1.497 kWh) and 50% cell recovery rate (0.7485 kWh). This scenario significantly extends the utility of battery materials and exemplifies a high-circularity strategy by offsetting the need for virgin materials and reducing overall environmental impacts.

Step 3: For the decision regarding which individuals, groups, or regions should be included when measuring and valuing costs and benefits, the focus was placed on stakeholders, processes, and consequences directly relevant for the set of alternative EoL scenarios depicted in Step 2. The production and use phases are excluded from the analysis. Therefore, the costs and benefits of all production-related processes are not assessed, since they do not differ between the analysed alternatives. This encompasses, for instance, the exclusion of costs related to the extraction of raw materials for producing parts, financial benefits for the sharing company of the e-moped, user costs for the sharing service and emission savings when e-moped trips replace car journeys. Consequently, the boundaries of the analysis include only the EoL phase. Figure 2 illustrates the scope of the CBA, encompassing costs and benefits associated with dismantling, shredding, recycling, repurposing, incineration, and landfill processes. The analysis focuses on the relevant stakeholders, facilities, and individuals involved in these activities (e.g., shredding and recycling facilities, enterprises engaged in battery repurposing). Costs of treatment processes and benefits from recovered materials, energy, components, and repurposed energy storage are assessed primarily in the German context, incorporating EU-related data when necessary. In contrast, the ecological impacts of the different EoL scenarios are considered from a global perspective, capturing emissions and potential environmental damages. In general, costs and benefits of the processes, facilities, and individuals relevant for the different EoL scenarios are considered at a national level while the environmental impact is examined within a global dimension. It is important to emphasize that the CBA conducted in this study assesses the procedural costs and benefits of the EoL pathway as a whole. The analysis does not aim to evaluate the individual business models or financial performance of specific actors along the value chain, but rather to understand the systemic economic implications of the chosen EoL configuration.

Step 4: A key step in the CBA is to systematically identify all relevant impacts of a project or measure, categorize them into costs and benefits, and specify appropriate metrics for each impact category [30]. The costs and benefits were determined using a process modelling approach based on a preceding LCA [27], where all relevant processes were represented to track inputs and outputs, and economic values were assigned accordingly. In addition, impact categories were derived by reviewing categories used in other CBAs [15,18] and by considering the availability of relevant data for quantification. To evaluate the economic impact of the recycling process, operational expenditures (OpEx) were quantified across several categories. Capital expenditures (CapEx) were not considered directly; instead, only their effect through depreciation was taken into account, assuming

that the required infrastructure and facilities for EoL treatment are already established. Table 3 shows the identified impacts, their categorization in terms of costs and benefits, and their metrics. Of the cost factors, the transportation of goods is included, and its impact is dependent on the travelled distance and the carried load. Human labour is recorded in terms of working time, capturing the effort requirements of machine operators, dismantlers, and all staff involved in each process. Auxiliary material costs refer to the consumption of process-related substances (e.g., Ammonia, CaO), expressed by their mass. The impact related to the treatment of residues refers to the efforts involved in the treatment of by-products of a process such as sludge or wastewater, measured in mass. Process operation costs, such as those arising from shredding, separation, or recycling machinery, are calculated based on the consumption of operating resources, including process energy, and water use. Fixed costs comprise depreciation, insurance, general expenses, and maintenance, which occur independently of treatment volume.

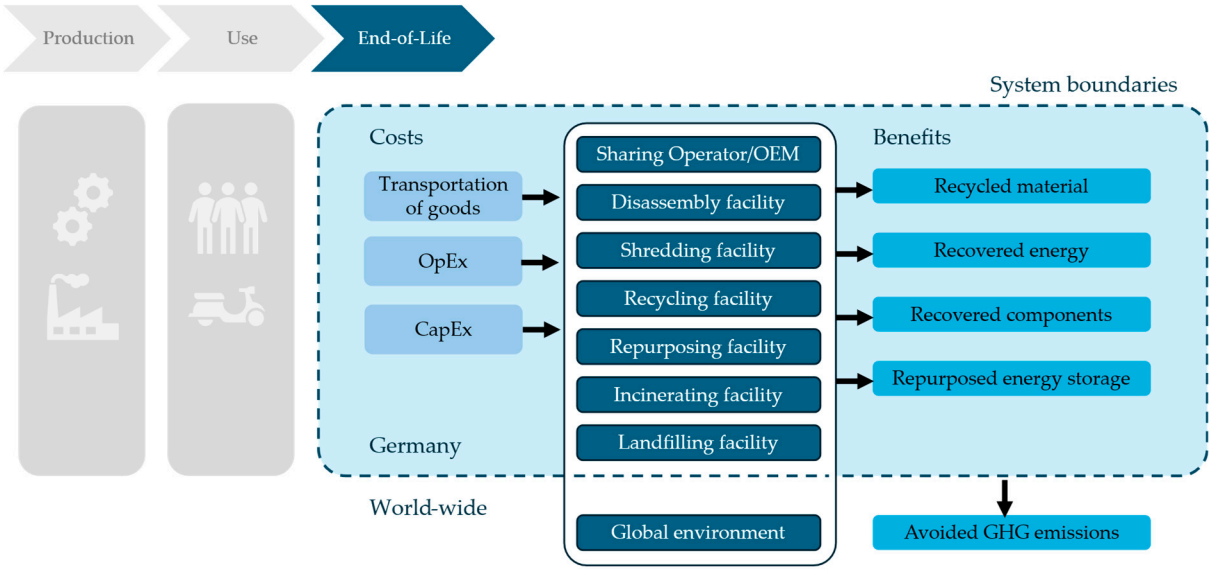


Figure 2. System boundaries of the assessed cost–benefit analysis. OpEx = operational expenses; CapEx = capital expenses.

Table 3. Overview of impacts, categories and their respective metrics. CapEx = capital expenses; OpEx = operational expenses.

Impact	Category	Metric
Human labour	Cost	Working time
Process operation	Cost	Energy, water use
Fixed costs	Cost	CapEx, fix OpEx
Goods transportation	Cost	Distance, load
Auxiliary material	Cost	Mass
Treatment residues	Cost	Mass
Avoided GHG emissions	Benefit	GWP_{100} (kg CO_2 eq)
Recycled materials	Benefit	Mass
Repurposed energy storage	Benefit	Storage capacity
Recovered energy	Benefit	Energy output
Recovered components	Benefit	Components

The environmental impact is categorized as a benefit since environmental credits arise from all EoL alternatives [27]. This impact is expressed as avoided GHG emissions and measured in kg of CO_2 equivalent. Furthermore, benefits arise from the recovery of

materials through recycling processes such as recycling of plastics and battery, and the shredding of the RVB, in all assessed scenarios, as quantified by mass. Additionally, benefits arising from excess energy production at incineration and landfill facilities are classified as belonging to the impact category of recovered energy and are quantified as energy output. Within all scenarios, some components are selected to be reused for another life cycle. This benefit is considered to be the recovery of components and is measured by the type and number of each component. Finally, additional benefits arise for Scenario 3 in the form of energy storage from a repurposed battery. This impact is measured after the amount of storage capacity provided by the repurposed stationary storage system.

Step 5: This step requires the prediction of all incremental costs and benefits for each year of the project's lifetime, so that impacts can be aggregated and compared over the course of the entire discount period [30]. In this analysis, this step is not applicable, as there are no recurring cash flows or impacts across multiple years. Instead, the assessment is conducted as a one-time accounting of costs and benefits, with the FU defined as one e-moped. Costs such as transportation, human labour for the processing steps, and auxiliary materials are considered to be one time per FU. Similarly, benefits from recovered materials, energy or components are accounted once, as these benefits arise once per FU. In the repurposing scenario, revenues from the sale of the repurposed battery system are therefore considered only once, as financial returns occur solely at the point of sale rather than through continued operation over subsequent years. Depreciation costs, although inherently periodic, are normalized on a per-ton basis based on treatment capacities and considered as a one-time value for the FU.

Step 6: Finding a valuation method for each metric (monetizing) was implemented using publicly available data, data from expert interviews, and best practices from the literature. When necessary, data were complemented with reasonable assumptions shown in Table 4. Since data availability varied in terms of temporal representation, the year 2024 was defined as a reference year for this study. All data originating from other years were adjusted on inflation using currency-specific consumer price indexes (CPIs) (Table A2). This measure is applicable mainly for fixed costs or CapEx costs; nevertheless, some impact categories rely on consumable goods, which are subject to socio-economical price changes in addition to inflation. In these cases, for example, when calculating costs implicating energy, water, or other auxiliary materials, costs were modified using average prices for consumable goods for 2024 (Table A2). When data referred to United States dollars (USD), values were first adjusted to 2024 USD and then represented in EUR using the average exchange rate for the reference year given by the European Central Bank [46]. This section addresses first the monetization of costs and subsequently the monetization of benefits.

To account for all costs of the different EoL scenarios, the costs of all involved treatment facilities were modelled. Some process costs of the different treatment facilities depend on the treatment capacity of the facility and its necessary labour load. Table 5 shows the facility characterization of the implemented economic model.

All impact costs of the disassembly and battery recycling facilities were obtained from primary data from a German vehicle and battery disassembly facility with experience treating the e-moped model of this study. Operative costs were therefore not theoretically modelled but represent real world data of the facilities in 2025. To adjust the values for the reference year, a deflation rate of 0.983, as provided in [49] was used. For the disassembly process, the interviewed facility assumed a required disassembly time of one hour for Scenario 1 (current treatment) and Scenario 3 (battery repurpose), and two hours for Scenario 2 (component reuse). The corresponding time demand was monetized using the reported disassembly cost rate of EUR 59.90 per hour, as obtained from the interviewed facility. In addition, costs arising from the receipt of goods, initial inspection,

goods classification, insurance, and administrative expenses were jointly monetized with a cost rate provided by the disassembly facility of EUR 35 per e-moped. The costs of the battery recycling process were monetized based on the weight of the batteries to be treated, as well as the battery chemistry. For a nickel–manganese–cobalt (NMC) battery an aggregated cost rate of EUR 2.20 per kilogram for a German battery recycling facility was provided by the interviewed disassembly facility was used. This rate encompasses all costs associated with the battery recycling process for a NMC battery cell composition.

Table 4. Assumptions made in the process of monetizing the respective impacts.

Impact	Assumption
Human labour	• Average yearly salary [1] = EUR 52,500
Process operation	• For incineration and landfill facility internal electricity use is included in net electricity generation; no separate electricity demand is modelled
Fixed costs	• General expenses = 10% of total costs [47]
Goods transportation	• Intracity average driving velocity = 20 km/h· • Intercity average driving velocity = 80 km/h· • Loading time = 0.5 h/process· • Unloading time = 0.5 h/process
Recovered material	• Cathode metal sulphates from battery are recovered with battery-grade quality
Recovered energy	• Excess steam is sold for district heating (160 kW load)· • Excess electricity is sold back to grid

[1] Value derived from the 50th percentile of median annual salaries in Germany (2024), including yearly bonuses (EUR 52,159) [48].

The shredding facility costs are taken from [50]. Although the data represent the Portuguese context, the implementation of the model within an EU framework is supported by the authors [50]. The facility modelled in [50] uses data from 2001; therefore fixed costs such as depreciation, maintenance, and insurance were adjusted according to inflation to obtain the estimated values for 2024. Additionally, the number of workers needed to run the facility was used combined with defined yearly salaries for the reference year relative to the output capacity in order to monetize human labour costs per treated weight. Since insurances, maintenance, and electricity costs are provided as the aggregated value [50], to adjust the energy costs, 40% of the aggregated costs were assumed to be allocated to electricity expenses. The electricity expenses were then recalculated using German electricity prices for the reference year, in accordance with [51] (Table A2).

The monetization of the impacts of the mechanical recycling facility for plastic parts was based on [52]. Although the economic model represents plastic recycling within a Belgian context, reference data from a German recycling facility, obtained from [47], are used by the authors. Accordingly, the application of data from [52] is considered appropriate for the German context. The modelled facility includes all costs after collection, including the shredding, washing, density separation, drying, and extrusion of re-granulates [52]. The economic model of the recycling facility encompasses fixed costs, treatment residues, and process operation costs. The fixed costs as well as treatments residue costs have been adjusted according to inflation from 2012 to 2024 at a rate of 1.3, in accordance with [53]. Operational energy costs are provided in an aggregated form by the authors, including electricity, gas, and fuel costs [52]. Based on the supplementary information on energy demand of the modelled machinery, as given by [52], it is assumed that 80% of the necessary energy is obtained from electricity. To adjust energy prices, 80% of the energy costs are multiplied by a price difference factor of 2.47, calculated as the ratio between the

energy price of the reference year (0.173 EUR/kWh) and the value used by the authors (0.07 EUR/kWh). The remaining 20% of the energy costs are left unchanged since the gas prices implemented by the authors are analogue to the reference year. Furthermore, the water demand values are taken from [52] and monetized using average water prices (4.68 EUR/m^3) for the reference year [54]. Labour costs are monetized using the established average yearly salary and a labour force of 1.5 workers per kt processing capacity [55].

Assigning EUR values to cost impacts arising from landfill and incineration facilities was carried out based on data from [56]. Both modelled facilities represent the German context. Fixed costs comprise depreciation, insurance, general expenses, and maintenance and show an aggregated value without inflation adjustment of 34.18 EUR/t for the incineration facility and 3.26 EUR/t for the landfill facility [56]. Both fixed costs and residue treatment costs were adjusted according to inflation from 2000 to 2024 at a rate of 1.58, based on [53]. At the landfill plant, costs arising from securing the facility, performing sample analysis, and after-care restoration were also adjusted at the same inflation rate. The monetization of costs of consumable goods was carried out by multiplying the demand of gas, water, calcium oxide, ammonia (for incineration plants), and fuel (for landfill plants) by prices of the reference year, as shown in Table A2.

For the third scenario additional process operation costs arise from repurposing battery cells into a stationary storage system. To monetize these costs, the economic model from [20] was used. The model of the authors represents the repurposing process from a traction battery with NMC cathode composition into a stationary storage system in Germany. The process includes disassembling batteries to the module level, classifying them by SoH, and balancing their capacities to ensure uniform performance. After this, the modules are integrated with the required materials into storage cabinets and undergo safety and functional tests [20]. Since Scenario 3 represents battery repurposing on a cell level and not on a module level, higher process costs are expected. Material costs are calculated by the authors at 250 EUR per kWh, including the purchase price of the battery system to be repurposed [20]. In this study, the material costs were adjusted to 143.5 EUR per kWh, excluding the battery purchase costs since only operation costs are evaluated. The purchase of all necessary components of the stationary storage system besides the battery modules such as battery management system, cooling system, cabinet, rails, and busbars are included under material costs [20]. Both cost rates were adjusted at an inflation rate of 1.02 between 2023 and 2024 [53].

The impact arising from the transportation of goods was monetized individually for each scenario using the amount of diesel needed to transport the different goods mass between each facility. Data from the LCA process, encompassing vehicle type, travelled distance, and transported load, according by [27], were implemented and combined with diesel prices of the reference year, obtained from [57]. The labour costs were monetized by calculating the driven time by using the defined travel distances and the assumed travel velocities (Table 4). Additionally, a loading and unloading aggregated time of 1 h was assumed for each transportation process. The time demand was then multiplied with gross salary rate per hour, assuming a 40 h/week workload, 52 weeks per year, and a yearly gross salary of EUR 52,500. For the transportation of the battery from the disassembly facility to the battery recycling facility, aggregated values of 35 EUR/FU were used, based on an expert interview with a German disassembly facility manager, since battery transportation includes for additional costs that arise from the requirements of transporting hazardous materials.

Table 5. Treatment facility characterization after labour demand, treatment capacity and reference year of implemented data.

Treatment Facility	Labour [Number of Workers]	Treatment Capacity	Reference Year
Disassembly	6	6000 [t/a]	2025
Shredding [50]	16	69,500 [t/a]	2001
Plastic Recycling [52]	15.75	10,500 [t/a]	2012
Battery Recycling	N/A [1]	6000 [t/a]	2025
Battery Repurposing [20]	1	500 [units/a]	2023
Landfill [56]	30	300,000 [t/a]	2000
Incineration [56]	50	200,000 [t/a]	2000

[1] N/A = not available.

The impact categories defined as benefits are the reduced environmental impact of the different scenarios, the recovered material from recycling, the recovered components to be reused, the recovered energy from the incineration and landfill processes, and the storage capacity from repurposed cells, according to [27]. To monetize these categories, publicly available data were used along with data from best practices.

In the case of the avoided environmental impact of each scenario, the social cost of carbon (SCC) was implemented as monetization method, in accordance with [58]. The SCC is defined as the monetary valuation of the net societal damages associated with the emission of an additional metric ton of carbon dioxide into the atmosphere in a given year [58]. For this study, SCC values (208 EUR/t) were taken from [58] for the year 2024 and a near-term Ramsey discount rate of 2.0%. Furthermore, the SCC encompasses the projected economic consequences of climate change, including effects on agricultural productivity, human health, property and infrastructure, ecosystem services, and the frequency and severity of natural disasters. Conceptually, the SCC also reflects the marginal social benefits of emissions abatement, thereby serving as the theoretically appropriate metric for use in the cost–benefit analyses of policies that alter GHG emissions [58]. Due to inherent data and modelling constraints, current estimates provide only a partial accounting of climate damages, and thus likely understate the full benefits of emission reductions. The SCC was then multiplied by the avoided GHG emissions of each Scenario taken from [26,27]. After [26,27], all assessed scenarios show negative emissions (credits) and are therefore considered benefits in this study. Credits were allocated for material recycling, energy recovery through incineration and landfilling, and the reuse or repurposing of components [26,27]. When components of the FU were reused, the life cycle inventory (LCI) was adjusted by the authors by accounting for the number of reuse cycles during both the production and EoL phases of the FU after [59]. To represent the life extension of these components, two reuse cycles were assumed, encompassing both the first and second life after [15]. Accordingly, the weight of each reused component was divided by the number of reuse cycles across all relevant processes, from production to EoL. Additionally, the system boundary for component repurposing was modelled by the authors following [45], where credits are assigned based on the burdens that the product system of the new application would have incurred without incorporating the repurposed components but where instead avoided.

For the materials recovered through the different recycling processes, available market data were used to monetize the value of each recycled material. Price references were primarily taken from the German market, including market reports by federal trade associations, specialized industry publications and institutional price listings. For most materials, secondary market prices were applied. Following the assumptions given in Table 4, primary prices were used instead (e.g., nickel, cobalt, and manganese sulphates). The market values

for secondary raw materials were averaged over the year of 2024, and the differing data were converted to a per-kilogram basis to ensure comparability. These per-kilogram values were then multiplied by the amount of secondary material obtained from the recycling of one e-moped based on [27].

Monetizing the reuse of components can be carried out by adjusting the original price of components with a value loss factor. For example, when assessing the cost of a reused battery pack, the authors of [20] defined a value loss compared to the original price of 50%. Nevertheless, as the original equipment manufacturer (OEM) of the assessed e-moped discontinued production in 2018, original spare part price data are no longer available. To overcome this limitation, we contacted 21 former authorized OEM dealers and workshops in Germany and conducted informal expert interviews with six partners who continue to provide service support for remaining vehicles. This approach enabled the monetization of spare parts based on their current market assessments. Of the six partners interviewed, two workshop managers provided information on spare parts prices without requiring an inquiry fee. All recovered components were monetized based on these experts assessments with the exception of the cables, which were monetized in accordance with [60]. Cables and rims were monetized for Scenario 1 and Scenario 3 since these components were selected for reuse in those scenarios [27]. The battery case and the main frame were also monetized in Scenario 2, following the increased reuse of components. The valuation of spare parts is inherently complex, as it is subject to significant uncertainty and volatility in costs, availability, and intermittent demand [61]. The influence of different monetization values for the selected parts to be reused is therefore assessed for all scenarios in Section 3.2.

Throughout the EoL treatment, energy is recovered from landfilling and incinerating different parts of the e-moped [27]. Recovered landfill energy was monetized by multiplying the reference year electricity price with the treated mass and the recovery rate (kWh per kilogram), as determined by the applied LCA process [27]. For recovered incineration energy, 70% was allocated to electricity and 30% to steam, based on the applied landfill process given in [27]. Accordingly, 70% of the recovered energy was monetized using the reference year electricity price, while the remaining 30% was assumed to be sold as district heating and monetized using the average district heating price in Germany for the reference year, based on an assumed connection load of 160 kW [62].

The benefits of a stationary storage system were monetized based on the value of storage capacity (500 EUR/kWh) of a storage system with repurposed battery packs, as discussed in [20]. This value was multiplied by the capacity (0.7485 kWh) provided by the repurposed cells of one e-moped, as shown in [27]. Since the storage capacity is an assumed value in [20], a sensitivity analysis was carried out to address the influence of different selling prices per kWh (Section 3.2).

All data implemented for monetizing the impact categories is shown in detail in Appendix A.

Step 7: As all costs and benefits are expressed on a per-e-moped basis and no intertemporal cash flows are involved; discounting is not required and has therefore been omitted from this analysis.

Step 8: In this study, the net present value (NPV) and the cost–benefit ratio (CBR) were used as main indicators for the CBA.

The NPV represents the difference between the total benefits and the total costs of the system under consideration shown in Equation (1). Since no temporal dimension or discounting was included, the NPV is calculated as the direct sum of all monetary benefits minus the total costs. A positive NPV indicates that the benefits outweigh the costs, while a negative NPV suggests that costs exceed benefits.

$$NPV_i = \sum_{j=1}^{n} B_{ij} - C_{ij} \quad (1)$$

NVP_i: net present value of scenario i;
B_{ij}: benefit j of scenario i;
C_{ij}: cost j of scenario i;
n: number of benefits and costs considered.

The CBR is used to provide a relative measure of economic efficiency by expressing the ratio of the total benefits to the total costs shown in Equation (2). A CBR greater than one signifies that the benefits are larger than the costs, whereas a CBR below one indicates the opposite. This ratio is particularly useful for comparing alternatives with different scales of costs and benefits.

$$CBR_i = \frac{\sum_{j=1}^{n} B_{ij}}{\sum_{j=1}^{n} C_{ij}} \quad (2)$$

CBR_i: cost–benefit ratio of scenario i;
B_{ij}: benefit j of scenario i;
C_{ij}: cost j of scenario i;
n: number of benefits and costs considered.

Step 9: To assess the robustness of the CBA results across the different EoL scenarios, a sensitivity analysis was conducted. For the analysed end-of-life scenarios, results were calculated only for discrete data points corresponding to the tested parameter values. To visualize and evaluate the trends between these points, a linear interpolation approach was applied. Additionally, the sensitivity coefficient (SC) was calculated for each trend function. This coefficient quantifies how responsive the NPV of a given scenario is to changes in a selected input parameter, such as a specific cost or benefit item; these are depicted in Equation (3).

$$SC_x = \frac{\Delta NPV_i / NPV_i}{\Delta x / x} \quad (3)$$

SC_x: sensitivity coefficient of parameter x;
x: parameter being varied;
Δx: cost j of scenario i;
ΔNPV: resulting change in NPV.

3. Results

The results section of this study includes the computation of the NPVs for all assessed alternatives (step 8), shown in Section 3.1, and a sensitivity analysis of identified influential parameters (step 9), given in Section 3.2. To help identify sensible parameters, the cost and benefit distributions along the EoL pathways of each scenario are also assessed in Section 3.1.

3.1. Net Present Values

The results for each alternative scenario, presented in Figure 3, include the costs, benefits, and NPV of the EoL treatment of a single e-moped, encompassing the entire process from collection to material, component, and energy recovery. In Scenario 3, the analysis additionally considers the manufacturing of a repurposed stationary energy storage system.

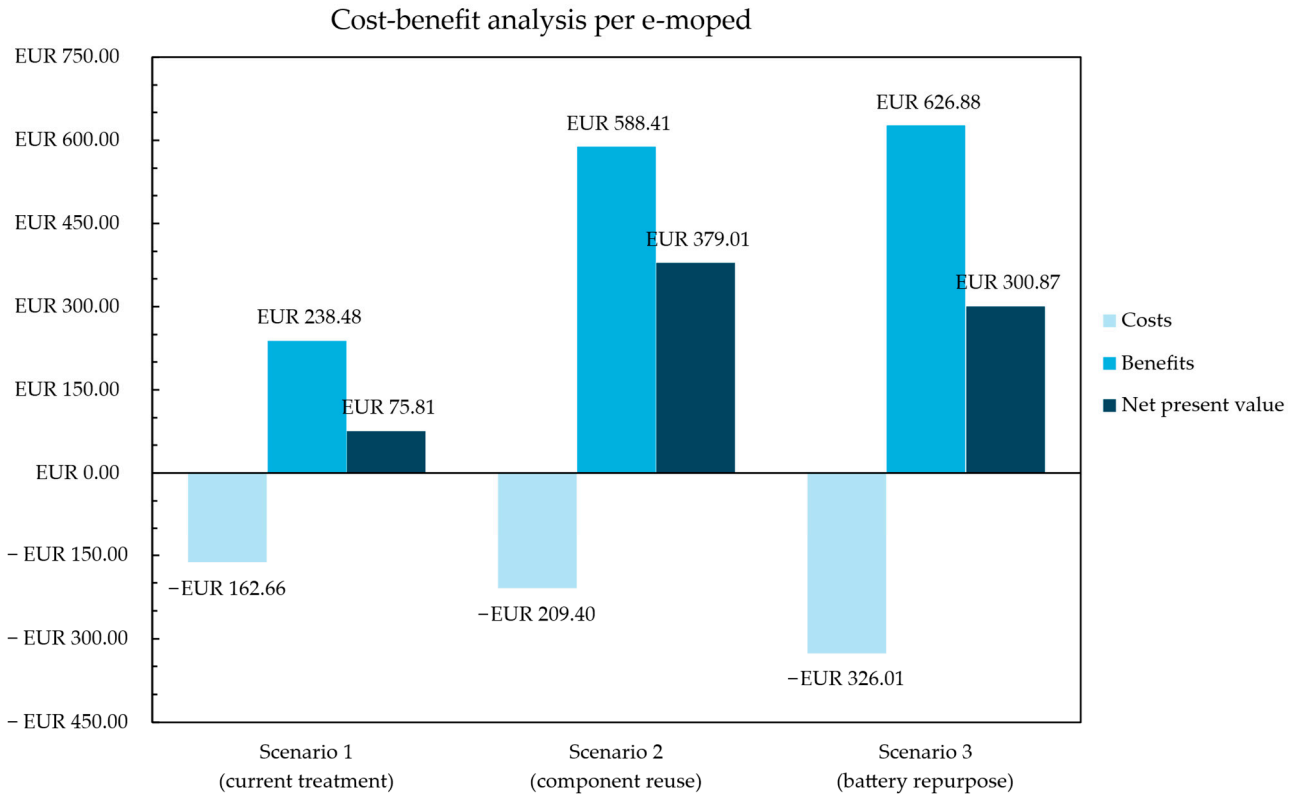


Figure 3. Comparison of aggregated values for costs, benefits, and net present value across the analysed scenarios.

Scenario 1 (current treatment) results in an NPV of EUR 75.81 per FU, while Scenario 2 has an NVP that is 400% higher than Scenario 1 at EUR 379.01 per FU. Scenario 3 presents the highest costs at EUR 326.01 per FU and a NPV of EUR 300.87 per FU. Under the defined assumptions and system boundaries, Scenario 2 shows the highest CBR of 2.8, compared to 1.9 in Scenario 3 and 1.5 in Scenario 1, thereby indicating the most favourable relative return per unit of cost.

While Figure 3 presents the aggregated costs and benefits, Figure 4A–C illustrates the distribution of costs across the analysed scenarios, which result from the different treatment processes as well as the transportation of goods. The disassembly process accounts for the largest share of overall costs, contributing 58% in Scenario 1 and 73% in Scenario 2, primarily due to the high expenses associated with manual disassembly. This effect is particularly pronounced in Scenario 2, where a deeper level of disassembly was required to extract the main rack as a reusable component.

In addition, the battery recycling process represents a significant cost factor, contributing 16% in Scenario 1 and 10% in Scenario 2. By contrast, in Scenario 3, only half of the battery cells were recycled, resulting in a substantially lower cost contribution of 5% for this process.

Across all scenarios, transportation emerges as the third largest cost contributor, accounting for approximately 14% in Scenario 1, 8% in Scenario 2, and 7% in Scenario 3. The main driver of transportation costs is the handling of the battery, which requires certified conditions for the transport of hazardous materials. This specialized requirement constitutes nearly 80% of the total transportation costs.

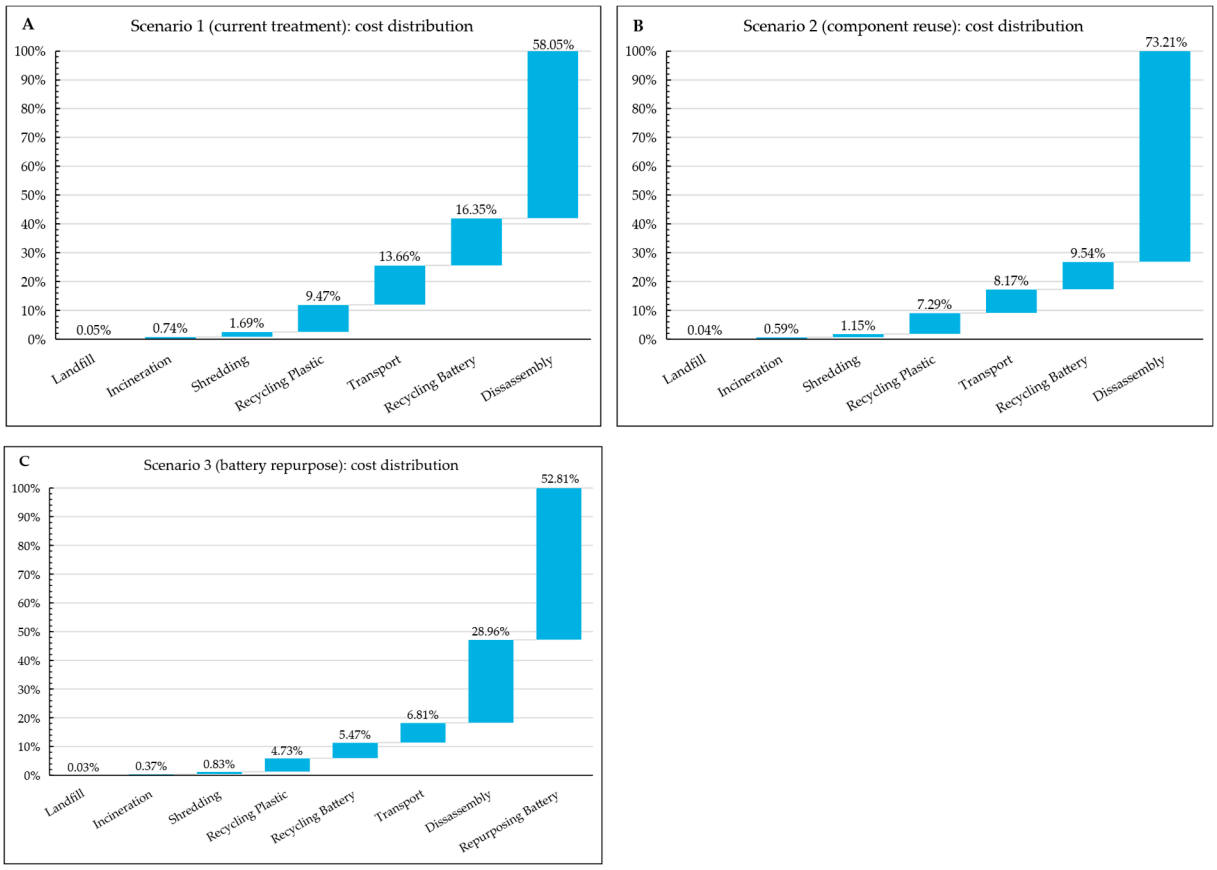


Figure 4. Cost distribution for (**A**) Scenario 1, (**B**) Scenario 2, and (**C**) Scenario 3, under standard cost assumptions.

The recycling of plastics contributed between 5% and 10% of the total costs across the analysed scenarios. Within this process, labour and energy were identified as the dominant cost drivers, jointly accounting for approximately 60% of the total costs associated with plastic recycling.

In Scenario 3, the repurposing of the battery into a stationary energy storage system account for approximately 53% of the total costs. Of these costs, about 64% are attributed to the acquisition of materials required for the manufacturing of the storage system, while the remaining 36% are associated with the repurposing process itself, primarily driven by labour expenses.

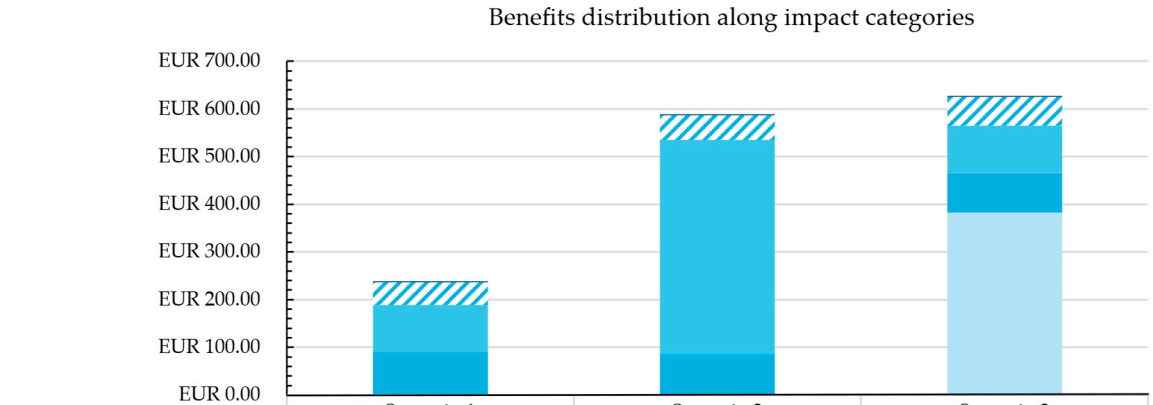
The costs arising from landfill, incineration and shredding jointly account for less than 3% for all scenarios.

The distribution of benefits is presented in Figure 5, illustrating the attribution of benefits to the assessed impact categories. In Scenario 1 (current treatment), benefits derived from the reuse of selected components account for approximately 42% of the total benefits, followed by the recovery of secondary materials (37%) and avoided greenhouse gas (GHG) emissions (20%). The contribution of recovered energy is comparatively minor, at less than 2%.

In Scenario 2 (component reuse), the overall distribution trend remains similar to that observed in Scenario 1. However, the relative contribution of recovered components increases to 76%, while the shares of secondary material recovery (14%) and avoided GHG emissions (9%) decrease in comparison. The influence of the recovered energy remains marginal, at less than 1%.

In Scenario 3 (battery repurpose), most of the benefits are attributed to the energy storage capacity generated from repurposed battery packs (61%). Additional benefits arise

from the reuse of components (16%), followed by the recovery of secondary materials (13%) and avoided GHG emissions (10%).



	Scenario 1 (current treatment)	Scenario 2 (component reuse)	Scenario 3 (battery repurpose)
Recovered energy	EUR 1.97	EUR 1.97	EUR 1.97
Avoided GHG emissions	EUR 47.24	EUR 51.63	EUR 59.64
Recovered components	EUR 100.00	EUR 450.00	EUR 100.00
Recycled materials	EUR 89.26	EUR 84.81	EUR 82.79
Repurposed energy storage	EUR 0.00	EUR 0.00	EUR 382.48

Figure 5. Attribution of benefits to the assessed impact categories across the analysed scenarios.

3.2. Sensitivity Analysis

The previous section outlined the costs and benefits structures and NPV of the three EoL treatment scenarios under standard cost and benefits assumptions. Building on this foundation, Section 3.2 evaluates the impact of varying selected parameters whose values may substantially influence the overall NPV of the assessed alternatives.

Considering that in both Scenario 1 and Scenario 2 most of the costs arise from the dismantling process—where labour costs represent the largest cost component—different labour cost values were assessed by varying the required dismantling time. Variations on the required disassembly time are shown in Figure 6 where, for example, 50% indicates that the required dismantling time is 1.5 times greater than the baseline assumptions described in Section 2. For Scenario 1, the SC of NPV with respect to labour cost is −0.78, meaning that a 1% increase in labour cost results in a 0.78% decrease in NPV. In contrast, the influence of labour costs is considerably smaller in Scenario 2 (−0.16) and Scenario 3 (−0.2), showing that the NPVs of these scenarios are less sensitive to changes in dismantling labour costs. Based on these sensitivity coefficients, the break-even point occurs at a 129% increase in dismantling labour costs (corresponding to 2.29 h of dismantling) for Scenario 1, a 322% increase (8.4 h) for Scenario 2, and a 511% increase (6.1 h) for Scenario 3.

Given that the valuation of spare parts is prone to volatility, variations in the monetization of spare parts and their influence on the NPV are shown in Figure 7. Both variations in the spare parts' value and the NPV are given in percentages in relation to the baseline values for each scenario. For Scenario 1, the SC of NPV with respect to spare parts benefits is 1.43, meaning that a 1% increase in spare parts benefits results in a 1.43% increase in NPV. Similarly to Scenario 1, the influence of the recovered component value shows a SC of 1.21. In contrast, Scenario 3 shows a smaller SC value (0.36), demonstrating that the NPV is less sensitive to changes in the monetization value of spare parts. The break-even point occurs at a 70% decrease in recovered components benefits (benefits = EUR 32.37) for Scenario 1 and an 83% decrease (benefits = EUR 79.17) for Scenario 2. For Scenario 3 a break-even

point only occurs if the benefits arising decrease by 278%, turning benefits into costs of EUR 192.69.

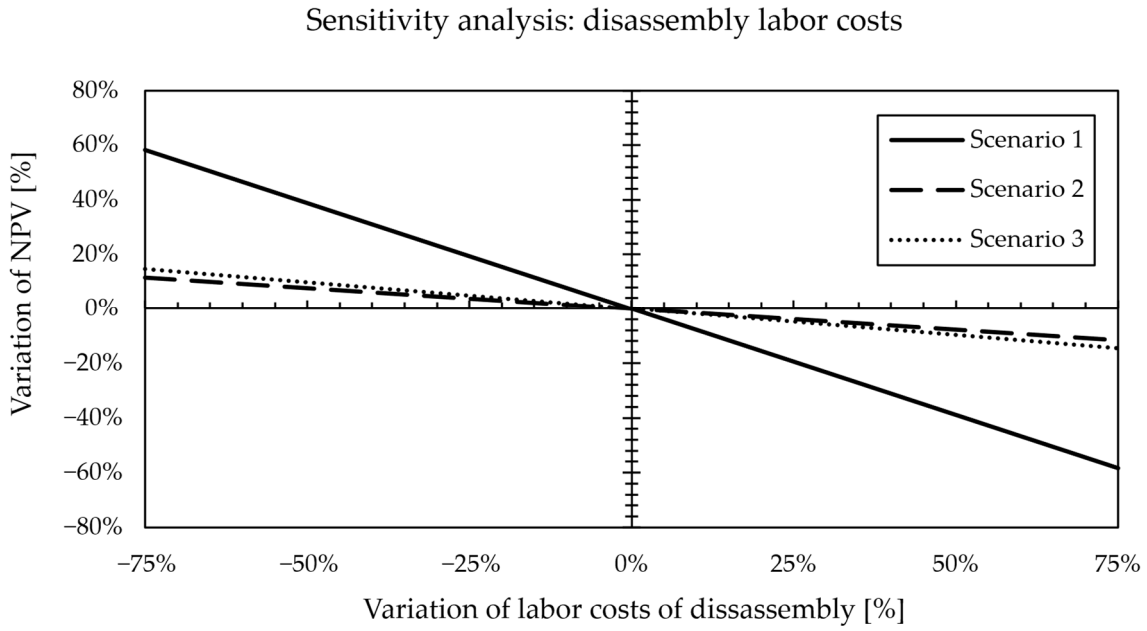


Figure 6. Sensitivity analysis: influence of dismantling labour costs on the net present value of all assessed scenarios.

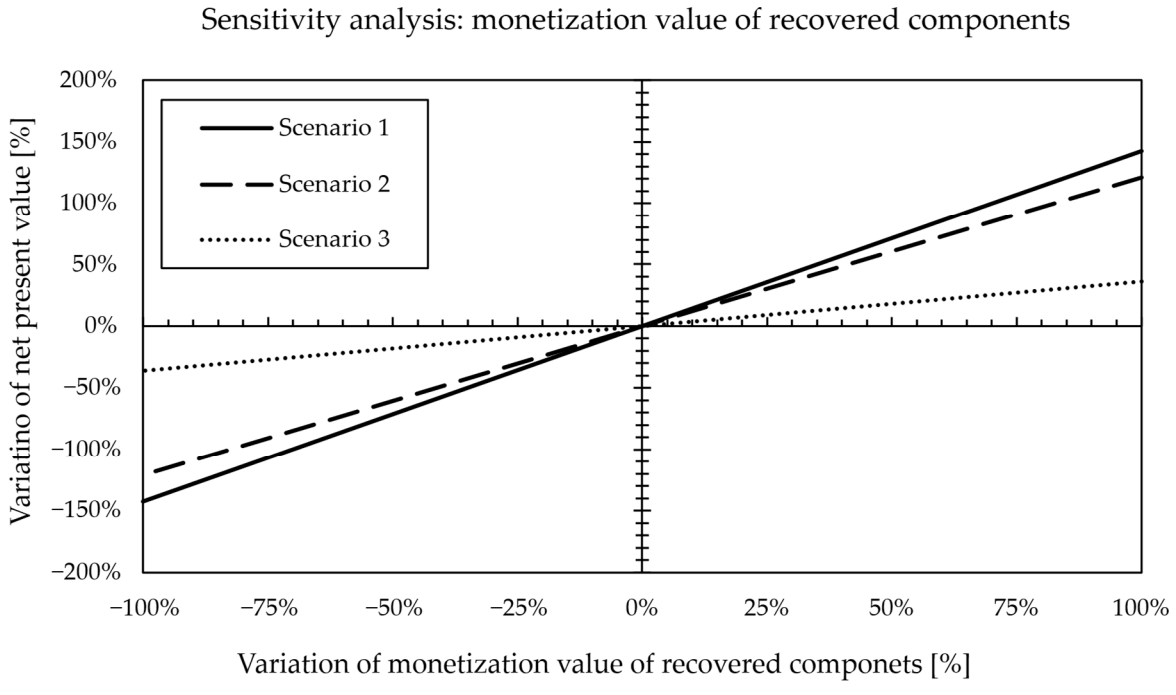


Figure 7. Sensitivity analysis: influence of recovered components monetization on the net present value of all assessed scenarios.

For the sensitivity analysis, the SCC was also assessed by applying different near-term Ramsey discount rates consistent with observed ranges of real market interest rates obtained from [58]: 1.5% (128 EUR/t), 2.0% (baseline 208 EUR/t), and 2.5% (356 EUR/t). In this framework, a higher near-term discount rate corresponds to a higher SCC value because it reflects a higher projected consumption growth rate in the Ramsey model, which

affects how future damages are discounted over time. This variation captures uncertainty in economic growth and intertemporal preferences, and illustrates, how assumptions about future consumption and welfare influence the SCC and the resulting cost–benefit outcomes of the EoL pathways. Given that all EoL scenarios show negative environmental impacts (credits), a higher SCC translates into greater climate-related benefits and higher NPVs.

In Figure 8, Scenario 1 shows the highest sensitivity of the NVP regarding variations in the SCC with a SC of 0.62, meaning that a 1% value increase in the SCC causes a 0.62% value increase in the NVP. For Scenario 2 and Scenario 3 this effect is lower, with SC values of 0.14 and 0.20, respectively. For every ±50 EUR/t change in the SCC (from 208 EUR/t), the NPV increases by approximately EUR 11.3, EUR 12.8, and EUR 14.5 for Scenarios 1, 2, and 3, respectively.

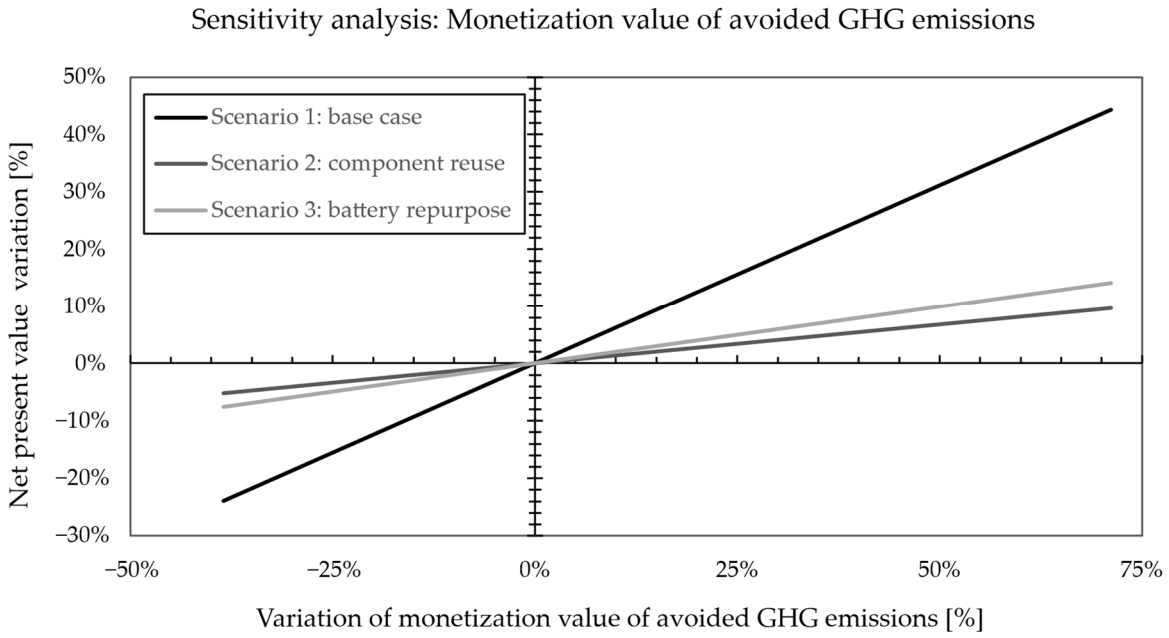


Figure 8. Sensitivity analysis: influence of social cost of carbon on the net present value of all assessed scenarios.

For Scenario 3, the sensitivity of the NPV to the assumed sale price per kWh of storage capacity in a repurposed energy storage system is presented in Figure 9. The analysis simultaneously varies the sale and cost values per kWh. Sales values are represented along the *x*-axis, while five different cost scenarios are depicted as functions within the graph. These cost scenarios include both material and process costs per kWh. The analysis considers four alternative repurposing costs (125 EUR/kWh, 175 EUR/kWh, 275 EUR/kWh, and 325 EUR/kWh) in comparison to the baseline cost of 225 EUR/kWh. At baseline costs, the break-even point is maintained as long as the sale price does not fall below 106.69 EUR/kWh. For lower repurposing costs, the break-even point occurs at a sale price of 6.64 EUR/kWh for 125 EUR/kWh costs and 56.64 EUR/kWh for 175 EUR/kWh costs. In contrast, when higher repurposing costs are assumed, the breakeven point is reached at a lower decrease in sale prices of 156.64 EUR/kWh for 275 EUR/kWh costs and 206.64 EUR/kWh for 325 EUR/kWh costs. Conversely, when maintaining the established sale price of 500 EUR/kWh, the break-even point is achieved at repurposing costs of 618.36 EUR/kWh. Figure 9 shows absolute values which have not been normalized. The absolute sensitivity has a value of 0.77, indicating that an increase of 1 EUR/kWh in the assumed sale price corresponds to an increase of 0.77 EUR per e-moped in the NPV.

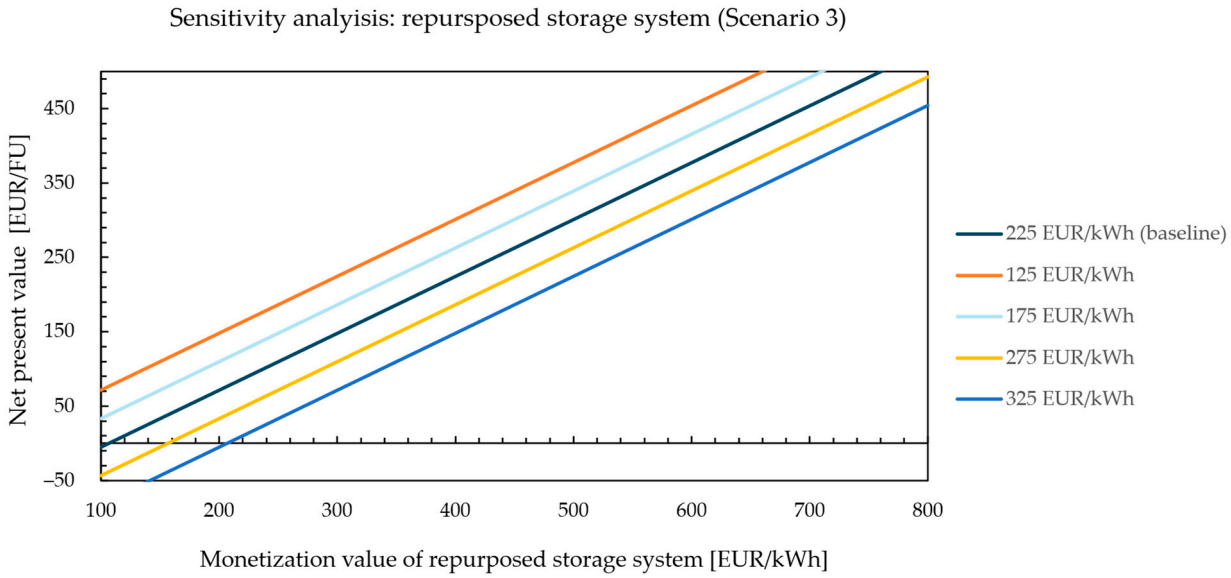


Figure 9. Sensitivity analysis: influence of the sales price in relation to the purchase price of a repurposed storage system on the net present value in Scenario 3.

4. Discussion

The results of this study demonstrate a considerable economic potential of alternative EoL scenarios when compared with the status quo EoL treatment in Germany. However, the system boundary of the present analysis reflects an "in-house" or joint venture setting in which no entry fees are considered and a constant supply of material inputs is assumed across all phases. In practice, such conditions are rarely met, since individual facilities operate under business models designed to ensure their own profitability, typically through entry fees and other market mechanisms. The entry fee serves as an economic adjustment mechanism. When the revenues generated along the treatment chain fail to fully cover operational costs, facilities are more likely to impose entry fees to bridge this financial gap and maintain economic viability. These fees effectively transfer part of the financial responsibility from the processing facility to upstream actors—primarily the OEM under the Extended Producer Responsibility (EPR) principle [63]. The underlying intent of such fees is to incentivize more sustainable EoL pathways, such as recycling, over less desirable options like landfilling, without causing economic losses for recycling facilities. Previous studies have shown that battery recycling scenarios often remain economically unviable without entry fees, given the high processing and investment costs relative to the value of recovered materials [20].

The magnitude of the entry fee is typically determined by the quality and recovery potential of incoming material streams. High-purity materials generally incur lower fees due to greater recoverable value, whereas heterogeneous or contaminated inputs yield fewer recoverable fractions and thus attract higher fees.

The results of this study indicate that, although Scenario 1 (current treatment)—which focuses predominantly on recycling—generates positive returns, it remains the least profitable among the three evaluated scenarios. Consequently, recycling facilities operating under such conditions are likely to impose higher entry fees to compensate for the limited economic benefit. Introducing these market-based entry fees would, in effect, shift part of the economic burden from processing facilities to OEMs. In contrast, in-house or joint venture models, assuming constant material throughput, could achieve economic viability more independently, reducing reliance on entry fees as a financial balancing tool.

As such, the current results can be understood as reflecting a pilot concept in which all operations are integrated within a single enterprise or business model. Despite this limitation, these findings nevertheless highlight the strong potential of collaborative approaches, underlining the need for further research on circular business models that actively integrate multiple stakeholders. Whereas CBAs quantify the monetary impacts and benefits of measures, CBMs address the structural and strategic design of business models necessary for the implementation and scaling of circular solutions. A decisive factor in the cost structure across the scenarios is the dismantling process, which emerges as one of the most significant cost contributors. This is primarily due to the reliance on manual dismantling, itself a consequence of the high variability in vehicle and battery architectures. Most of the reuse and repurposing strategies assessed in this study depend heavily on effective dismantling, thereby underscoring the crucial role of CD in future LEV concepts. Design for disassembly, in particular, will be essential to facilitate component reuse and material recovery at the EoL stage and should be considered a priority for future product development. Moreover, technological advancements over time can substantially alter process efficiencies and cost structures, introducing uncertainty when using historical data. In the context of our study, cost data for key processes such as shredding and plastic recycling are based on older sources and may not fully capture recent innovations. For example, developments in recycling of electric vehicle batteries and improved disassembly processes have the potential to increase efficiency and reduce operational costs. These considerations highlight that the economic outcomes reported here should be interpreted with an awareness of potential process improvements over time.

An additional contribution of this study lies in the explicit integration of environmental valuation into the economic assessment through the implementation of a CBA. This is, to our knowledge, the first study to incorporate such a perspective for the chosen LEV. The results reveal a high sensitivity of the current treatment scenario to environmental impact valuation, with its NPV reduced by 20% in the absence of monetised climate impacts. This demonstrates the critical role of including environmental considerations in economic evaluations. However, excluding this metric would not affect the ranking of the scenarios in terms of NPV.

Applying the SCC to monetise the GWP_{100} enables explicit valuation of climate change impacts, thereby strengthening the economic assessment of circular strategies. Nevertheless, it must be acknowledged that the monetisation of environmental impacts, as well as broader aspects such as quality of life, within CBAs remains ethically and methodologically contested. This raises questions about whether monetary valuation can adequately capture their societal significance. Future analyses could further enhance environmental inclusion by also considering impacts on air pollution, human health, and ecosystems, as well as on land and water resources.

Finally, in line with Step 10 of this study's CBA framework based on [30], a recommendation is warranted. Of the three alternatives assessed, Scenario 2 emerges as the most economically attractive option with the highest NPV. However, the characteristics of Scenario 3, particularly the repurposing of batteries, offer complementary advantages and could be fruitfully combined with the approach used in Scenario 2, suggesting that integrating both scenarios may result in an even higher NPV. Overall, the results suggest that strategies extending the useful life of products—whether through component reuse or battery repurposing—yield greater economic benefits than direct recycling, incineration, or landfilling. At the same time, the volatility of secondary markets for used components, particularly in cases where vehicle models are rapidly discontinued, must be recognised as a challenge. Nevertheless, the sensitivity analysis indicates, that the results of this study are robust with respect to key assumptions and data uncertainties. Even under substantial

variations in dismantling labour costs and recovered component values, all alternative EoL scenarios (S2 and S3) maintain a positive NPV compared to the current treatment. The break-even analysis shows that labour costs would need to increase by more than 100% (up to 511% in Scenario 3) or recovered component revenues would need to decrease by over 70% for Scenario 1 and 83% for Scenario 2 before negating their economic advantage. These results suggest that, despite relying on certain assumptions and older cost data for specific processes (e.g., shredding), the relative economic ranking of the scenarios remains stable. While technological and market developments may shift absolute cost levels over time, they are unlikely to overturn the comparative performance between the investigated EoL configurations. Furthermore, ongoing developments towards standardisation, as observed in other product–service systems such as mobile device charging, suggest that the current lack of harmonisation in LEVs should not be regarded as a disqualifying barrier to pursuing alternative EoL pathways. In the European context, regulatory initiatives such as the Battery Passport, the Circular Design Directive, and the Right to Repair are already advancing in this direction. A comparable precedent can be found in the electromobility sector, where EU regulation has mandated the adoption of Type 2 connectors for AC charging and CCS Combo 2 for DC fast charging across all public charging points. This regulatory framework has effectively reduced market fragmentation and secured interoperability across member states. Such precedents indicate that regulatory intervention can play a decisive role in enabling circular practices, and the results of this study therefore reinforce the economic viability of strategies promoted by current regulations, including battery repurposing.

As this study focusses on the use-case of one specific shared LEV in one specific geographic region, it appears promising to systematically investigate the transferability of results in future studies. Based on our findings, however, we generally expect similar results for LEVs with comparable material and component composition. For LEVs, in which the battery has a lower share of the vehicle's weight, Scenario 2 is likely to become an even more economically attractive option for the revenues of recovered components and therefore the NPV are expected to be even higher than for our use-case. Vice versa, LEVs, in which the battery has a higher share of the vehicle's weight, Scenario 3 might be the most economically attractive option with higher revenues from repurposing the battery. While we do not expect significant differences in the results for shared and non-shared LEVs, the geographic region might have a major impact on the results. This particularly applies to the variation in key cost and benefit parameters (e.g., labour costs, revenues of recovered components). While our study provides sensitivity analyses on these and other parameters, future studies can systematically investigate the influence of the geographic region on the economic performance of the different scenarios allowing a higher transferability of the results.

It is important to emphasise that while repurposing the battery is shown here to be technically, environmentally, and economically feasible, it does not yet form part of the prevailing status quo. The realisation of these alternatives will depend on parallel progress at the regulatory, business model, and infrastructural levels. Only through coordinated efforts across these domains can the confirmed potential of CD and EoL strategies for LEVs be effectively implemented. Furthermore, cell chemistry and battery size influence not only recovery efficiency and the overall environmental impact but also the economic value of recovered precious metals. Future studies should investigate variations in these parameters within the context of LEVs, as well as their implications for performance, economic feasibility, and environmental outcomes in stationary second-life applications.

As the primary goal of this paper was to economically analyse different EoL scenarios previously evaluated from an ecological perspective [26,27], the results of the present study must be considered in relation to those ecological findings. The earlier studies indicated

that Scenario 3 (battery repurposing) achieves the highest ecological benefits, followed by Scenario 2 (component reuse) and Scenario 1 (current treatment). In contrast, from an economic perspective, Scenario 2 (component reuse) performs best, followed by Scenario 3 (battery repurposing) and Scenario 1.

Because our economic assessment incorporates environmental valuation, a direct discussion of trade-offs between the economically optimal scenario and the ecologically optimal scenario is not straightforward. Nevertheless, future investigations could consider separating the ecological and economic dimensions to better explore the potential trade-offs among scenarios, taking into account environmental impact, economic performance, and the likelihood of adoption.

5. Conclusions

This study ties into the ecological assessment of EoL strategies of LEVs conducted in [26,27] and provides an economic assessment of different EoL scenarios for LEVs. For this purpose, an extensive CBA was carried out. The current EoL treatment amounts to an NPV of EUR 75.81 per e-moped. Scenario 3 (battery repurpose) accounts for a NPV of EUR 300.87 per e-moped. Scenario 2 (component reuse) shows the highest NPV of EUR 379.01 per e-moped. In relative terms, Scenario 3 performs around 296% better than the current EoL treatment, while Scenario 2 surpasses it by 400% and Scenario 3 by 26%, respectively. The sensitivity analysis reveals that the current EoL treatment scenario is particularly sensitive to variations in dismantling labour cost (SC = 1.43), monetization of recovered components (SC = -0.78), and the social cost of carbon (SC = 0.62). Scenario 2 is most sensitive to variation in the monetization of recovered components (SC = 1.21). The sale value of the repurposed storage capacity shows the highest influence on the NPV in Scenario 3.

Based on these results and the surrounding discussion, both practical and research-oriented implications can be formulated. From a practical perspective, our study provides impulses for a product design approach that aligns with the principles of eco-design and circularity design. It encourages product designers to prioritize easy disassembly already during the product development phase, aiming to reduce the labour time required for disassembly processes. From a research perspective, our study lays the foundation for further scientific exploration into how the generated insights can be translated into a circular business model and how such a model can be applied profitably in real-world contexts over the long term. Furthermore, future research could build on the ecological assessment of the examined EoL scenarios and the economic analysis presented in this study in order to also investigate relevant parameters related to the social dimension of sustainability. By doing this, a fully comprehensive sustainability assessment of the EoL phase of LEVs could be achieved.

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Abbreviations

The following abbreviations are used in this manuscript:

ABS	Acrylonitrile butadiene styrene
CaO	Calcium oxide
CapEx	Capital expenditure
CBA	Cost–benefit analysis
CBR	Cost–benefit ratio
CE	Circular economy
CO_2	Carbon dioxide
CO_2-eq	Carbon dioxide equivalent
CPI	Currency specific consumer prices indexes
DE	Germany
EoL	End-of-life
EU	European Union
EUR	Euro
EVs	Electric vehicles
FU	Functional unit
GHG	Greenhouse emissions
GWP_{100}	Global warming potential over 100 years
HSS	House storage system
LCA	Life cycle assessment
LEVs	Light electric vehicles
LIB	Lithium-ion battery
NMC	Nickel, manganese, cobalt
NPV	Net present value
OEM	Original equipment manufacturer
OpEx	Operational expenditure
PTFE	Polytetrafluoroethylene
RQ	Research Question
RVB	Remaining vehicle body
SBR	Styrene butadiene rubber
SC	Sensitivity coefficient
SCC	Social costs of carbon
SLF	Shredder light fraction
SoH	State of health
USD	United States dollar
WEEE	Waste electrical and electronic equipment

Appendix A

All prices and costs implemented for the monetization of costs and benefits for all assessed scenarios are shown in Table A1.

Table A1. Prices implemented for the valuation of all costs and benefits for the assessed scenarios.

Affected Process/Input-Waste/Recycled/Recovered Product	Price/Cost	Unit	Time Period	Source
Diesel	1.65	EUR/L	2024	[57]
Electricity	0.173	EUR/kWh	2024	[51]
Gas	0.0713	EUR/kWh	2024 (July–December)	[64]
Heat [1]	141.91	EUR/MWh	2024 (1 April)	[62]
Water [2]	4.68	EUR/m^3	2024 (January)	[54]
Gross salary [3]	52,159	EUR/year	2024	[48]
SCC [4]	226.76	EUR/tCO_2	2024	[58]
CaO [5]	151.52	EUR/t	2024	[65]
Ammonia [6]	227.28	EUR/t	2024	[66]
Secondary aluminium ingot	134.33	EUR/100 kg	2024	[60]
Stainless steel scrap	1827.63	EUR/mt	2024	[67]
Steel scrap	338.44	EUR/t	2024	[68]
Copper	6750	EUR/t	2025	[69]
Secondary ABS	1346.67	EUR/t	2024	[70]
Secondary SBR	0.9	EUR/kg	2025	[71]
Secondary PP	908.33	EUR/t	2024	[70]
PTFE	2	EUR/kg	2025	[72]
Cobalt sulphate [7]	5593.41	EUR/mt	2025	[73]
Nickel sulphate [8]	3387.73	EUR/mt	2025	[74,75]
Iron	338.44	EUR/t	2024	[68]
Manganese sulphate [7]	634.91	EUR/t	2025	[73]
Used battery case	50	EUR	2025	OEM dealer
Used cables [9]	7503.75	EUR/t	2024 (January, February)	[60]
Used aluminium rim	100	EUR	2025	OEM dealer
Used steel frame	300	EUR	2025	OEM dealer
Repurposed stationary storage system	374.25	EUR/kWh	2023	[20]

[1] 160 kW connection load. [2] Sum of water supply and wastewater prices. [3] 50th percentile of median annual salaries in Germany (2024), including yearly bonuses. [4] Modified from source with USD inflation rate 2020–2024 and an exchange rate USD to EUR = 0.9239. [5] Expressed in EUR with an exchange rate (2024) USD to EUR = 0.9239. [6] Expressed in EUR with an exchange rate (2024) USD to EUR = 0.9239. [7] Primary material price. [8] Primary material price, expressed in EUR with an exchange rate (2025) USD to EUR = 1.1711. [9] Price for cable wire 1.

Data adjustments and normalization were carried out implementing different factors. Table A2 shows a detailed depiction of used inflation, deflation, conversion, and exchange factors.

Table A2. Implemented conversion and adjustment rates for the normalisation of data.

Description	Factors (Conversion, Inflation, Deflation, Exchange)
Conversion: Kilogram to litre (Diesel) [1]	1.20 [76]
Inflation: EUR (2001) to EUR (2024)	1.548 [53]
Inflation: EUR (2012) to EUR (2024)	1.301 [53]
Inflation: EUR (2000) to EUR (2024)	1.58 [53]
Inflation: USD (2020) to USD (2024)	1.212 [77]
Deflation: EUR (2025) to EUR (2024)	0.983 [49]
Exchange: USD (2024) to EUR (2024)	0.924 [75]
Exchange: USD (2025) to EUR (2025)	1.17 [75]

[1] Conversion rate calculated assuming density value for diesel of 833 kg/m^3, in accordance with [76].

A depiction of all monetized costs and benefits after activities are presented in Table A3.

Table A3. Valuation in EUR of all costs and benefits according to activity type for all assessed scenarios.

Activity	Scenario 1 (Current Treatment) [EUR]	Scenario 2 (Component Reuse) [EUR]	Scenario 3 (Battery Repurpose) [EUR]
Transport total	22.218	17.115	22.218
E-moped: Collection: Diesel	0.173	0.173	0.173
E-moped: Collection: Labour	2.678	2.678	2.678
RVB: Dismantling to Shredding: Diesel	0.331	0.055	0.331
RVB: Dismantling to Shredding: Labour	0.166	0.146	0.166
Plastic: Dismantling to Recycling: Diesel	0.208	0.208	0.208
Plastic: Dismantling to Recycling: Labour	0.104	0.104	0.104
Battery: Dismantling to Recycling: Total	18.431	13.624	18.431
SLF: Shredding to Incineration: Diesel	0.011	0.011	0.011
SLF: Shredding to Incineration: Labour	0.006	0.005	0.006
SLF: Shredding to Landfill: Diesel	0.024	0.024	0.024
SLF: Shredding to Landfill: Labour	0.012	0.012	0.012
Electronics: Recycling to Incineration: Diesel	0.003	0.003	0.003
Electronics: Recycling to Incineration: Labour	0.002	0.002	0.002
Plastic: Recycling to Incineration: Diesel	0.046	0.046	0.046
Plastic: Recycling to Incineration: Labour	0.023	0.023	0.023
Dismantling total	94.419	153.301	94.419
Labour	58.882	117.763	58.882
Insurance	1.132	1.132	1.132
General expenses	34.405	34.405	34.405
Shredding Total	2.746	2.414	2.695
Labour	0.971	0.854	0.925
Maintenance, Insurance, Electricity	0.802	0.705	0.802
Depreciation	0.699	0.614	0.699
General expenses	0.275	0.241	0.269
Recycling (Plastic) total	15.407	15.274	15.407
Labour	2.512	2.490	2.512
Depreciation	2.708	2.685	2.708
Maintenance	0.591	0.586	0.591
Insurance	0.103	0.103	0.103
Treatment residues	1.248	1.237	1.248
Energy	6.629	6.572	6.629
Water	0.075	0.074	0.075
General expenses	1.541	1.527	1.541
Incinerating total	1.203	1.237	1.203
Labour	0.245	0.245	0.245
Treatment residues	0.315	0.314	0.315
Depreciation	0.309	0.308	0.309
Maintenance	0.034	0.034	0.034
Insurances	0.097	0.097	0.097
Gas	0.055	0.086	0.055
Water	0.014	0.014	0.014
CaO	0.009	0.009	0.009
Ammonia	0.005	0.005	0.005
General expenses	0.120	0.124	0.120
Landfill total	0.082	0.082	0.082
Labour	0.010	0.010	0.010
Treatment residues	0.004	0.004	0.004
Depreciation	0.010	0.010	0.010
Maintenance	0.000	0.000	0.000
Insurances	0.001	0.001	0.001
Fuel	0.003	0.003	0.003
Aftercare and Restoration	0.044	0.044	0.044
Guard and Analysis	0.003	0.003	0.003
General expenses	0.008	0.008	0.008

N/A = not available.

Table A3. *Cont.*

Activity	Scenario 1 (Current Treatment) [EUR]	Scenario 2 (Component Reuse) [EUR]	Scenario 3 (Battery Repurpose) [EUR]
Recycling (Battery) total	26.589	19.982	17.833
Repurposing total	N/A	N/A	172.156
Material	N/A	N/A	109.773
Process	N/A	N/A	62.383
Total costs	162.664	209.405	326.012
Avoided GHG emissions total	47.241	51.629	59.638
Recovered material total	81.083	76.635	74.608
RVB: aluminium	20.109	20.109	20.109
RVB: stainless steel	3.180	3.180	3.180
RVB: copper	4.523	4.523	4.523
RVB: ABS	7.218	7.218	7.218
RVB: SBR	16.200	16.200	16.200
RVB: steel	13.639	11.042	13.639
RVB: polypropylene	3.188	3.188	3.188
Battery: PTFE	0.160	0.075	0.080
Battery: cobalt sulphate	2.070	2.070	1.035
Battery: aluminium	3.600	2.029	1.800
Battery: nickel sulphate	2.808	2.808	1.404
Battery: polypropylene	0.845	0.670	0.422
Battery: steel	0.355	0.335	0.178
Battery: iron	0.142	0.142	0.071
Battery: copper	2.970	2.970	1.485
Battery: manganese sulphate	0.076	0.076	0.076
Recovered components total	108.179	458.179	108.179
RVB: rims	100.00	100.00	100.00
RVB: main frame	N/A	300.00	N/A
Battery: cables	8.179	8.179	8.179
Battery: case	N/A	50.00	N/A
Repurposed energy storage total	N/A	N/A	382.484
Recovered energy total	1.973	1.970	1.973
Landfill: electricity	0.077	0.077	0.077
Incineration: electricity	0.652	0.651	0.652
Incineration: heat	1.244	1.242	1.244

N/A = not available.

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