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Environmental and socio-economic evaluation of a groundwater bioremediation technology using social Cost-Benefit Analysis: Application to an in-situ metal(loid) precipitation case study

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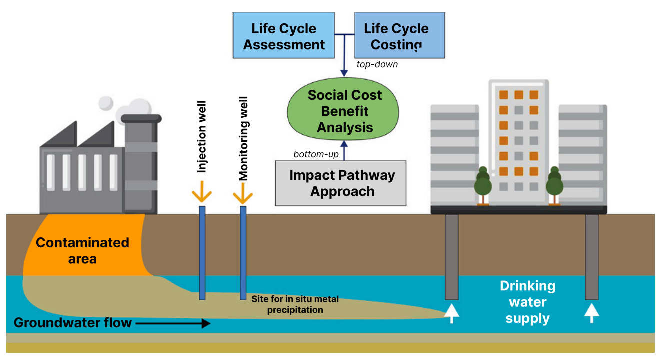
HIGHLIGHTS

- Life Cycle Assessment and environmental Life Cycle Costing are calculated in a bioremediation project at field scale.
- Impact Pathway Approach is applied to illustrate the monetization of external revenues at full scale.
- Social Cost Benefit Analysis can be performed using a top-down or a bottom-up approach.
- Sensitivity and scenario analyses are performed to determine the point at which the Net Present Value becomes positive.
- Social discount rates should be carefully chosen for the evaluation of long term projects.

GRAPHICAL ABSTRACT



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ABSTRACT

Bioremediation can be an alternative or complementary approach to conventional soil and water treatment technologies. Determining the environmental and socio-economic impacts of bioremediation is important but rarely addressed. This work presents a comprehensive sustainability assessment for a specific groundwater bioremediation case study based on In-situ Metal(loid) Precipitation (ISMP) by conducting a social Cost-Benefit Analysis (CBA) using two different approaches: environmental Life Cycle Costing (eLCC) and Impact Pathway Approach (IPA). Externalities are calculated in two ways: i) using Environmental Prices (EP) to monetize Life Cycle Assessment (LCA) results and metal(loid)s removed at field scale, and ii) following the IPA steps to determine the social costs avoided by removing arsenic contamination at full scale. The results show that, in the baseline scenario, the project is not socio-economically viable in both cases as the Net Present Value (NPV) is

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−129,512.61 € and − 415,185,140 € respectively. Sensitivity and scenario analyses are performed to identify the key parameters and actions needed to reach a positive NPV. For instance, increasing the amount of water treated per year to 90 m^3 and assuming a 20 % increase in operation costs and a 60 % increase in construction costs can make the project socio-economically viable at the field scale, while a reduction in the social discount rate from a 4 % to a 2 % can lead to a positive NPV at the full scale. The approaches proposed in this work may be useful for practitioners and policymakers when evaluating the environmental and socio-economic impacts of bioremediation technologies at different scales and regions, as well as human health impacts caused by contaminants at the current legal limits.

1. Introduction

Since the Industrial Revolution, industrial activities have played a vital role in shaping the modern world. They have revolutionised production processes, fuelled economic growth, and provided large scale employment opportunities. However, along with these benefits, industrial activities have also caused significant environmental challenges (Quito et al., 2023). One of the major problems associated with industrialization is the contamination of air, soil and water by heavy metals and metalloids (Haileslassie and Gebremedhin, 2015). Metal(loid)s, including nickel, zinc, cadmium and arsenic, are often released into the environment through mining, manufacturing, and improper waste disposal, among other activities, and their accumulation in soil and water poses serious risks to human health and ecosystems, resulting in long-term environmental degradation (Elumalai et al., 2017). Human exposure to heavy metals and metal(loid)s can cause health problems such as cancer, chronic bronchitis, IQ loss or diabetes (Briffa et al., 2020). The exposure pathways to these pollutants can be through direct inhalation from the air, through dermal contact with contaminated materials, and through ingestion by consuming contaminated agricultural products (food and beverages) and drinking water (ETC/ATNI, 2021).

Treatment of metal(loid)-contaminated soil and (ground)water resources is challenging and generally costly due to their persistence and high water solubility, which can result in large contamination plumes that extend well beyond the boundaries of the contamination source (Han et al., 2023). Clean groundwater is of paramount importance as it is a critical source of drinking water and plays an important role in agriculture, industry and natural resources (e.g. surface water systems). Conventional groundwater treatment is generally represented by pump-and-treat on-site measures, where contaminated groundwater is pumped to the surface (ex-situ) and it is treated either physically (by ion exchange or adsorption) or chemically (by precipitation) (Pohl, 2020; US EPA, 2005).

Alternatively, innovative in-situ biological treatment methods (bioremediation) are often considered to be more cost-effective and sustainable remediation treatment options than conventional technologies (Bidja Abena et al., 2019; Caliman et al., 2011; Syafiuddin et al., 2020). In-situ bioremediation uses natural processes involving specific plants and/or microorganisms to assist in the remediation of metal(loid)s in soil and groundwater directly at the contaminated site. Common in-situ bioremediation approaches include phytoremediation (use of plants to remediate contaminated areas), natural attenuation (natural microbial activity to degrade or precipitate contaminants without human intervention), bioaugmentation (microorganisms introduced to enhance degradation and transformation capabilities) and biostimulation (addition of nutrients or electron donors to enhance the metabolic activity of native microorganisms) (Bala et al., 2022; Hashim et al., 2011). Despite their potential, the implementation of bioremediation processes is not necessarily straightforward and can be limited by environmental, technical and regulatory challenges (Mondal et al., 2024).

Determining whether a particular bioremediation technology is a sustainable approach or not is a site-specific question that should take into account technical, environmental, social and economic factors as well as the local and governance context (ISO, 2017). This is particularly important given the fact that groundwater remediation is generally costly, long-term and perceived as an economic burden by those legally responsible for the remediation. To evaluate this, Cost-Benefit Analysis (CBA) can help decision makers as a tool to assess the impact of a particular project accounting the costs incurred and the potential revenues (OECD, 2018). This tool can be based on a private or social perspective, depending on the stakeholder. While private CBA focuses on the evaluation of a particular stakeholder, social CBA assesses the impact on society by monetizing environmental and human health effects, also called externalities, and including them in the Net Present Value (NPV) calculation, a financial indicator used to determine the social profitability of a project or policy (European Commission, 2014). The externalities can be monetized by using different methods to include them in the social CBA, both as costs and as benefits. One way is by using Life Cycle Assessment (LCA) to calculate the environmental impacts and applying valuation techniques to convert these impacts into monetary values, following the environmental Life Cycle Costing (eLCC) approach (Hunkeler et al., 2008; Swarr et al., 2011). Another is to follow the Impact Pathway Approach (IPA), a framework originally developed to quantify and monetize the environmental impacts caused by air pollution in different European regions (Bickel, Friedrich, et al., 2005).

The application of social CBA to analyse the financial profitability and the sustainability of a remedial measure for the treatment of contaminated soil or groundwater has been explored in various studies (Foglia et al., 2021; Lavee et al., 2012; Mumbi and Watanabe, 2022; Söderqvist et al., 2015; Van Wezel et al., 2008; Volchko et al., 2017; Wan et al., 2016). The inclusion of LCA in social CBA has been previously conducted in analyses such as that of Huysegoms et al. (2019), where the authors compare the social profitability of two remediation techniques in a social CBA, applying Environmental Prices (EP) as the monetary valuation method to calculate the social revenues of the project. In another study, the same authors compare the monetized LCA results with external costs and benefits calculated by the IPA for a soil remediation project, concluding that both tools should be complementary (Huysegoms et al., 2018).

Following these studies, in the present work we evaluate the environmental and socio-economic aspects of a groundwater bioremediation case study using monetized LCA (eLCC) and IPA to calculate the social CBA. The bioremediation approach consists of the in-situ (in the aquifer) precipitation of metal(loids) (ISMP) as stable sulfide minerals formed by the activity of native sulfate-reducing bacteria (SRB) following their biostimulation by addition of an organic substrate. The ISMP approach was previously validated in a real contaminated emplacement in Belgium (Pérez-de-Mora et al., 2024). This case study was selected for various reasons, including: i) the novelty and potential of the bioremediation approach for full-scale implementation and long-term treatment, ii) the representativeness of the in-situ approach, including the injection and distribution of reagents into the aquifer (also valid for in-situ chemical oxidation or reduction treatments), and iii) the representativeness of a biostimulation treatment, which requires changing the physico-chemical conditions of the aquifer for optimal bioremediation activity.

To the best of our knowledge, this work is the first environmental and socio-economic evaluation of an ISMP bioremediation technology for groundwater treatment using social CBA. In addition, this work provides two main innovations that can be applied in future studies for the evaluation of other (emerging) (bio)remediation technologies in-situ. One is the extension of the research outcomes from the previous studies, considering that monetized LCA and social CBA are complementary. The former serves as a tool to account for externalities that are included in the latter for the technology implemented at field scale. Therefore, the main objective of this paper is to evaluate, through social CBA, the socio-economic profitability of a bioremediation approach to reduce contaminant concentrations not only below the regulatory limits for groundwater, but also below those for drinking water. For this purpose, LCA is used to calculate the environmental impacts of the different phases of the project's life cycle. Then, eLCC is followed to account for the costs incurred, using EP to monetize external costs and benefits in a top-down approach. The second innovation of this paper is to illustrate how external cost and benefits can be calculated in a bottom-up perspective. In this respect, the IPA framework is followed in a full scale hypothetical ISMP case study focusing on arsenic.

The ultimate goal of this work is to provide a working methodology for assessing the sustainability potential of a remediation project from a socio-economic perspective. Therefore, the proposed methodology is a complementary tool that i) is not intended to relieve the responsibility to remediate a contaminated site when it may not be socio-economically profitable, but legally is required, and ii) does not replace the environmental risk assessment guidelines established by national or regional legislation for decision-making on the need to remediate or not to ensure the quality of groundwater as a resource.

2. Materials and methods

2.1. Case study description and assumptions

The case study used in this paper is based on a real contaminated site situated in an area in Flanders (Belgium) with a large history of metallurgical activity and groundwater affected by elevated metal(loid) concentrations including As, Ni, Fe and Zn. As a potential treatment measure for metal(loid)s removal from groundwater, biological ISMP via biostimulation of aquifer native SRB is considered. By adding an organic substrate, the activity of SRB is biostimulated and sulfate is reduced to sulfide, which in turns reacts with metal(loids) in solution to form insoluble meta(loid)sulfide minerals. This treatment has been tested at field scale as part of an EU research and innovation action aimed at achieving zero pollution in Europe (GREENER Grant 826312). The long-term remediation potential of this approach was successfully demonstrated previously (Pérez-de-Mora et al., 2024).

For the purposes of this study the following assumptions are made: i) the contamination plume can reach drinking water resources beyond the site boundaries if no active remediation/containment measures are implemented, ii) the reservoir supplies 100 % of the total drinking water needs of a population of 18,000 habitants located a few kilometres away from the industrial site, iii) despite the high concentration of various metal(loid)s in groundwater, arsenic is the only contaminant considered to the illustration of the IPA case study, and iv) arsenic concentration is reduced below the drinking water limit (10 μg/L), which is half the value of the regulatory limit for groundwater (20 μg/L), to improve the socio-economic impact of the remediation (Government of Flanders, 2008).

2.2. Environmental impact assessment

The environmental impacts of the ISMP case study are calculated following the attributional Life Cycle Assessment (LCA) method according to the International Standards Organization (ISO) 14,040 and 14,044 (ISO, 2006a, 2006b) which consists in four steps. The first step is aimed at defining the goal and scope of the study. The second step consists of the inventory analysis, in which all data necessary for the impact assessment are gathered for the process under study and for the background system to create the Life Cycle Inventory (LCI). In the third step, the LCA results are expressed in midpoint indicators. For that, SimaPro 9.5.0.0, a widely applied LCA software, is used (PRé Sustainability B.V., 2024). Ecoinvent 3.9.1, the primary database for LCA, serves as the background data source (Wernet et al., 2016). The system model "allocation, cut-off by classification", is applied, as it is the default system where the waste burdens are associated to the producer. The fourth and final step consists in the interpretation of the results from the previous three steps.

2.2.1. Goal and scope

The functional unit is set to the treatment of 10 m^3 of contaminated groundwater during the 2 years that the technology was tested at field scale. The system boundaries are from cradle to grave, as the whole project life cycle is assessed.

2.2.2. Life Cycle Inventory (LCI)

The data needed for the LCI, meaning the exhaustive inventory of all the inputs and all the outputs of the studied system, was obtained from the company TAUW. This company implemented the ISMP technology in the EU H2020 project GREENER (Grant No. 826312) at field scale during 2 years. The data for the implementation and the results was collected directly by the company and updated at the end of the project in 2023. All the data collected and the complete LCI can be found in Tables S1 and S2 of the Supplementary material.

For the inputs and outputs not directly available in the ecoinvent database, an approximation with a close compound or a creation of a model based on the data provided by TAUW was performed to model all the steps of the technology (Table S2). For instance, consumables such as monitoring well piping, bottles and tubing for sampling were modelled after estimation of their diameter, weight and length.

The flowchart of the ISMP implemented at field scale during 2 years is shown in Fig. 1 and consists of three main steps: i) initial sampling to quantify the contamination of the site and determine the area where ISMP needs to be implemented, which involves collecting samples using filters, plastic tubing, and PE bottles, with the polluted water sent to a lab for analysis, ii) construction and implementation, which includes the installation of 1″ monitoring wells using materials like bentonite, gravel, and stainless steel, iii) operation and maintenance, involving periodic injections of organic substrates and other reagents into the groundwater, along with regular sampling to monitor and evaluate the progress of the bioremediation process over the project's lifetime.

The waste management of each product is included directly in the stage where the product is used. For instance, the end of life of the concrete and gravel used as backfill of the boreholes as the wells are dismantled is included in the construction phase.

2.2.3. Environmental impact assessment methods for LCA

In the present work, the environmental impact assessment for the LCA results are calculated by two different methodologies: i) Environmental Footprint (EF 3.0), developed and recommended by the European Commission to measure and communicate the environmental performance of products and organizations in the European Union (Fazio et al., 2018; Manfredi et al., 2012), and ii) ReCiPe 2016, developed by a consortium of Dutch research institutes to provide a harmonised and consistent framework for impact assessment at midpoint and endpoint level (Huijbregts et al., 2017). The purpose of providing the results for both methods is to compare them in Section 3.2.1 to observe the differences in the monetary valuation of LCA, used in the socio-economic evaluation of the technology implemented at field scale.

2.3. Socio-economic impact assessment

The socio-economic analysis performed in this work is focused on the inclusion of externalities in the economic assessment by drawing from

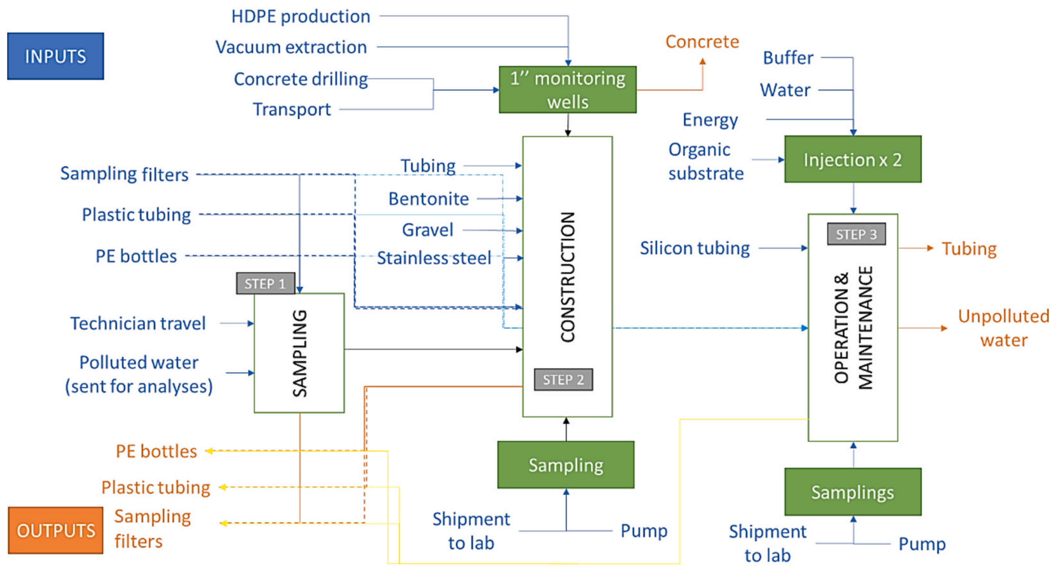

Fig. 1. Flowchart of inputs and outputs for the In-Situ Metal Precipitation process steps implemented at field scale. Step 1: initial sampling of the contaminated site to determine the pollution levels. Step 2: construction and installation of the monitoring wells. Step 3: operation and maintenance phase considering the injections of the substrates into the groundwater.

environmental economics principles. In this field, Environmental Management Accounting (EMA) appears as a sustainability assessment method that incorporates the monetary valuation of environmental impacts into the economic assessment (Jasiński et al., 2021). There are five main tools in the EMA methodology: Life Cycle Costing (LCC), Full Cost Accounting, Cost–Benefit Analysis (CBA), Balanced Scorecard for Sustainability, and Material Flow Cost Accounting (Qian and Burritt, 2008). Among them, LCC and CBA are used in this study to evaluate the socio-economic impacts of the ISMP implementation, including external costs and benefits in order to integrate these sustainability tools together with LCA. A summary of the methodologies performed in this paper is shown in Fig. 2.

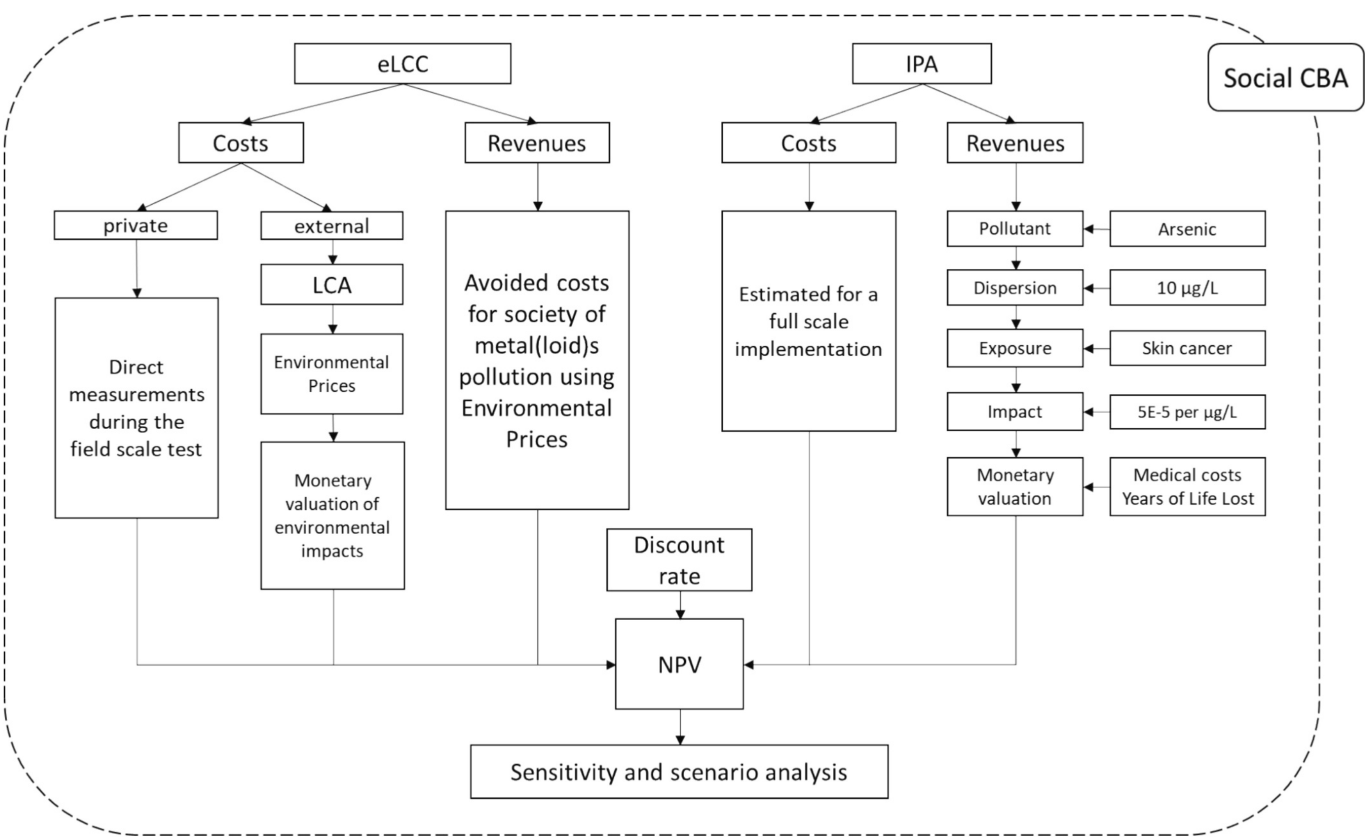

Fig. 2. Diagram of the steps followed in this study to calculate the social Cost-Benefit Analysis (social CBA) using environmental Life Cycle Costing (eLCC) and Impact Pathway Approach (IPA). For eLCC, external costs are calculated considering Life Cycle Assessment (LCA) results using the Environmental Prices values. The Net Present Value (NPV) is the main financial indicator used to determine the socio-economic viability in both approaches.

LCC is a methodology that aims to include the total costs of a product throughout its life cycle including capital investment, purchase and installation costs, and future costs such as energy, maintenance and operating costs over the lifetime of the project, product or service (Fuller, 2005; Swarr et al., 2011). There are three different types of LCC can be defined: conventional, environmental and social. While conventional LCC focuses only on internal costs, environmental LCC (eLCC) follows the four steps indicated in the ISO standard for LCA, including external costs of the process, and social LCC extends the boundaries of the previous and includes all the external costs and social impacts of the system associated with the study (Hunkeler et al., 2008). In this study, the eLCC approach is applied because the environmental impacts obtained from the LCA are monetized and included as costs in the life cycle of the project.

This is extended by applying a social CBA, a tool that evaluates the attractiveness of projects by considering internal and external economic, environmental, and social concerns, by calculating the Net Present Value (NPV) of a project by applying a discount rate to cash flows over a given period (Arler, 2006; Hoogmartens et al., 2014; Johansson, 1993). The calculation of the NPV allows to aggregate the revenues and costs that occur over a predefined time horizon (t) as the Cash Flows (CF), using a chosen discount rate (r_t) to account for the depreciation of the value of money over time (Eq. (1)):

$$NPV = \sum_{t=0}^{T} \frac{CF_t}{(1 + r_t)^t} \tag{1}$$

A positive NPV means that the project is profitable, whereas if it is negative, the project should be rejected and redesigned, as it indicates that it is not profitable under the established conditions. In this case, a sensitivity analysis is performed to identify the hotspots that affect the economic viability and determine different scenarios to make the project profitable.

One of the key issues in all types of economic valuations is the choice of an appropriate discount rate. Discounting is a method used to determine the present value of future CFs. It is crucial in CBA for evaluating social investments, mainly because discounting reflects the balance between present and future welfare (Philibert, 2006). A financial discount rate (private discount rate) or a social discount rate can be used, depending on the valuation perspective (Boardman et al., 2011; OECD, 2018; Van der Kamp, 2017). Governments and experts suggest ranges of social discount rates that vary between different countries (European Commission, 2014; Freeman et al., 2020; Groom et al., 2022; Mouter, 2018). Recently, the European Commission (2021) estimated an average social discount rate in the EU of 3.6 %. Based on this estimation and following the study performed by Huysegoms et al. (2018), the discount rate used in this study is set at 4 %. The choice of the discount rate is further discussed in Section 4.3.

2.3.1. Private costs

The inventory with the private costs, which is showed in Table S3, is provided by the company TAUW based on the real costs of the field pilot demonstration performed at the industrial site during the EU H2020 research project GREENER (Grant 826312) (Pérez-de-Mora et al., 2024). Due to confidentiality issues the data is aggregated for the different stages of the implementation, which is a common practice in the evaluation of research projects where companies limit their data to be published (Bachmann et al., 2024; Spierling et al., 2018). In the first stage, all costs related to the initial characterisation sampling phase are considered, including transportation and laboratory experiments. In the construction stage, the costs of drilling the injection and monitoring wells, raw materials, labour, machinery, transportation and disposal costs are collected. The operating costs include all costs related to the coordination of activities during the life of the project, including the injection of the organic substrates, the collection and analysis of periodic samples and the disposal of waste at the end of the project. Finally, general costs are estimated to include coordination and administration and other indirect costs associated with the project. Due to the small size of the wells used for injection and sampling (1 in.), there are no end-of-life costs as the wells can be left in place and sealed when no longer needed. Additionally, uncertainty of the input parameters is indicated in Table S3 in the column data quality indicator, where it is specified if the data was measured, calculated or estimated.

2.3.2. External costs and benefits

The monetization of external costs used in social CBA is related to welfare and environmental economics. Externalities are defined as effects caused by one agent that affect others and are not accounted for because they occur as side effects of economic activity (Hunkeler et al., 2008; Pearce and Barbier, 2000) and therefore create market failures if they are not included in the costs and prices of the products, services or processes (Pigou, 2017). In this sense, the monetary valuation of environmental impacts can be done through different approaches (e.g. budget constraint, abatement costs and damage costs) and methods (e.g. contingent valuation, hedonic pricing or stated preferences) (Amadei et al., 2021; Atkinson and Mourato, 2008; De Zeeuw et al., 2008; Pizzol et al., 2015). The monetary valuation of environmental and social impacts can be a questionable method to show the external costs and benefits of a project due to the subjectivity and uncertainty of the methodologies that can be applied, as well as the oversimplification of complex data of environmental impacts, or social aspects into monetary values, as it can mean the reduction of the environment to a market dimension (Gluch and Baumann, 2004). However, it is a practice generally included in social CBA and accepted by various public institutions to evaluate decision-making for alternatives, considering the sustainability of the innovations (De Zeeuw et al., 2008).

Currently, there is no consensus on which monetary valuation method to use in an LCA study, as the monetary valuation coefficients differ depending on the method used. Nonetheless, damage cost and abatement cost are the most commonly used approaches (Amadei et al., 2021). These methods calculate the marginal value of a good that is not in the market by using "willingness to pay" to estimate the demand for environmental quality, i.e. how much of their income are people willing to sacrifice for an additional unit of environmental quality (de Bruyn et al., 2023).

In this study, the methodology used for the monetization of the LCA to be used in the social CBA is Environmental Prices (EP). This methodology is chosen because it shows transparency in the calculation of the monetary valuation coefficients used to calculate the social damage of environmental pollution, expressing it in euros per kilogram of pollutant, offering a range for each of the compounds due to the uncertainties in the calculations used to determine these environmental charges, which are used in the sensitivity analysis (de Bruyn et al., 2023). This methodology has been fully developed for the ReCiPe Midpoint (H) impact results and some impact categories are currently being calculated for EF 3.0 as can be seen in Tables S4 and S5. The calculation of the EP follows the Impact Pathway Approach (IPA), that is explained in Section 2.3.3. The monetary valuation of damages is mainly based on medical costs, willingness to pay, and replacement costs, which are different depending on the conditions (emission source, population density, etc.) and the regions where a specific pollutant is assessed. For instance, EP are calculated specifically for different contaminants in air, water and soil in the Netherlands, and provides average values for the 27 European state members. These values differ because the same contaminant can have different impacts on human health depending on whether the intake is through ingestion (water, food) or inhalation (air). Additionally, costs are different as people's willingness to pay depend on their incomes and medical costs are different between regions. All this information is gathered and explained in the Environmental Prices Handbook (de Bruyn et al., 2023).

In this case study, benefits for groundwater bioremediation are calculated using non-market values and EP is applied because the method provides the cost to society of specific substances in air, soil, freshwater and seawater (de Bruyn et al., 2023). In the case study, EP coefficients are taken for freshwater, based on the fact that groundwater accounts for 99 % of the total freshwater in the world (UNESCO, 2022). These coefficients (Table S6) are applied directly to the amount of pollutant removed to obtain a monetary value. To illustrate how these benefits are calculated for a specific region, the IPA steps described in Section 2.3.3 will be followed.

2.3.2.1. Sensitivity and scenario analysis. When estimating the financial analysis, especially to long-term projects and including externalities, uncertainties should be studied. Sensitivity analysis can identify critical parameters that affect the estimated viability of the projects at different levels. After the evaluation of the socio-economic viability of the project under the baseline scenario (treatment of 10 m^3 of contaminated groundwater at field scale during 2 years), sensitivity analysis was performed to identify the factors that have the greatest effects on the NPV (CRC CARE, 2019). A one-at-a-time sensitivity analysis was conducted assuming a $+-10$ % variation of the different input parameters aggregated for each project stage (Initial sampling, construction, operation and general costs) to observe the variation of the NPV. This is adapted from the recommendation of the European Commission (2014) to identify the critical variables of the project. Then, a bivariate analysis is performed to identify the scenarios in which the project is socio-economically profitable.

2.3.3. Impact Pathway Approach

In contrast to the methods used to account for the externalities using data generalised at European level (Environmental Prices), a bottom-up perspective can be taken by following the IPA to calculate the monetary values of external costs and benefits in a specific region and population. The IPA framework was first developed within the ExternE projects, a series of research projects funded by the European Commission from 1991 to 2005 (Bickel, Friedrich, et al., 2005), focused on the external costs of energy production and use, such as the environmental and health impacts of air pollution, climate change, noise, accidents, etc. The project series adopted the IPA for the assessment of the externalities and associated costs resulting from the supply and use of energy following five steps: i) identification of a burden (e.g., emissions), ii) its dispersion, iii) the exposure of the population, iv) the impact (e.g., on human health) and a subsequent v) monetary valuation (Friedrich and Bickel, 2001). With these five stages, it is possible to estimate the costs associated with an impact or the benefits of eliminating a pollutant (ETC/ATNI, 2021). This framework, used in the calculation of EP for Europe and the Netherlands (de Bruyn et al., 2023), will be followed and adapted to illustrate in a hypothetical case study the estimation of the external benefits of eliminating arsenic in groundwater as a source of drinking water.

2.3.3.1. Identification of the pollutant. Arsenic has been identified as the main pollutant as in groundwater, inorganic forms of arsenic (As(III), As (V), or a mix of both) are common, while organic forms resulting from biological processes are infrequent in water (Fatoki and Badmus, 2022). According to this information, it is determined that most of the ingestion dose of arsenic in drinking water is inorganic (Bickel, Friedrich, et al., 2005), and that food and drinking water are the principal routes of exposure to arsenic (FAO/WHO, 2011; IARC, 2012) that can be caused by the groundwater contamination (Alam et al., 2021).

2.3.3.2. Dispersion. The dispersion has been quantified considering different international standards and regulatory limits for acceptable levels of arsenic in food and drinking water. In Europe, the European Commission has set a limit of 10 micrograms per litre (μg/L), without differentiating between the different forms of arsenic (European Commission, 2020). However, the maximum arsenic levels found in some European countries are higher than the limits established by the EC (Khosravi-Darani et al., 2022) and there is also an uncertainty about the safety of this limit (10 μg/L) for human health (Ahmad and Bhattacharya, 2019). In fact, in some European countries, such as Germany, this limit will be lowered to 4 μg/L in 2028 (Federal Ministry of Health, 2023).

2.3.3.3. Exposure. Inorganic arsenic is a human carcinogen, based on convincing epidemiologic evidence of a causal relationship between human oral exposure to inorganic arsenic and cancer (Arcella et al., 2021; ECHA, 2013; U.S. EPA, 2010). Inorganic As, especially As(III), is readily absorbed from the gastrointestinal tract to the blood. The impact of this substance on human health is important due to its toxic effects (Mochizuki, 2019), and skin lesions are the most common human health effects of long-term exposure to high levels of inorganic arsenic by ingestion, which are considered precursors of non-melanoma skin cancer (Food and Authority, 2014; Howard, 2003; Lin et al., 2022).

2.3.3.4. Impact. According to the U.S. Environmental Protection Agency (EPA) (2010), the cancer unit risk for drinking water contaminated with arsenic is 5E-5 per μg/L, which indicates the additional cancer cases expected for each microgram of arsenic per litre of drinking water over a lifetime of 70 years.

To simplify the calculations for the impact in this case study, it is assumed that the amount of arsenic in drinking water is reduced from the current legal limit of 10 μg/L to 0.05 μg/L, using the same efficiency of the ISMP technology in reducing arsenic (99.5 %) as observed in Table 5. It is also assumed that groundwater is the main source of drinking water for the population living in the vicinity of the contaminated area, which is consistent with the fact that groundwater meets approximately 40 % of the drinking water needs in Belgium (EurEau, 2021).

2.3.3.5. Monetary valuation. To monetize these figures and estimate the avoided social costs (benefits), the medical costs for the treatment of skin cancer in Belgium and the Years of Life Lost (YLL) are considered. Among other metrics for integrating social aspects in CBA, YLL is used to quantify a reduction in the expected life span of populations or individuals due to specific causes (Weidema, 2006).

3. Results

3.1. Environmental impact results

The LCA results are the starting point for the social CBA of implementing the ISMP case study at field scale. As indicated in Section 2.2.3, EF 3.0 and ReCiPe methods were used to evaluate the impacts of the ISMP technology that will be used later for the monetary valuation of these results. The results obtained cannot be directly compared as there are substantial differences between the methodologies. For instance, while EF 3.0 covers 16 impact categories, ReCiPe 2016 covers 18 categories. Terrestrial and marine ecotoxicity are unique to ReCiPe 2016 and some of the impact categories that are common to both methodologies are expressed in different units (e.g. ionising radiation, land use, ecotoxicity, etc.).

From Figs. 3 and 4 it can be observed that unpolluted water is represented as impacts below 0, which means it is a positive impact on the environment. This is due to the fact that depollution has been modelled as the removal of pollutant emissions (see Table S1). In both methods, unpolluted water is related to the non-carcinogenic toxicity, although the calculations are different to determine this impact category.

The main contributor to the total environmental impact regardless of the method used (EF 3.0 or ReCiPe) is “Step 3: operation and maintenance”. This is because all injection steps and samplings are included in the calculation and represents from 52 % to 100 % of the total impact for

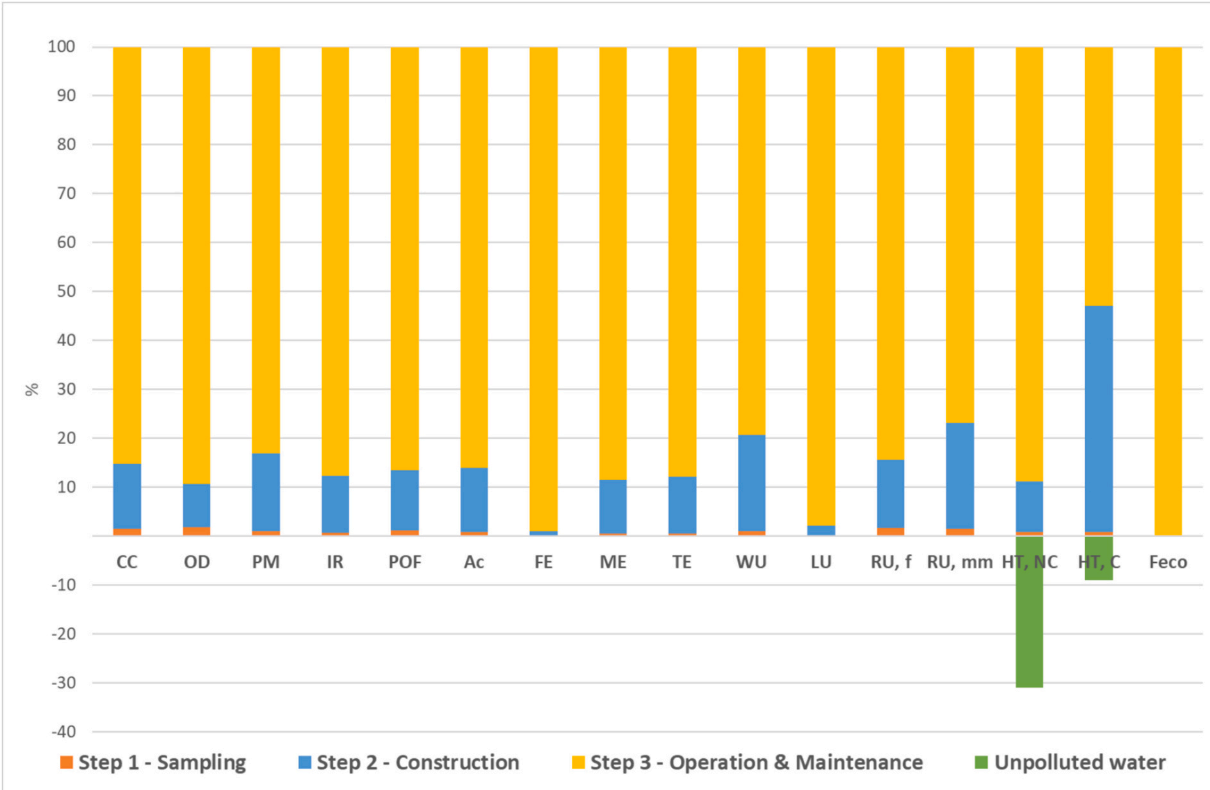

Fig. 3. Midpoint impact category contributions for each step of the In-Situ Metal Precipitation (ISMP) life cycle using EF 3.0. Note: Climate change (CC), ozone depletion (OD), Particulate matter (PM), ionising radiation (IR), photochemical ozone formation (POF), acidification (Ac), freshwater eutrophication (FE), marine eutrophication (ME), terrestrial eutrophication (TE), water use (WU), land use (LU), resource use fossils (RU, f), resource use minerals and metals (RU, mm), human toxicity non-cancer (HT, NC), human toxicity cancer (HT, C), freshwater ecotoxicity (Feco). Unpolluted water represents positive environmental impacts.

the EF 3.0 method and from 35 % to almost 100 % for the ReCiPe method. To explore this issue in more depth, the impacts in Step 3 are disaggregated in Fig. 5 using EF 3.0. It can be noted that the use of Potassium carbonate (K_2CO_3) has the major contribution due to the large quantity needed during the operation of the project (300 kg) and also its toxicity effects on freshwater (Maul et al., 2014). In addition, organic substrate (EOS PRO) contributes to around 90 % of the impacts in the eutrophication of freshwater and land use due to the use of soybean oil and water needed to produce the substrate as indicated in the inventory (Table S2).

The complete ecoprofiles for cleaning up of 10 m^3 of polluted water are shown in Tables 1 and 2. The results for the common impact categories using the same units can be compared, i.e. climate change/global warning, ozone depletion, freshwater and marine eutrophication.

As can be observed in Table 3, although these impact categories are measured in the same units, the results are similar only for the kg CO2-eq quantities. Otherwise, the differences are due to the geographical scope of both methodologies. For instance, ReCiPe provides characterisation factors that are representative at global scale while EF3.0 is tailored for European conditions and the characterisation factors are calculated and intended to their use at European scale (Huijbregts et al., 2017; Manfredi et al., 2012).

3.2. Socio-economic impact results

3.2.1. Costs and benefits calculation

Once the environmental impacts are calculated, cost and benefit data are gathered and treated following the same functional unit as in LCA. The inventory of the (private) internal costs is shown in Table 7, provided directly by the company TAUW and aggregated for the different stages of the project, applying a discount rate of 4 % for the costs and benefits not incurred during the first year.

The calculation of external costs follows the same functional unit as in the LCA, the treatment of 10 m^3 of contaminated groundwater during two years. This allows to translate the environmental impacts obtained from the LCA calculations into monetary values for the two environmental impact assessment methods used in EP (Table 4).

The environmental benefits obtained from the unpolluted water LCA calculations are translated into monetary revenues, taking into account externalities. The difference between the results of the two methods is due to the fact that there are currently only monetary values for 9 impact categories in EF 3.0, whereas there are monetary values for all 18 impact categories for ReCiPe. We could restrict the results to the values obtained from the four categories that have common units in both methods as can be seen in Tables S4 and S5. However, for better consistency in the analysis, all the impact categories are considered so the LCA results are those calculated using the ReCiPe 2016 method due to its completeness. The central values obtained are taken for the baseline scenario.

To calculate the benefits, we first estimate the total grams of heavy metals removed in the functional unit based on the data from TAUW (Table 5) collected during the sampling phase at the contaminated site.

Second, the benefits are obtained by multiplying the total grams removed by the EP factors (Table S6) for heavy metals in freshwater.

From Table 6 it can be observed that Zn has the higher values of all the pollutants since it is present in higher concentration (see Table 5). However, Zn is an essential micronutrient for human health and it is only considered harmful only in high ingestion doses (Nriagu, 2007). Based on this fact, and on the EP values in Table S6, As is assumed in this study to be the most impactful metal(loid) and is therefore further analysed in the IPA.

The revenues considered for the baseline scenario are those

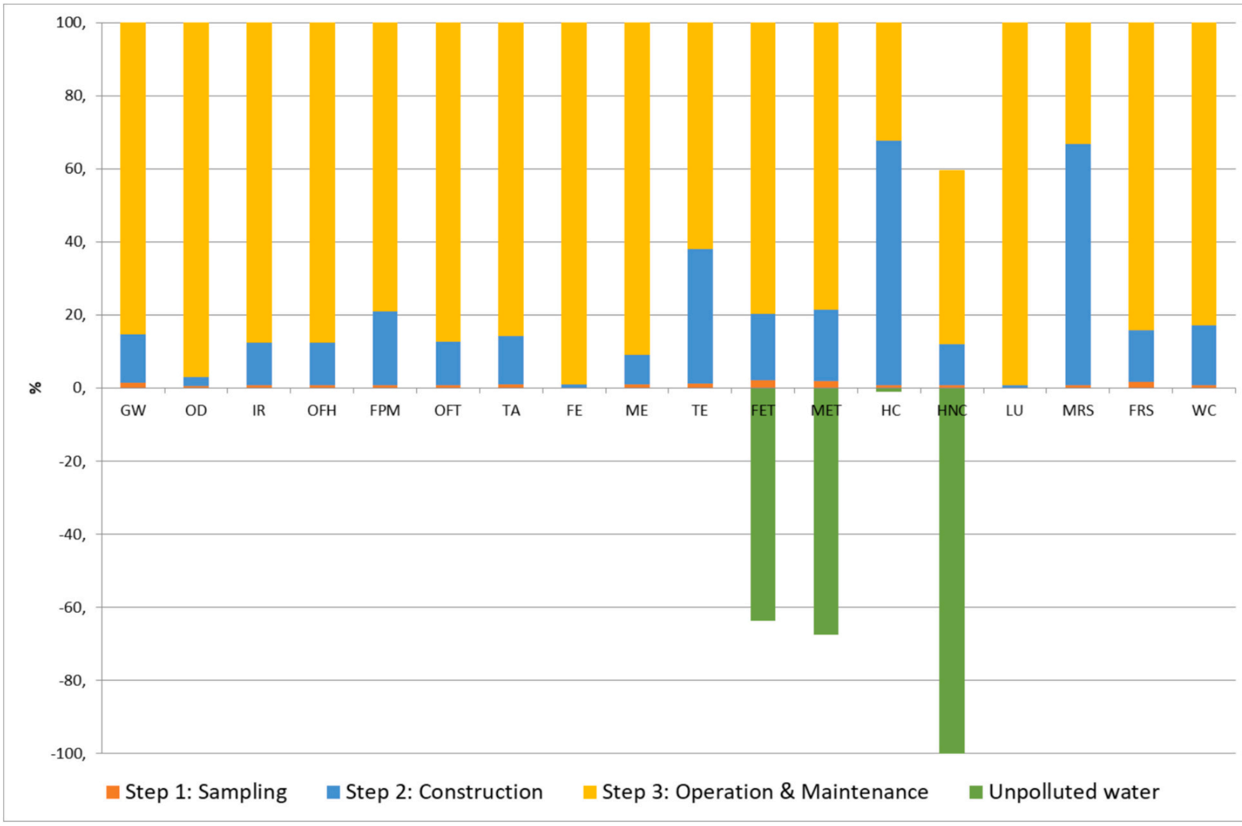

Fig. 4. Midpoint impact category contributions for each step of the In-Situ Metal Precipitation (ISMP) life cycle using ReCiPe 2016 Midpoint (Hierarchist model). Note: Global warming (GW), stratospheric ozone depletion (OD), ionizing radiation (IR), ozone formation human health (OFH), fine particulate matter formation (FPM), ozone formation terrestrial ecosystems (OFT), terrestrial acidification (TA), freshwater eutrophication (FE), marine eutrophication (ME), terrestrial ecotoxicity (TE), freshwater ecotoxicity (FET), marine ecotoxicity (MET), human carcinogenic toxicity (HC), human non-carcinogenic toxicity (HNC), land use (LU), mineral resource scarcity (MRS), fossil resource scarcity (FRS), water consumption (WC). Unpolluted water represents positive environmental impacts.

calculated with the central values. Using Eq. (1), the NPV for the baseline scenario in this case, considering the treatment of 10 m^3 of contaminated water at field scale during 2 years, is −129,512.61 €. This means that the project is not socio-economically profitable in this scenario, as the private and external costs are higher than the revenues. This does not mean that society should not invest in the technology based on these results. There are other benefits that can be also added to the financial analysis and other scenarios that should be examined to determine whether or not the technology can be profitable at any point. To evaluate a potential scenario where the project is profitable, a sensitivity analysis is performed (see next section).

3.2.1.1. Sensitivity and scenario analysis. This section focuses on the analysis of how the inputs affect the NPV and under which scenario the technology can be socio-economically profitable. The new scenario for the economic analysis of ISMP follows the CBA recommendations of the EC (European Commission, 2014), extending the life of the project from 2 to 30 years and using a discount rate of 4 % to calculate the NPV for the different cost scenarios using the upper, central and lower bounds indicated in the EP. The NPV using the lower bounds for the external costs and revenues is −173,466.61€, while using the upper bounds the NPV is −159,559.02€. This is due to the fact that upper values for external costs do not influence the NPV in the same way as the upper values for external revenues do as can be deduced from Tables 4 and 6. Therefore, as the upper limits may achieve a positive NPV treating a lower quantity of water per year than using the central and lower limits, the upper NPV scenario is used in the next steps of the sensitivity analysis.

The next step is to identify the variables that most influence the NPV result. The sensitivity analysis is performed by varying the inputs using a coefficient of variation of 10 % for the initial identification of the input costs aggregated by project stage (Initial sampling, construction, operation and general costs). The Functional Unit (FU) is also included as one of variable because the quantity of water treated is directly related to the social benefits. The results are presented in a tornado diagram in Fig. 6.

The operating costs is the most critical variable in the project and therefore any variation will have a greater impact on the NPV. In order to identify the point at which the project is acceptable, a bivariable sensitivity analysis is performed by assuming a variation of the operation costs by − + 60 % and increasing the volume of water treated per year up to 100 m^3.

If the operating costs remain the same, the project is socio-economically profitable if the amount of water treated is increased to 90 m^3 per year during 30 years, but if the operating costs can be reduced in a 60 %, the amount of water needed to be treated per year is 60 m^3.

In addition, it is evident that the construction costs are the inputs that affect the NPV variation the less, but if we consider an increase in the amount of water to be treated to increase the benefits, the construction costs will be higher as more injection wells will be needed. For this reason, the second NPV sensitivity analysis is carried out for the relationship between an increase in the construction cost and the amount of water treated per year.

In this case, it was assumed that the construction costs cannot be reduced due to the increase in the amount of water treated, and the NPV is positive if the amount of water treated per year for 30 years is higher than 90 m^3.

This analysis, applying EP to the LCA results and for calculating the benefits, is useful to evaluate the innovative technologies from a field

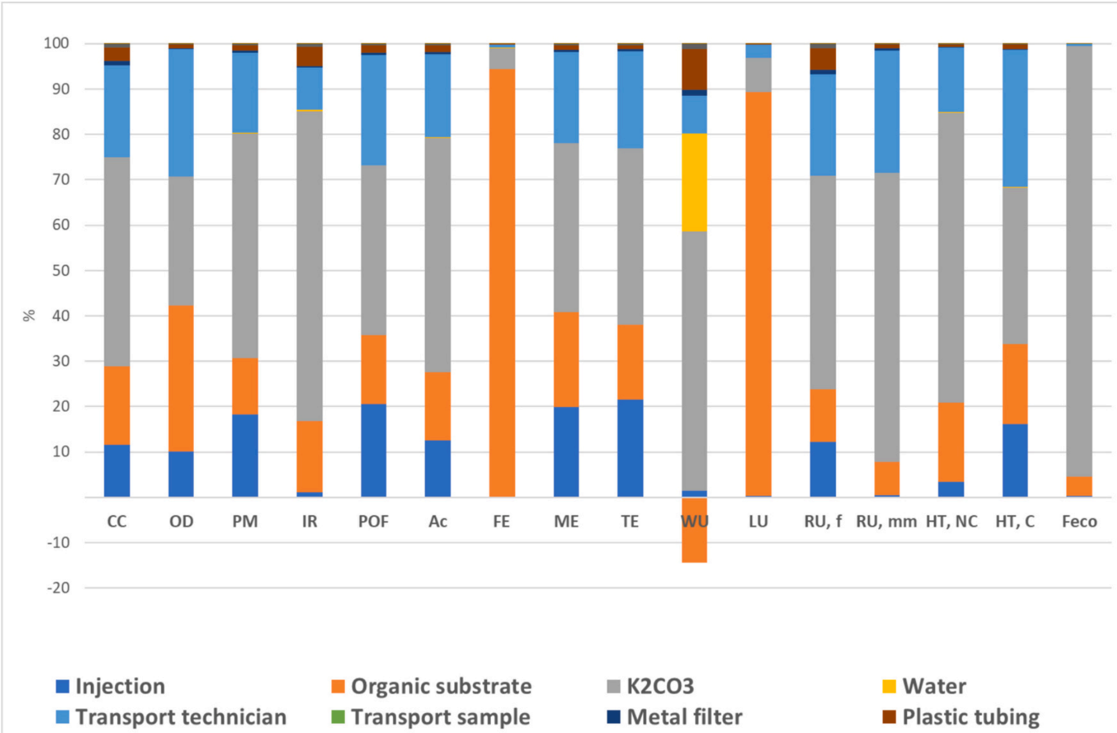

Fig. 5. Main contributions to step 3 (Operation & Maintenance) using EF 3.0. Note: Climate change (CC), ozone depletion (OD), particulate matter (PM), ionising radiation (IR), photochemical ozone formation (POF), acidification (Ac), freshwater eutrophication (FE), marine eutrophication (ME), terrestrial eutrophication (TE), water use (WU), land use (LU), resource use fossils (RU, f), resource use minerals and metals (RU, mm), human toxicity non-cancer (HT, NC), human toxicity cancer (HT, C), freshwater ecotoxicity (Feco).

Table 1
Environmental impacts for each step of the In-Situ Metal Precipitation (ISMP) life cycle using EF 3.0 for 10m^3 of contaminated groundwater.

Impact category	Unit	Step 1: Sampling	Step 2: Construction	Step 3: Operation & Maintenance	Unpolluted water	Total
Climate change	kg CO2-eq	7.32E+01	6.30E+02	4.07E+03	0.00E+00	4.77E+03
Ozone depletion	kg CFC11 eq	1.59E−06	7.31E−06	7.42E−05	0.00E+00	8.31E−05
Particulate matter	disease inc.	3.04E−06	4.39E−05	2.30E−04	0.00E+00	2.77E−04
Ionising radiation	kBq U-235 eq	2.19E+00	3.24E+01	2.46E+02	0.00E+00	2.80E+02
Photochemical ozone formation	kg NMVOC eq	2.39E−01	2.54E+00	1.78E+01	0.00E+00	2.06E+01
Acidification	mol H+ eq	1.98E−01	2.91E+00	1.92E+01	0.00E+00	2.23E+01
Eutrophication, freshwater	kg P eq	9.38E−03	1.47E−01	1.46E+01	0.00E+00	1.48E+01
Eutrophication, marine	kg N eq	3.78E−02	6.68E−01	5.39E+00	0.00E+00	6.10E+00
Eutrophication, terrestrial	mol N eq	3.84E−01	7.05E+00	5.39E+01	0.00E+00	6.13E+01
Water use	m3 depriv.	9.44E+00	1.94E+02	7.76E+02	0.00E+00	9.79E+02
Land use	Pt	3.22E+02	2.86E+03	1.44E+05	0.00E+00	1.47E+05
Resource use, fossils	MJ	9.76E+02	8.34E+03	5.01E+04	0.00E+00	5.94E+04
Resource use, minerals and metals	kg Sb eq	6.80E−04	9.46E−03	3.36E−02	0.00E+00	4.37E−02
Human toxicity, non-cancer	CTUh	8.93E−07	1.15E−05	9.78E−05	−3.41E−05	7.61E−05
Human toxicity, cancer	CTUh	4.92E−08	2.38E−06	2.72E−06	−4.65E−07	4.68E−06
Ecotoxicity, freshwater	CTUe	5.15E+02	2.71E+03	1.16E+06	−6.28E+02	1.17E+06

scale, identify hotspots and estimate different scenarios for the upscaling process. To illustrate how the monetary valuation coefficients can be calculated, in the next section the IPA is followed to calculate the human health benefits of arsenic removal considering a population living near the contaminated site.

3.2.2. *Impact Pathway Approach for arsenic in groundwater*

In the previous sections, the socio-economic viability of the ISMP case study has been evaluated following a top-down approach. In this part, social CBA will be calculated in a bottom-up perspective following the IPA. The inventory in this case is subject to more assumptions and therefore more uncertainty. Knowing the amount of arsenic in tap water supplied to households from the groundwater is not possible without the direct measurement of the supply. The amount of contaminant used as a reference in the calculation of benefits in the previous section cannot be applied to the IPA. Thus, in the hypothetical case study, the ISMP eliminates 99.5 % of the arsenic concentration in the groundwater, lowering the 10 μg/L (legal limit) of arsenic assumed to be present in the drinking water and thus, avoiding 9.95 μg/L for the nearby population. Authors are aware that these values may not correspond to the technical capacity to carry out this reduction of arsenic in a full scale ISMP and further technical developments should be necessary. However, we would like to remark that this case study serves to illustrate how to carry out a social CBA from a bottom-up approach, focused on the calculation of social benefits.

Following the IPA steps described in Section 2.3.3, we apply the cancer unit risk to the amount of arsenic avoided per litre of drinking water (0.00005*9.95). The risk of cancer is therefore reduced in

Table 2
Environmental impacts for each step of the In-Situ Metal Precipitation (ISMP) life cycle using ReCiPe 2016 Midpoint (H) method for 10m^3 of contaminated groundwater.

Impact category	Unit	Step 1: Sampling	Step 2: Construction	Step 3: Operation & Maintenance	Unpolluted water	Total
Global warming	kg CO2 eq	7.47E+01	6.43E+02	4.19E+03	0.00E+00	4.90E+03
Stratospheric ozone depletion	kg CFC11 eq	3.54E−05	1.81E−04	6.79E−03	0.00E+00	7.01E−03
Ionizing radiation	kBq Co-60 eq	1.92E+00	2.81E+01	2.13E+02	0.00E+00	2.43E+02
Ozone formation, Human health	kg NOx eq	1.09E−01	1.73E+00	1.29E+01	0.00E+00	1.47E+01
Fine particulate matter formation	kg PM2.5 eq	5.83E−02	1.47E+00	5.78E+00	0.00E+00	7.31E+00
Ozone formation, Terrestrial ecosystems	kg NOx eq	1.24E−01	1.81E+00	1.34E+01	0.00E+00	1.53E+01
Terrestrial acidification	kg SO2 eq	1.33E−01	1.89E+00	1.21E+01	0.00E+00	1.41E+01
Freshwater eutrophication	kg P eq	1.39E−02	2.29E−01	2.39E+01	0.00E+00	2.41E+01
Marine eutrophication	kg N eq	2.44E−03	2.12E−02	2.34E−01	0.00E+00	2.58E−01
Terrestrial ecotoxicity	kg 1.4-DCB	3.71E+02	1.15E+04	1.93E+04	−1.09E−14	3.12E+04
Freshwater ecotoxicity	kg 1.4-DCB	5.71E+00	5.10E+01	2.23E+02	−1.78E+02	1.02E+02
Marine ecotoxicity	kg 1.4-DCB	7.42E+00	7.28E+01	2.94E+02	−2.53E+02	1.22E+02
Human carcinogenic toxicity	kg 1.4-DCB	5.85E+00	5.93E+02	2.85E+02	−8.32E+00	8.76E+02
Human non-carcinogenic toxicity	kg 1.4-DCB	6.31E+01	1.02E+03	4.27E+03	−9.01E+03	−3.65E+03
Land use	m2a crop eq	1.70E+00	1.64E+01	2.61E+03	0.00E+00	2.63E+03
Mineral resource scarcity	kg Cu eq	3.70E−01	3.33E+01	1.67E+01	0.00E+00	5.04E+01
Fossil resource scarcity	kg oil eq	2.18E+01	1.82E+02	1.08E+03	0.00E+00	1.28E+03
Water consumption	m3	2.43E−01	4.68E+00	2.40E+01	0.00E+00	2.89E+01

Table 3
Variation of common impact categories according to the results of ReCiPe 2016 in relation to EF3.0.

Impact category	Unit	Step 1: Sampling	Step 2: Construction	Step 3: Operation & Maintenance	Total
Climate change/global warming	kg CO2 eq	0.57 %	0.53 %	0.52 %	0.53 %
Ozone depletion/stratospheric ozone depletion	kg CFC11 eq	−182.78 %	−184.46 %	−195.66 %	−195.30 %
Freshwater eutrophication	kg P eq	−39.14 %	−43.51 %	−48.13 %	−48.08 %
Marine eutrophication	kg N eq	175.74 %	187.70 %	183.35 %	183.77 %

Table 4
Monetized Life Cycle Assessment (LCA) results per project's phase using Environmental Prices (EP) for EF 3.0 and ReCiPe 2016 and considering 10m^3 of contaminated groundwater.

Phase	EF 3.0	ReCiPe 2016 lower	ReCiPe 2016 central	ReCiPe 2016 upper
Initial sampling	13.68 €	31.20 €	52.17 €	77.25 €
Construction	143.13 €	1810.89 €	2736.38 €	4102.99 €
Operation	948.53 €	2322.42 €	3819.01 €	5644.71 €
Unpolluted water	0.00 €	−457.98 €	−677.20 €	−1010.57 €
Total	1105.34 €	3706.53 €	5930.36 €	8814.38 €

Table 5
Calculation of the amount of metal(loid)s removed in 10m^3 based on the initial and final concentrations sampled.

Metal	Inlet	Outlet	Removal efficiency	Total grams removed in 10 m^3
As (μg/L)	2400	12	99,5 %	23.9
Cd (μg/L)	0.33	0.05	85 %	0.0028
Ni (μg/L)	350	5	99 %	3.48
Zn (μg/L)	84,000	1	100 %	840

Table 6
Revenues calculation for the total amount (grams) of metal(loid)s removed in 10m^3 using monetary values from Environmental Prices.

Pollutant	Lower cost	Central cost	Upper cost
As	5.56 €	78.52 €	370.00 €
Cd	0.00 €	0.00 €	0.00 €
Ni	0.05 €	0.17 €	0.64 €
Zn	5.42 €	205.80 €	1018.91 €
Total	11.03 €	284.49 €	1389.55 €

Table 7
Summary of costs and revenues of the In-Situ Metal Precipitation implementation at baseline scenario for treating 10m^3 of contaminated water in 2 years.

Impact	Phase	Cost discounted	Time of occurrence
Private costs	Initial sampling	−16,172.25 €	Within first year
	Construction	−15,510.00 €	Within first year
	Operation	−80,093.76 €	During 2 years
	General	−9430.47 €	During 2 years
External costs	Initial sampling	−77.25 €	Within first year
	Construction	−4102.99 €	Within first year
	Operation	−5323.23 €	Second year
Revenues	HM reduction	263.03 €	Second year
	Unpolluted water	934.33 €	

0.0004975 units and, considering a population of 18,000 inhabitants living in the vicinity of the contaminated area, this results in almost 9 skin cancer cases avoided over a lifetime.

To monetize the impact, it is assumed that the average duration of skin cancer is 5 years (Buekers et al., 2012) and the marginal reduction in life expectancy is estimated at 7.4 years (Steen, 2019). The average medical costs of treating skin cancer in Belgium were 750€ in 2014 (Pil et al., 2016) and applying the Consumer Price Index (Statista, 2023), this value currently increases to 920€. The indicator value used to monetize the YLL is 107,067€ per unit, calculated by Steen (2019). Therefore, the average medical cost for each skin cancer treatment is 4600€ and the total cost of one case for the YLL is 792,295.80€. Applied to the lifetime of the 9 cases in the total population, this results in 41,193€ for the medical costs and 7,130,662.20€ for the YLL (Table 10). As it is not possible to conclude at which time the cancer occurs, the medical costs avoided are considered as benefits at the first year because its economic valuation is calculated using monetary values from the present. Meanwhile, the external avoided costs (YLL) are considered at the end of the

Table 8
Net Present Value sensitivity analysis: Variation of operation costs (%) and amount of water treated annually (m^3 per year).

		Operation costs						
		−60 %	−40 %	−20 %	0 %	20 %	40 %	60 %
m^3 treated per year	10	−105,114.88	−123,262.93	−141,410.97	−159,559.02	−177,707.06	−195,855.10	−214,003.15
	20	−82,422.89	−100,570.94	−118,718.98	−136,867.02	−155,015.07	−173,163.11	−191,311.16
	30	−59,730.90	−77,878.94	−96,026.99	−114,175.03	−132,323.08	−150,471.12	−168,619.17
	40	−37,038.91	−55,186.95	−73,335.00	−91,483.04	−109,631.09	−127,779.13	−145,927.18
	50	−14,346.92	−32,494.96	−50,643.01	−68,791.05	−86,939.10	−105,087.14	−123,235.18
	60	8345.07	−9802.97	−27,951.02	−46,099.06	−64,247.10	−82,395.15	−100,543.19
	70	31,037.06	12,889.02	−5259.03	−23,407.07	−41,555.11	−59,703.16	−77,851.20
	80	53,729.05	35,581.01	17,432.97	−715.08	−18,863.12	−37,011.17	−55,159.21
	90	76,421.05	58,273.00	40,124.96	21,976.91	3828.87	−14,319.18	−32,467.22
	100	99,113.04	80,964.99	62,816.95	44,668.90	26,520.86	8372.81	−9775.23

Table 9
Net Present Value sensitivity analysis: Variation of construction costs (%) and amount of water treated annually (m^3).

		Construction costs						
		0 %	10 %	20 %	30 %	40 %	50 %	60 %
m^3 treated per year	10	−159,559.02	−161,183.97	−162,808.92	−164,433.87	−166,058.82	−167,683.77	−169,308.72
	20	−136,867.02	−138,491.97	−140,116.93	−141,741.88	−143,366.83	−144,991.78	−146,616.73
	30	−114,175.03	−115,799.98	−117,424.93	−119,049.88	−120,674.83	−122,299.79	−123,924.74
	40	−91,483.04	−93,107.99	−94,732.94	−96,357.89	−97,982.84	−99,607.79	−101,232.74
	50	−68,791.05	−70,416.00	−72,040.95	−73,665.90	−75,290.85	−76,915.80	−78,540.75
	60	−46,099.06	−47,724.01	−49,348.96	−50,973.91	−52,598.86	−54,223.81	−55,848.76
	70	−23,407.07	−25,032.02	−26,656.97	−28,281.92	−29,906.87	−31,531.82	−33,156.77
	80	−715.08	−2340.03	−3964.98	−5589.93	−7214.88	−8839.83	−10,464.78
	90	21,976.91	20,351.96	18,727.01	17,102.06	15,477.11	13,852.16	12,227.21
	100	44,668.90	43,043.95	41,419.00	39,794.05	38,169.10	36,544.15	34,919.20

Table 10
Summary of cost and revenues for implementing In-Situ Metal Precipitation at full scale.

Impact	Phase	Cost not discounted	Costs discounted	Time occurrence
Costs	Initial sampling	−20,000.00 €	−20,000.00 €	Within first year
	Design	−40,000.00 €	−40,000.00 €	Within first year
	Bidding	−30,000.00 €	−30,000.00 €	Within first year
	Construction	−163,000.00 €	−163,000.00 €	Within first year
	Operation/ maintenance	−350,000.00 €	−201,740.39 €	Thirty years
	Monitoring/ engineering	−670,000.00 €	−386,188.74 €	Thirty years
	Indirect/ unforeseen	−127,300.00 €	−73,375.86 €	Thirty years
Revenues	Medical costs	41,193.00 €	41,193.00 €	Within first year
	YLL	7,130,662.20 €	457,926.85 €	Seventy years

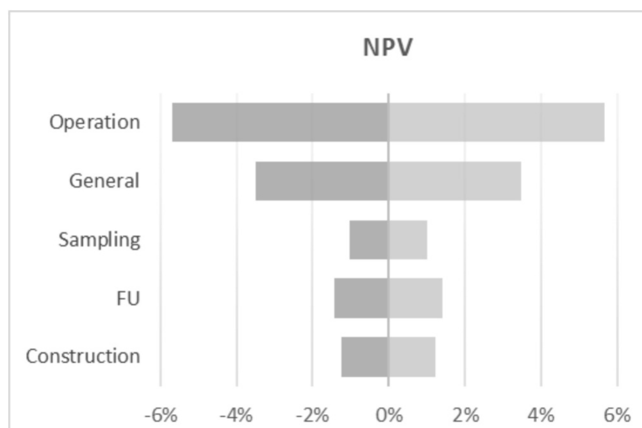

Fig. 6. Tornado diagram representing the sensibility of the Net Present Value (NPV) to the 10 % variation of some parameters to identify the most impactful parameters of the process. Note: FU stands for the variation of the Functional Unit (amount of groundwater treated).

study, which is 70 years.

In terms of private costs, in this case the technology is considered to be implemented at full scale and not at field scale due to the amount of water that needs to be treated to impact the entire population (18,000 inhabitants). For this, the costs are estimated at an full scale level and the data is provided by TAUW, as the company has experience in the estimation of ISMP implementation costs. This data is provided in Table 10 considering the costs provided by the company and the costs obtained after the discounted cash flow analysis considering a 4 % discount rate. External costs such as CO_2 emissions during the implementation of the project, noise, odour and air pollutants are not consider in this case due to the lack of data for the implementation of the upscaled technology.

Applying a 4 % discount rate, the NPV is in −415,185,140 €, meaning that the bioremediation approach is socio-economically not profitable/viable. As can be observed in Table 10, even though the revenues are higher than the costs when the values are not discounted, they are taken into account at the end of the project and are therefore strongly influenced by the discount rate due to the long life.

The sensitivity analysis in this case focuses on the variation of the discount rate. As shown in Fig. 7, the NPV is very sensitive to a small variation in the discount rate. For instance, a positive NPV can be obtained by considering a 2 % discount rate. This topic is further discussed in Section 4.3.

The inclusion of other private and external benefits can make the NPV positive in the 4 % discount rate scenario. In this case, only the monetary value of the human health impacts caused by arsenic is

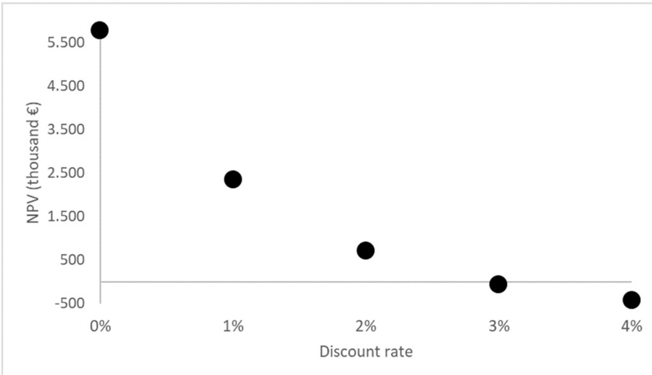

Fig. 7. Representation of the Net Present Value (NPV) showing that slight variations in the social discount rate can significantly affect the socio-economic viability of the In-Situ Metal Precipitation process at full scale following the Impact Pathway Approach.

considered, focusing on skin cancer, but this analysis can be extended to include other human health impacts caused by arsenic in human health that can be avoided, and also extended to the evaluation of the other heavy metals detected in the contaminated groundwater. However, the calculation of the benefits obtained from arsenic removal is sufficient to illustrate the steps that have to be taken. Additionally, other private benefits could be included in the assessment, such as increased land value, avoidance of fines for exceeding legal pollution limits and improve liability for the client.

4. Discussion

4.1. Socio-economic assessment of innovative bioremediation techniques

A challenge in this study was to identify potential benefits to evaluate the viability of technologies tested at field scale. The analysis of the profitability of an innovative bioremediation project during its design phase is complex, as there are no specific guidelines for estimating its potential benefits. This study proposes a methodology to address this problem, evaluating the economic implementation of an innovation such as the ISMP, and its social profitability by monetizing the pollutants eliminated and including the environmental impacts in the financial analysis. During the GREENER project (Grant No. 826312), this technology was technically verified at field scale, which is reflected in the baseline scenario of this work (Pérez-de-Mora et al., 2024). However, as shown in Section 3.2.1, the results at this scale are insufficient to assess the social cost-effectiveness of the technology and further steps should be taken.

Sensitivity and scenario analysis allow different stakeholders involved in the remediation project, such as contractors and researchers, to determine strategies for the future scaling of the technology and to gain insight into the point at which the project would be socio-economically profitable and therefore, beneficial for the local community and governments that are directly affected by the pollution and indirectly paying for the impacts caused. This analysis should include not only social or external benefits, but also private returns that are crucial for investors or land owners, such as the increased value of contaminated land or potential savings on administrative penalties.

In this case study, although the NPV is negative for the ISMP at field scale in the baseline scenario due to the high costs and low benefits, the scenario analysis determine what conditions must be met for the project to be socio-economically viable. The social benefits considered are directly linked to the quantity of water treated, and therefore, increasing the water treated per year makes the NPV positive if no other variable is affected. However, it is not realistic to increase the amount of water treated without increasing at least the construction costs (to install more injection wells) or the operation costs (increase the lifetime of the project). Therefore, we can conclude from Tables 8 and 9 that the most realistic strategy to reach a positive NPV for implementing the project at field scale is to treat 90 m^3 of water per year, assuming an increase of 20 % in operation costs and 60 % in construction costs. However, these considerations need to be further supported by the technical capacity of the process. In addition, this case study includes the external costs related to the implementation of the ISMP process and compares the monetary valuation of an updated LCA methodology recommended by the European Commission (EF 3.0) with a more mature and complete, but less updated, methodology where monetary values of all impacts are available (ReCiPe). The choice to include the ReCiPe monetized results in this case is due to the fact that the inclusion of all impact categories of the LCA gives consistency to the study. In the future, once EP are provided for all categories in the EF, the data can be updated and both methods compared to provide different insights and conclusions.

The aim of this work is to provide guidance to different stakeholders involved in evaluating the viability of emerging bioremediation projects on how to account for external revenues from processes at low Technology Readiness Levels (TRLs). The use of EP to determine the social costs avoided provides a quick calculation of the revenues to be used in social CBA and allows to estimate the NPV, which should be complemented with sensitivity and scenario analysis. The results obtained in this paper are not compared to other studies due to the lack of literature in the environmental and socio-economic evaluation of an in-situ bioremediation technique to treat metal(loid)s in groundwater. To the best of our knowledge, this is the first study where this technique is assessed and the approach used can be applied to other similar projects with the aim to work on a common methodology that allow the comparison of the results between different bioremediation techniques.

4.2. Approaches for social CBA

In this paper, it has been demonstrated that the monetary valuation of environmental impacts can be performed using either a bottom-up (inventory to indicators) or a top-down (damage to indicators) strategy. The two approaches used in this study have different advantages and limitations.

On the one hand, the application of the top-down approach, particularly through monetization using methodologies such as EP, provides an overarching view of the external costs and benefits at the European level. This can be particularly useful for researchers and engineers in assessing innovative projects with low TRL, which require time and effort to determine the monetary values needed for the social CBA. However, it is important to note that the values obtained by this method are calculated using the average European population, which limits its direct applicability to specific contexts or regions (de Bruyn et al., 2023).

On the other hand, the bottom-up approach following the IPA uses concentration-response functions that require more rigorous data collection process to ensure accuracy, particularly when focusing on specific regions or areas, such as the risk associated with the exposure to a particular pollutant, and the costs associated with the health problems that should be adapted to the specific population. However, despite the additional effort required, the bottom-up approach offers a distinct advantage in its ability to focus on the impacts of a specific pollutant on a directly affected population (Krewitt et al., 1998). Conversely, this method requires the assessment of the up-scaled projects. The use of indicators that influence broader populations necessitates a broader data set, which may surpass the scope of what can be obtained at the field scale.

There are some studies in the literature that use these approaches in different ways. Since there is no standardisation of the steps for conducting a CBA, the outcomes of the studies vary considerably, making it challenging to draw meaningful comparisons. For instance, the monetization of LCA is used in a CBA performed by Foglia et al. (2021), comparing the environmental and social benefits of innovative technologies for wastewater treatment at different TRLs, including externalities by monetizing some indicators such as new employments created based on the market wage, the CO_2 emission taken the European Trading System carbon market price and the costs of mineral depletion based on the social value of avoiding extraction of minerals. Some studies also include monetized externalities focusing on one impact factor. For instance, Harclerode et al. (2016) and Van Passel et al. (2013) calculate and analyse the monetary values of externalities focusing on the carbon footprint.

A different perspective is taken in Mumbi and Watanabe (2022), where the authors perform a CBA for textile wastewater treatment considering the private costs of implementing the technology, the costs on human health based on information from hospitals, and the benefits calculated by the "willingness to pay" of citizens asked in a survey, following a contingent valuation method, but not including LCA results. A similar study is done by Wan et al. (2016), as they calculate the CBA for the phytoremediation of heavy metals in soil in a two-year project where only internal costs are considered. In this case the benefits are estimated by accounting the revenues obtained from commercialising the crops used during the phytoremediation, and the valuation of the ecosystem services function recovered after the remediation. This is calculated by the estimation of avoided losses caused by the contaminants in soil, the improvement in quality of the products and in human health for the nearby population.

The top-down and bottom-up approaches are illustrated together in Huysegoms et al. (2018), where the authors compare monetized LCA results using different monetary valuation methods, with a social CBA following the IPA in a full scale remediation project. The study provides information on the differences and similarities of each method for the same case study. It is concluded that both methods can be used to complement each other, but they differ in scope, data requirements and time needed to perform the assessment. Therefore, social CBA following the IPA can be reinforced when introducing monetized LCA, as the former includes the monetary values for all (or almost all) impact categories and the latter is usually focus on some specific impacts such us the leukaemia caused by benzene in soil or the skin cancer induced by arsenic in water.

The present study offers a different perspective to the previous works mentioned. It aims to demonstrate that social CBA can be conducted in two distinct ways, each with its own pros and cons. The eLCC approach is designed to assess the future profitability of an emerging technology prospectively, requiring a field scale assessment. The IPA approach is geared towards evaluating the long-term social benefits of a scaling-up scenario in a particular region. Additionally, it considers that both methods are complementary and that the comparison between them is not possible when assessing different scales of a technology.

The complementarity of the two approaches lies in the fact that the bottom-up approach is needed to calculate the top-down values. For instance, IPA is followed to calculate the EP coefficients that are applied to the LCA impact categories and contaminants, so a comparison between the two methods should carefully consider the same elements to be taken into account for the calculation at the European level in the case of EP, and at the regional level in the case of IPA (de Bruyn et al., 2023). Additionally, the bottom-up approach can complement top-down strategies as it considers the preferences of local stakeholder, and therefore, it can be used to adjust general policies or indicators to a regional level (Carolus et al., 2018).

Another crucial aspect to be considered in both cases is the uncertainty. Whereas EP already includes upper and lower bounds that should be used for sensitivity analysis, IPA may be subject to higher uncertainties depending on the source of information used and the assumptions made, which may be different for each stakeholder (De Zeeuw et al., 2008; Steen, 2019). These uncertainties could have a significant impact on the profitability of the projects and can be treated for decision making by, for instance, performing Monte Carlo simulations, and include the results in Multi Criteria Decision Analysis (MCDA) tools in order to facilitate the decision making considering the balance between environmental, economic and social sustainability (Rosén et al., 2015; Söderqvist et al., 2015; Volchko et al., 2017).

Finally, it should be noted that LCA and social CBA are powerful tools that are increasingly being used, but they do not have be considered for final decisions (Hoogmartens et al., 2014; Nyborg, 2012). These methods should be complemented by additional assessments, such as MCDA, especially when assessing the environmental, economic and social sustainability of emerging technologies (Van Schoubroeck et al., 2021). It is therefore important to remark that the methodologies proposed here are complementary to the decision-making process, but cannot relieve the legally responsible party of its obligations, even if the project is not considered socio-economically viable.

4.3. The choice of discount rate

Historically, the selection of the discount rate has been controversial, especially in the evaluation of public investments and long-term or intergenerational projects (Drenning et al., 2023; Söderqvist et al., 2015). Social discount rates are used to calculate the present value of future costs and benefits of projects and policies reflecting society's preference of how future benefits and costs are to be valued against present ones, reflecting the opportunity cost of capital from an intertemporal perspective (European Commission, 2014).

Neglecting intergenerational effects can have far-reaching consequences, potentially creating inequalities and injustices that future generations will inherit. By failing to account for the long-term consequences of current actions, there is a risk of burdening future populations with the adverse outcomes of present decisions, undermining their quality of life and hindering their ability to meet their own needs. This fact is consistent with the principles of sustainable development, which seek to balance economic prosperity, social equity, and environmental conservation, recognising that the pursuit of short-term gains at the expense of long-term sustainability is inherently flawed and ultimately self-defeating (Torrijos Regidor, 2021).

As shown in Section 3.2.2, the use of fixed discount rates between of 0 % and 4 % can have a significant impact on the future CFs and therefore, the viability of the project due to its long time horizon. For this reason, some authors affirm that conventional discounting methods inadequately capture the complexity of intergenerational environmental impacts, in this case monetized, particularly in the context of climate change and sustainability (Philibert, 2006). Fixed discount rates often undervalue future benefits and underestimate the true costs of environmental degradation, failing to adequately account for the long-term implications of present actions. In response, the use of Declining Discount Rates (DDR) can be beneficial for the assessment of projects related to future well-being in social CBAs (Gollier et al., 2008; Nesticò et al., 2023).

The rationale behind DDR lies in its acknowledgment of the diminishing marginal utility of consumption and the ethical imperative of intergenerational equity. By adjusting discount rates on a declining trajectory, DDR methodologies assign different weights to future benefits and costs, emphasizing the preservation of environmental resources and the welfare of future generations. As underscored by Arrow et al. (2013), DDR methodologies can offer a robust solution to the valuation of intergenerational benefits by providing a more flexible and adaptive approach to discounting.

Following the IPA case study in this paper, where investment costs are accounted in the early years and the benefits are considered at the end of the project, DDR could lead to a higher NPV, as the benefits are not penalised by discounting. For instance, the use hyperbolic, gamma distribution or time-variant discount rates has been proposed by various authors to replace the exponential discount rate in CBAs (Almansa and Martínez-Paz, 2011). Moreover, a zero discount rate could be applied, which can be understood as a way of largely compensating future generations for unmitigated climate damage (Davidson, 2014).

Additionally, the application of DDR can help to overcome the inherent uncertainty of future discount rates, which are challenging to predict.

5. Conclusions

This study combines different methodologies to analyse the socio-economic profitability of a bioremediation technique for the treatment of contaminated groundwater to levels below the regulatory limits for drinking water and, thus, significantly below the regulatory limits for groundwater. The NPV is calculated as the main indicator including external costs and benefits following two different approaches to perform a social CBA of the case study at different scales. The results show that the environmental impacts obtained by EF3.0 and ReCiPe 2016 in the LCA performed for the field scale are not comparable, even though some impact categories are quantified in the same units, and that the project is not socio-economically profitable under either approach in the baseline scenario. In the eLCC approach, mainly due to the high costs during the life cycle of the project in relation to the low quantity of water treated. In the IPA, the NPV is highly influenced by the discount rate, since the costs are incurred at the beginning of the project, and the benefits are assumed to be received in the long term (70 years). In both cases, however, a sensitivity and scenario analysis helped to identify the main hotspots and take actions to make the project socio-economically viable (e.g. reduce the costs of a particular phase of the project or increase the volume of contaminated water treated). These potential solutions should be carefully considered together with the technical modifications and improvements of the technology that are currently being developed.

This study is limited to the low-medium TRL of the ISMP case, which was tested at field scale during 2 years for the treatment of 10 m^3, in order to evaluate the technical functionality of the technology. The innovative social CBA performed in this paper is a comprehensive ex-ante analysis to evaluate the potential socio-economic viability of the investment required for the implementation of a groundwater bioremediation technology with full scale potential application, considering different scenarios. Future research on this topic should focus on finding a common methodology to perform social CBAs for low TRL bioremediation projects to allow the comparison between alternatives. Additionally, deeper analyses of the discount rates considered and deeper uncertainty analyses of the input parameters can be done by running Monte Carlo simulations to observe the probabilities of a positive NPV considering the distributions of the costs and revenues, as well as the comparison of innovative with conventional remediation technologies including private returns, such as the increased value of contaminated land or potential savings on administrative penalties. For instance, the regulatory limit (RW - richtwaarden) for As for property transactions in Flanders of 12 μg/L was achieved by the technology. Contaminant concentrations below the RW do not need to be reported, thus, having a significant impact in the private returns obtained from the transaction process.

The ultimate objective of this work is to provide a methodology to account revenues in (innovative) in-situ remediation field scale projects and thus assess their viability, focusing on the external costs and social benefits, which are necessary for the socio-economic evaluation of any (bio)remediation projects. The methodologies presented can help practitioners and policy makers to determine the pros and cons of using the monetary valuation of LCA (top-down approach) or following the IPA (bottom-up approach) depending on the scale of implementation, available data and if a regionalised information is needed for a specific case study.

Finally, the proposed methodologies may be particularly useful for evaluating the socio-economic impact of different remediation technologies for a particular contaminated site or project, pointing that even legal limits on water pollution have an associated external cost. The methodologies proposed are not intended to relieve the responsibility to remediate a contaminated site when this is legally required. The need to remediate or not is ultimately given by the outcome of the environmental risk assessment in accordance with national/regional guidelines and in consultation with the responsible authorities.

CRediT authorship contribution statement

Jesús Ibáñez: Writing – review & editing, Writing – original draft, Visualization, Software, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Alfredo Pérez-de-Mora:** Writing – review & editing, Validation, Supervision, Project administration, Data curation. **Mario Santiago-Herrera:** Writing – review & editing, Software, Formal analysis. **Benjamine Belloncle:** Writing – review & editing, Software, Formal analysis, Data curation. **Herwig de Wilde:** Project administration, Data curation. **Sonia Martel-Martín:** Writing – review & editing, Supervision, Funding acquisition. **David Blanco-Alcántara:** Writing – review & editing, Supervision. **Rocío Barros:** Writing – review & editing, Supervision, Project administration, Funding acquisition, Conceptualization.

Declaration of competing interest

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

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

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