Journal of Cleaner Production 413 (2023) 137352

Contents lists available at ScienceDirect

Journal of Cleaner Production

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

Integration of ecosystem services and life cycle assessment allows improved accounting of sustainability benefits of nature-based solutions for brownfield redevelopment

Khaled Alshehri [a,b,*], Michael Harbottle [a], Devin Sapsford [a], Alistair Beames [1], Peter Cleall [a]

[a] School of Engineering, Cardiff University, Cardiff, CF24 3AA, UK
[b] Department of Civil Engineering, College of Engineering, University of Bisha, Bisha, 61922, P.O. Box 001, Kingdom of Saudi Arabia

ARTICLE INFO

Handling Editor: Cecilia Maria Villas Bôas de Almeida

Keywords:
LCA
Ecosystem services
Brownfield redevelopment
Soil remediation
Sustainability assessment
Environmental decision support
Nature-based solutions

ABSTRACT

Nature-based solutions (NbS) are increasingly recognized as a sustainable alternative to conventional remediation for brownfield redevelopment. One of the key advantages of NbS is that they provide ecosystem services (ES) during and after remediation. However, traditional Life Cycle Assessment (LCA) does not fully account for the advantages of ES. To address this limitation, we propose a holistic sustainability assessment framework that integrates ES valuation and LCA for brownfield redevelopment planning. The framework is designed to support decision-making by providing a comprehensive environmental analysis of the impacts of different remediation scenarios. The proposed framework is applied to the London Olympic Park mega remediation project as a case study. Three considered scenarios are a business-as-usual scenario, a conventional remediation scenario, and an NbS scenario. The primary and secondary impacts of each scenario, the temporal efficacy of remediation, and the impact on ES are evaluated using the framework. The results suggest that the NbS scenario provides the best trade-off between mitigating contamination risks and economic costs within a reasonable timeframe due to the added benefits of ecosystem services. Overall, the proposed framework provides a comprehensive approach that considers the multiple aspects of environmental sustainability.

1. Introduction

Brownfield is any land or premises which has been previously developed and is not in full use currently because of its vacant, derelict, or contaminated condition (Alker et al., 2000; Tang and Nathanail, 2012). Soil contamination also has adverse impacts on the soil quality and biodiversity therefore an intervention is needed to decrease the associated health risks and to restore the economic value of a brownfield site (Stolte et al., 2016). Brownfield redevelopment aims to restore land to a state-of-use and is attractive because brownfield sites are often located in urban regions with a high-density population (European Commission. Joint Research Centre, 2017) resulting in a drop in the value of the adjacent real estate (Woo and Lee, 2016). For example, in 2018, there were about 18,000 registered brownfield sites in England with a total area of 26 thousand hectares which could provide space for around 1 million houses if remediated and redeveloped (Campagin to Protect Rural England 2018). Conventional brownfield remediation technologies are typically faster but more costly and energy-intensive than nature-based solutions (NbS) (Cundy et al., 2016). The International Union for Conservation of Nature (IUCN) define NbS as “actions to protect, sustainably manage and restore natural or modified ecosystems that address societal challenges effectively and adaptively, simultaneously providing human well-being and biodiversity benefits” (IUCN, International Union for Conservation of Nature, 2020).

Soil functions provide many ecosystem services (ES) such as carbon sequestration, nutrient recycling and food provisioning (Adhikari and Hartemink, 2016; Potschin et al., 2016; Andrea et al., 2018). Similarly, urban soils were found to deliver important ES including flood mitigation, urban heat island reduction, air quality regulations (O’Riordan et al., 2021), and passive CO_2 sequestration (Jorat et al., 2020). Hence, there are increasing calls to consider soil-related ES for responsible soil stewardship in general (Robinson et al., 2012) and relevant decision

* Corresponding author. School of Engineering, Cardiff University, Cardiff, CF24 3AA, UK.
E-mail addresses: AlshehriKM@cardiff.ac.uk (K. Alshehri), harbottlem@cardiff.ac.uk (M. Harbottle), sapsforddj@cardiff.ac.uk (D. Sapsford), alistair.beames@gmail.com (A. Beames), cleall@cardiff.ac.uk (P. Cleall).
[1] Independent researcher

https://doi.org/10.1016/j.jclepro.2023.137352
Received 11 February 2023; Received in revised form 25 April 2023; Accepted 28 April 2023
Available online 13 May 2023

frameworks have been proposed (Lilburne et al., 2020). It was found that soil contamination harms the delivery of ES in urban soils (Rate, 2022), hence it is crucial to consider ES while comparing the sustainability of different brownfield remediation scenarios (Volchko et al., 2013).

Brownfield remediation by NbS is an increasingly popular remedial strategy because not only it mitigates the risks of soil contamination but also provides additional benefits (Vangronsveld et al., 2019) such as bio-energy crop production (Lewandowski et al., 2006), edible crop production on slightly-contaminated soil (Zhang et al., 2021), and recreational activities in redeveloped public parks (Cundy et al., 2013). There are existing sustainability assessment frameworks for remediation activities, such as the UK SURF (Annex 1, 2011), however, ES are not always explicitly accounted for in assessing conventional remediation alternatives and NbS (Onwubuya et al., 2009; Song et al., 2019). The life cycle impacts of NbS remediation systems are still poorly understood (Chandra et al., 2017). Including ES in the appraisal of NbS provides a holistic and objective assessment of NbS's added-value benefits (Hou et al., 2020).

1.1. Literature review: Life cycle assessment & ecosystem services

Life cycle assessment (LCA): LCA systematically analyses the environmental performance of products/services from cradle to grave (i.e., from raw material acquisition to final disposal). The LCA framework consists of 4 phases namely 1) goal and scope definition; 2) life cycle inventory analysis(LCI); 3) life cycle impact assessment (LCIA) and 4) interpretation of results (ISO14044:2006).

Although LCA was employed to assess the sustainability of remedial options as early as 1999 (Diamond et al., 1999), wider adoption did not occur for some time with only two studies published in 2007. By 2018 the number of studies had increased to 12 studies (Visentin et al., 2019). LCA has mostly assessed the primary and secondary impacts of remediation but often overlooked the tertiary impacts (Lesage et al., 2007; Lemming et al., 2010; Hou et al., 2018). Noting that the primary impacts of remediation pertain to the impacts of contamination on human health and ecosystem functions, the secondary impacts stem from efforts to remove the relevant risk (remediation activities) while the tertiary impacts are the effects of reoccupying the treated land (future land uses) (Grifoni et al., 2022). Several remediation LCAs overlooked the tertiary impacts of NbS remediation (Witters et al., 2012) while some studies explicitly excluded the impacts of land use (Puccini et al., 2013; Vigil et al., 2015) and so although recent efforts attempted to account for the aforementioned impacts they are still lacking an encompassing view of NbS impacts (refer to section 1.3).

Ecosystem services (ES): An ecosystem is a dynamic complex of plant, animal, and microorganism communities and the non-living environment. ES are the benefits people obtain from ecosystems (Costanza et al., 1997). These include provisioning services such as food and water; regulating services such as flood and disease control; cultural services such as spiritual, recreational, and cultural benefits; and supporting services, such as nutrient cycling, that maintain the conditions for life on Earth (Costanza et al., 2017; Hassan et al., 2005). NbS remediation systems have been found to provide several ES such as carbon sequestration, improved human health, and recreational opportunities (Hou et al., 2023).

Since many ES are driven by land-use change (Metzger et al., 2006) such as the land-use change due to brownfield redevelopment, accounting of ES in brownfield remediation is attracting increased research in recent years (De Valck et al., 2019; Lin et al., 2021; Zhong et al., 2020a). The attention to ES in urban areas started early on in 1999 (Bolund & Hunhammar, 1999), the 2000s (Tratalos et al., 2007), the 2010s (Berghöfer et al., 2011; Elmqvist et al., 2015), in soil-related ES more recently (Minixhofer and Stangl, 2021), as well as in regional planning for brownfield redevelopment (Zhong et al., 2020b). Often the ES benefits of NbS remediation systems are based on economic assessments (Volchko et al., 2020), overlooking the life cycle impacts of NbS remediation systems (Song et al., 2019).

As discussed above, LCA and ES have been used to assess the sustainability of NbS remediation systems from different angles, a single approach assessment might likely miss some of the impacts/benefits of NbS systems. We argue that an integrated ES-LCA assessment could provide additional insights into the remediation planning and decision making of NbS systems.

1.2. State-of-the-art: ES-LCA integration

Given the similarities of system modelling approaches of both LCA and ES modelling in terms of the cause-chain model of impacts assessment (Alejandre et al., 2019; VanderWilde and Newell, 2021) (refer to Fig B.1 & B.2), there have been several attempts to integrate ES modelling into LCA to develop a holistic sustainability approach (Othoniel et al., 2019; Verones et al., 2017). Early attempts utilised soil organic carbon (SOC) as a proxy to reflect the impacts of land transformation on land productivity (Milà i Canals et al., 2007). Whereas LANCA, used continental-level spatial data to characterise the impacts of land use on soil-related functions such as erosion resistance, mechanical and physicochemical filtration and transformation of pollutants, and groundwater replenishment (Beck et al., 2010); later modifications to LANCA regionalised (national scale) and monetised the CFs to investigate the impacts on ES (Cao et al., 2015; Saad et al., 2011).

Other approaches to integrate ES-LCA made use of several techniques including emergy (Ingwersen, 2011; Park et al., 2016; Rugani and Benetto, 2012), exergy (Finnveden and Östlund, 1997; Y. Zhang et al., 2010), system dynamics modelling (Arbault et al., 2014), and cascade model (Othoniel et al., 2019). Yet the framework proposed by Rugani et al. (2019) arguably remains the most comprehensive attempt to operationalise ES-LCIA integration as demonstrated by the application to rice farming systems (Liu et al., 2020). In Rugani et al.'s approach, the ES-LCA framework consists of four steps to operationalise the proposed integration. The first step is the inventory step which involves the data collection of the FU. The second step is the impacts on ecological processes and comprises two parts; the first pertains to ES change due to human pressures while the second, the LCA aspect, transforms the outputs of ES modelling to midpoint and endpoint impact indicators (i.e., CFs) to characterise the impacts of LCI. The third step (impacts on ecosystem services) investigates the impacts of the second step outputs to affect the ecosystem's capacity to deliver ES (i.e., the impacts of land use change on the state of ES). Lastly, the fourth step, valuation, monetise the changes to ES due to human pressures. It should be noted that the second and third steps could overlap because of the multifunctionality of ES delivery as pointed out by Rugani et al. (2019) and evident in combining the two steps in Liu et al. (2020)'s approach. Though recent efforts attempted to account for the impact of remediation on soil quality using soil organic carbon (SOC) as a proxy of ecosystem quality (H.-P. Chen et al., 2021), such efforts do not completely capture the impacts of remediation on ES (Cappuyns, 2011) because several ES are not directly driven by the state of SOC such as abiotic filtration of nutrients and cultural ES (e.g. knowledge creation) (Potschin-Young et al., 2018).

1.3. Objectives and novelty

In this paper, we modify and tailor Rugani et al.'s framework to the soil remediation and brownfield redevelopment context by defining an additional goal and scope step with the aim to capture the tertiary impacts of remediation (post-remediation development) of conventional and NbS remediation. The novelty of this work lies in improved accounting of NbS remediation benefits through integrating ES accounting into a life-cycle based sustainability framework, development of detailed representative urban land use LCI flow utilising high-resolution spatial data, and sensitivity and qualitative uncertainty assessment of

ES-LCA integration. To demonstrate the framework, we make use of an illustrative case study based on the London Olympic Park (LOP) as a proof-of-concept by analysing four ES: carbon storage and sequestration (CSS), water purification (WP), groundwater recharge (GR), and air filtration (AF). Current obstacles and recommendations to operationalise the ES-LCA framework are discussed with the goal of advancing the integration of ES-LCA in the remediation decision-making process.

2. Refined ES-LCA framework

The overall objective of the refined framework is to facilitate the integration of ES and LCA providing for a holistic sustainability assessment approach to brownfield redevelopment. The refined framework expands on the work of Rugani et al. (2019) by introducing a goal and scope definition step (step I) to the four existing steps of Rugani et al.'s framework.

Fig. 1 illustrates the structure of the refined framework and the necessary procedures for each step which are described in more detail in

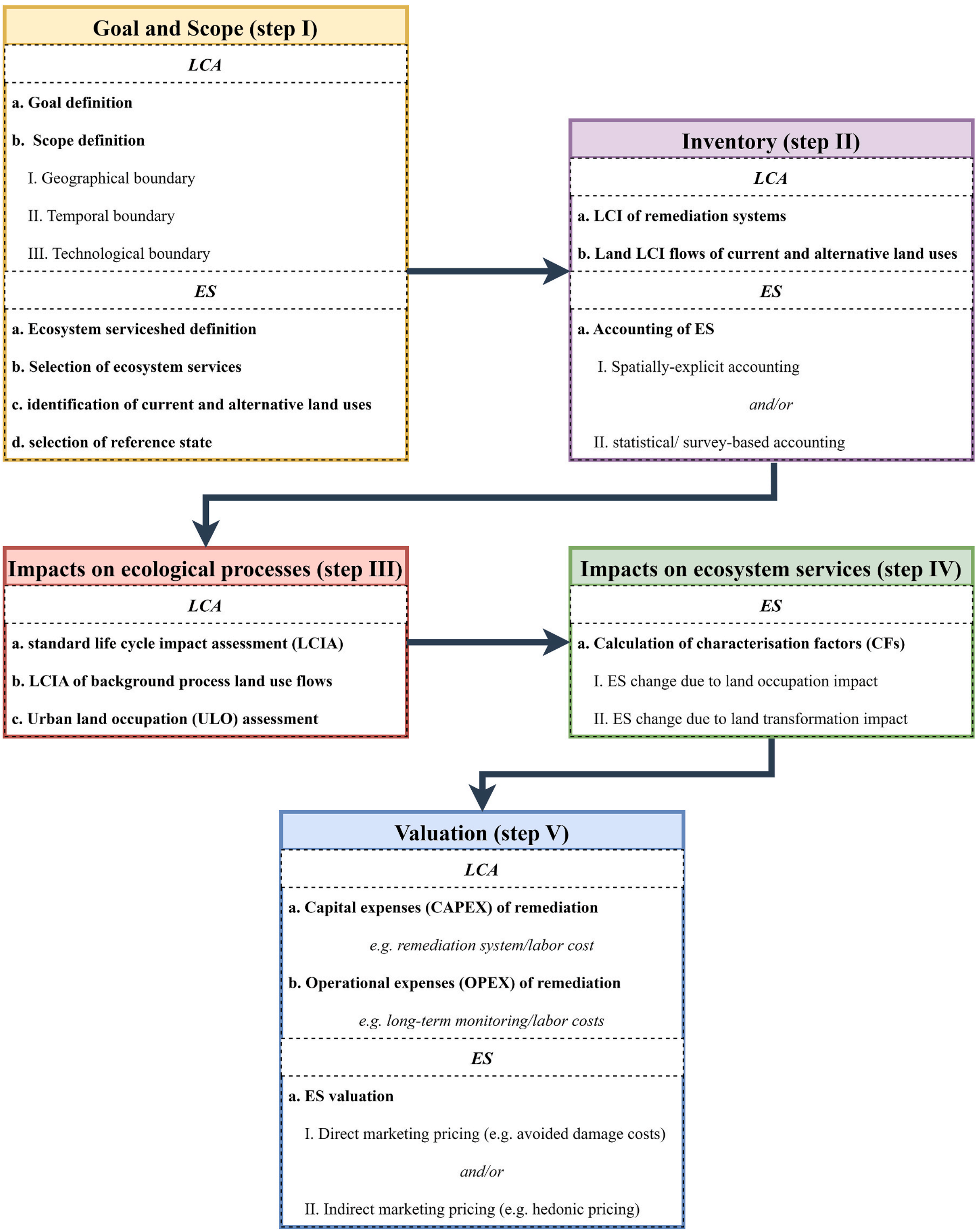

Fig. 1. Refined ES-LCA for the soil remediation context.

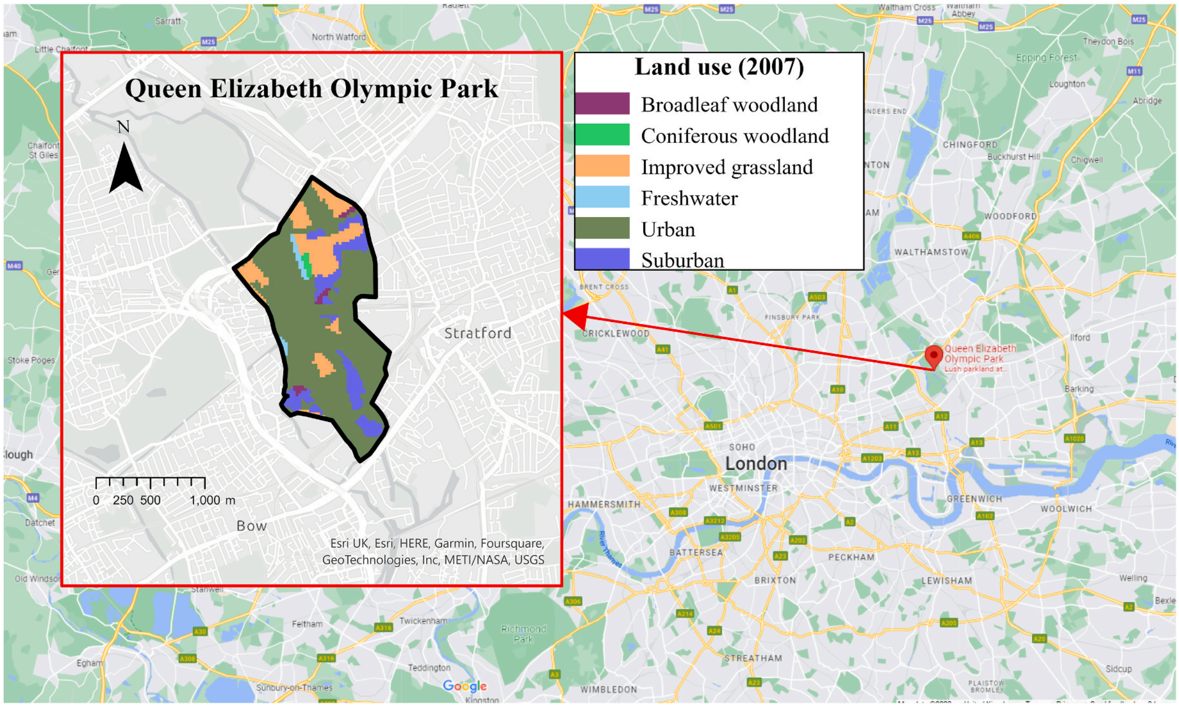

Fig. 2. Case study site in the Greater London area (Land use credit: Morton et al., 2014).

the following sections. The goal and scope, inventory, and valuation steps (steps I, II, & V) integrate LCA and ES. The impacts steps (steps III & IV) accommodate the inherent structures of LCA and ES separately and account for potential feedback loops. The refined framework presents a practical workflow to assess the sustainability of remedial alternatives against a set of metrics (see Fig. 2).

2.1. Goal and scope (step I)

The goal and scope step involves defining the basis for comparison (or baseline), model boundaries, necessary assumptions, and data collection procedures.

2.1.1. LCA

1. **Goal:** In this step, the goal of the study is stated. In the soil remediation or brownfield redevelopment context, the goal would typically be to remediate and recover the functionality of a parcel of land. The choice of an appropriate functional unit (FU) would depend on the goal of the study (European Commission, 2010a) although typically for remediation activities the FU would be to remediate 1 m^3 of soil as per applicable standards (Hou et al., 2014). If the scope of the study goes beyond the remediation activity and includes site redevelopment (i.e., tertiary impacts), then the proper FU would be surface area-based (Allacker et al., 2014). In such cases, the FU would be the redevelopment of 1 m^2 of contaminated land. Both, remediation alternatives (e.g., soil washing) and redevelopment scenarios (i.e., planned future land uses) would also need to be defined.
2. **Scope:** The scope of the analysis is governed by the study objective and specifically what remediation impacts would be evaluated (primary, secondary, and/or tertiary impacts) (Amponsah et al., 2018). For a remediation system, the components of the system boundary should be clearly defined to ensure the consistency of the analysis as follows (adapted from Favara et al., 2011):
   a. ***Geographical boundary*** defines the spatial scale of impacts to be assessed. Examples of boundary categories include: on-site (e.g., energy use due to earthworks), local (e.g., health impacts due to particulate matter formation), regional (e.g., regional water scarcity), and global (e.g., greenhouse gas emissions)
   b. ***Temporal boundary*** denotes the time frame of the analysis. For instance, the time period for the site redevelopment or the time horizon for the impact assessment incurred during and after remediation.
   c. ***Technological boundary*** defines the remediation system in use as well as technological maturity level (i.e., range/average of technology, best available technology, and/or emerging technologies).

2.1.2. ES

The geographical and temporal boundaries of LCA describe the ecosystem serviceshed of each remediation scenario. Firstly, the serviceshed is defined as the net land area that delivers a service consumed or enjoyed by a particular end user, be it a town or a tree (Bakshi et al., 2015). Selecting the proper serviceshed scale is crucial to identify the appropriate spatial scale for modelling the selected ES (Liu et al., 2018). Secondly, the relevant ES to the stakeholders will be identified in the remediation planning phase and defined in line with the LCA goal and scope. Thirdly, based on the goal and scope definition, current and future land uses should be identified to further improve the ES modelling. Fourthly, the choice of a proper reference state is crucial because it significantly influences the characterisation factors (Efroymson et al., 2004; Koellner et al., 2013b). For instance, Potential Natural Vegetation (PNV), is the vegetation cover in equilibrium with the climate, unconstrained by anthropogenic activities (Hengl et al., 2018). A PNV-based reference state would be more appropriate for biodiversity assessment while current mix use would be more appropriate for assessing alternative scenarios to the status quo (Koellner et al., 2013b):

a. Historic reference state:
   i. Current land use mix
   ii. Potential Natural Vegetation (PNV)
   iii. A Mix of Regional Biome (MRB) (i.e., a mix of vegetated land cover)

b. Future reference state (desired state):

The current land use mix might be a practical choice to compare remediation alternatives within one site (Cervelli et al., 2016; Holder et al., 2019). However, a common reference state such as PNV or MRB will be required when comparing different locations (e.g., screening a list of contaminated sites to determine the remediation priority).

2.2. Inventory step (Step II)

To enable impact assessment modelling of steps III & IV, collection of the necessary data is required with emphasis on relevant land use data of each remediation scenario.

2.2.1. LCA

Firstly, the LCI of remediation alternatives should be collected from primary sources (e.g., project documents) or secondary sources (e.g., the literature). LCA terminology distinguishes between foreground and background systems. Foreground systems include processes that are under the control of the system designer/decision-maker. On the other hand, background systems consist of processes which are not directly influenced or under the control of the decision maker (Frischknecht, 1998). In the remediation context, the foreground systems would be the remediation technologies and supporting enabling activities such as earthworks and on-site material transportation. The background systems would refer to the off-site systems such as the off-site transportation and energy generation and transportation systems.

Secondly, in order to inventorise the current and alternative (future) land uses, identified in the goal and scope step, they are modelled as elementary land resource flows based on the consensus-led UNEP-SETAC land inventory principles (Koellner et al., 2013a). Depending on the LCA spatial scope, land occupation and transformation flows may need to also be inventoried. The regionalised land use flows should be numbered according to the level of available details as recommended by the UNEP-SETAC guidance (Koellner et al., 2013a) (see Table B.1). If a remediation alternative is area and time-intensive, (e.g., phytoremediation), then a level-3 or level-4 land occupation flow and its associated ES should be modelled.

2.2.2. ES

1. Spatially-explicit accounting of ES:

Firstly, current and alternative land cover maps (reflecting the reference scenario and alternative remediation scenarios) should be obtained, each scaled to the spatial resolution of the finest available spatial input, to retain all available information. For example, if there is a land cover map of 25m grid size and a soil group map of 100m grid size, then the soil group map's grid size should be transformed to 25m grid size using a GIS platform (Mutel et al., 2019). Generally, a spatial resolution of less than 200m is recommended as coarser resolutions might negatively affect the accuracy of ES modelling (Redhead et al., 2018) while finer resolutions will enable ES accounting for smaller areas (Larondelle and Lauf, 2016). The spatial data representing the reference and alternative scenarios are then fed into a spatially-explicit model for ES accounting (e.g., InVEST models, refer to section 3.4) to spatially quantify the ES at the chosen spatial scale (e.g., site, city, regional, or national level). The results should be validated with on-site data where feasible. The result of ES accounting is then normalised and attributed to the LCI land flows. The choice of normalisation approach, mean or median, depends on the spatial distribution of ES and relative distance to project boundary.

2. Statistical/survey-based accounting of ES:

We recommend the use of spatially-explicit ES accounting methods where possible to ensure that all selected ES are accounted for at the same spatial scale. In case of the absence of spatially-explicit ES accounting models for a particular ecosystem service or lack of relevant spatial data, alternative ES accounting methods such as TESSA (Peh et al., 2013) or similar survey-based approaches can be used. Care should be taken to ensure the survey-based accounting results represent the chosen spatial scale. In the case of cultural ES, although some cultural ES might be transboundary such as educational opportunities (Hutcheson et al., 2018), they are mostly generated onsite (Schirpke et al., 2021). Since it is sometimes challenging to model cultural ES at a larger scale (Brown and Fagerholm, 2015), we recommend limiting the accounting of cultural ES to within the project boundary. Survey-based accounting is particularly useful for provisioning and cultural ES which are often provided on-site. However survey-based accounting is impractical for measuring ES at the serviceshed level due to the time and effort required (Lankia et al., 2020).

2.3. Impact step (Step III)

In this step, an amended life cycle impact assessment of remediation alternatives is adopted to characterise the potential life cycle impacts of foreground and background processes (i.e., associated elementary flows) of the remediation activities.

1. Standard LCIA modelling:

Standard life cycle impact assessment of background systems can be modelled based on generic CFs from available impact assessment methods such as ReCiPe2016 (Huijbregts et al., 2017) or TRACI (Bare, 2011). Since land remediation is a localised problem (H.-P. Chen et al., 2021), preference should be given to regionalised impact methods such as AWARE (Boulay et al., 2018) and IMPACT World+ (Bulle et al., 2019) where feasible.

2. LCIA modelling of land use of background processes:

It should be noted that the land use impact characterisation of background processes could be achieved using existing land use impact methods. However, the land flows of foreground systems are a proxy for ecological modelling (i.e., ES accounting) therefore they must be excluded in this step to avoid double counting.

3. Land competition impact modelling measured in urban land occupation:

The application of the Urban Land Occupation (ULO) proposed by Beames et al. is adopted to account for the temporal efficiency of urban land use by each remediation alternative (2015). ULO will be later (see section 2.5.1) monetised to reflect the cost of the land being out of use due to remediation.

$$ULO = LR \times C_{ULO} \tag{1}$$

$$LR = A \times t \times \frac{1}{R} \tag{2}$$

where LR is the land resource and C_{ULO} is the monetary cost per unit area. A is the area (in m^2), t is the time of land occupation in days/years), and R is the total land area subtracting the protected areas within the specified boundary (e.g., city limits, regional, or national) over the population of the area.

2.4. Impacts on ecosystem services (step IV)

In this step, land flows of foreground systems (i.e., remediation alternatives and on-site enabling supporting activities) are employed as a proxy to investigate the potential impacts of remediation alternatives on the local ecosystem's capacity to deliver services to beneficiaries.

1. Calculation of characterisation factors (CFs):

In this step, the output of ES accounting is used to analyse the impacts of foreground systems on ecosystem quality due to human pressures. The characterisation factor (CF) of land use is defined as

$$\mathrm{CF} = Q_{ref} - Q_{final} \tag{3}$$

where Q is the ecosystem quality at any given time (i.e., ecosystem capacity to deliver ES to beneficiaries) and the subscript refers to the scenario represented. Thus, Q_{ref} is the ecosystem quality of the reference scenario while Q_{final} is ecosystem quality of the alternative scenario (i.e., Q for a comparative/alternative scenario). Q depends on the output of spatially-explicit and/or survey-based accounting of ecosystem services with respect to time (as depicted in Fig B.3). The land use impact of remediation is then assessed with Equations (4)–(6) which are adapted from the UNEP-SETAC guidance on land use life cycle impact assessment (Koellner et al., 2013a). The land use life cycle impacts of remediation include land occupation during remediation (Beames et al., 2015) and land transformation due to improved soil conditions (H.-P. Chen et al., 2021).

a. Land occupation impact:

Here the impacts of land occupation, I_{occ}, are characterised in terms of ecosystem service unit per unit area multiplied by time period occupation as follows:

$$I_{occ} = CF_{occ} \times A \times t_{occ} \tag{4}$$

where A is the land area and t_{occ} is the duration of land occupation.

b. Land transformation impact:

The impact of land transformation, I_{trans}, is defined in the following manner:

$$I_{trans} = CF_{trans} \times A \times t_{reg} \tag{5}$$

Where t_{reg} is the time required for a parcel of land to regenerate the ecosystem quality to a natural state. In the remediation context, t_{reg} could be substituted with the time required for on-site natural attenuation to fully mitigate the pollutant to the applicable limits (t_{na}). t_{na} could be obtained with numerical modelling based on on-site sampling which takes place during the remediation planning and design phase. However, if obtaining a reasonable t_{na} proves infeasible, then it could be assumed that the transformation is instantaneous as recommended by the UNEP-SETAC guidance (Koellner et al., 2013b). Thus, $t_{reg} = 1$, hence:

$$I_{trans} = \left(Q_{from} - Q_{to}\right) \times A \tag{6}$$

2.5. Valuation (step V)

The valuation step aims to provide stakeholders with an aggregated score to facilitate the decision-making process (Cao et al., 2015). Here the life cycle costs of remediation are quantified in addition to the "land rent" value as represented by monetising ULO score. Further, the ES-related impacts are also evaluated based on the available ES valuations if any.

2.5.1. LCA

In this step, an environmental life cycle costing (eLCC) is adopted as per the ISO14044 standard and ILCD guidance (European Commission, 2010b; Hauschild et al., 2018; ISO 14008:2019 Monetary Valuation of Environmental Impacts and Related Environmental Aspects, n.d.). An eLCC is analogous to LCA as they share the four phases namely goal and scope; LCI; LCIA; and interpretation. In fact, many of the popular LCA software packages such as SimaPro and openLCA enable the LCA modeller to pre-define the cost of foreground processes in addition to LCA modelling. An eLCC in a remediation context would consists of the following costs items:

1. Capital and operational expenditures (CAPEX & OPEX):

The cost of earthworks and remediation systems including the cost of materials and labour cost is an example of CAPEX while the cost of long-term operations of remedial actions is an example of OPEX.

2. Urban land value:

Here we monetise the ULO results from step III to reflect the cost of the land being out of use during remediation; this will also guide the planning of future land uses which is a crucial part of brownfield remediation investment decision-making (I.-C. Chen et al., 2019).

3. The internalisation of environmental externalities:

Carbon tax or potential credit (e.g., a tax break for 100% renewable energy) are accounted for as a part of the eLCC. This step shouldn't be confused with the valuation of ES; as the effort here is made to monetise the environmental externality prior to or during the remediation (primary and secondary impacts of remediation). While ES valuation focuses on the tertiary impacts of remediation as discussed later. Finally, no discounting takes place in eLCC as it assumes a steady-state similar to LCA (Hunkeler et al., 2008).

2.5.2. ES

This last step aims to put a monetary value on the change of ecosystem quality (output of Step III). Several ES valuation methods are available in the literature (Greenhalgh et al., 2017; ISO 14008:2019 Monetary Valuation of Environmental Impacts and Related Environmental Aspects, n.d.; Potschin et al., 2016). ES valuation could be obtained from academic literature, public policy documents, and survey-based (e.g., willingness to pay); the Ecosystem Services Valuation Database (ESVD) is a useful resource which catalogues ES valuation studies from around the globe (*ESVD*, n.d.).

2.6. Truncation error estimate (TEE)

Truncation error in the LCA context is defined by Ward et al. as "the proportion of impact (investigated value) not covered by the system boundaries of the LCA" (2018). The tertiary impacts of remediation would be the overlooked impacts of traditional LCA as discussed above (see section 1.2). To investigate the effectiveness of ES-LCA, we consider the truncation error via:

$$TEE = 1 - \frac{MI}{EI} \tag{7}$$

where TEE is the truncation error estimate, MI is the measured impact and EI is the estimated total associated impact. In the remediation context, MI is the primary and secondary impacts of remediation represented by the traditional LCIA, and EI is MI in addition to the tertiary impacts of remediation represented by ES-LCA.

2.7. Sensitivity and uncertainty assessment

Uncertainty in LCA has been a long-standing concern and is rarely assessed in the LCA practice (Lloyd and Ries, 2008; Lo Piano and Benini, 2022; Ross et al., 2002). Sources of uncertainty in LCA include parameter uncertainty; model uncertainty; scenario uncertainty and relevance uncertainty (Hauschild et al., 2018). Several approaches emerged to deal with uncertainty in LCA (Igos et al., 2019); such as sampling methods (e.g., Ciroth et al., 2004), analytical methods (e.g., Heijungs, 1996), and fuzzy logic methods (e.g. Tan, 2008). However, the Pedigree Matrix proposed by Weidema & Wesnæs (1996), inspired by Funtowicz & Ravetz's work (1990), has become widely accepted as the established method to assess uncertainty in LCA and has been successfully applied to the main-stream LCI databases (Ciroth et al., 2016; Weidema et al., 2013, p. 3). The pedigree matrix assesses the data by means of selecting a data quality indicator (DQI) factor to represent the data quality (uncertainty) across five indicators; reliability; completeness; temporal correlation; geographical correlation; and technological correlation (see Table B.12). The aggregate DQI ranges from 1 to 5, a lower DQI indicates a lower uncertainty. Though the choice of an acceptable DQI score is not a straightforward exercise but a DQI score which is equal to or less than 2 was deemed to be "fair" and "acceptable" (Junnila and Horvath, 2003; Leroy and Froelich, 2010; Maurice et al., 2000).

Regionalised LCIA is also subject to concerns about the representation of uncertainty, although, there is no consensus or guidance for uncertainty assessment of regionalised LCIA as yet (Mutel et al., 2019). The application of the pedigree matrix has been also adapted to assess the parameter uncertainty in regionalised CFs (Alves et al., 2020). Therefore, we make use of the pedigree matrix to qualitatively assess the parameter uncertainty of 4 input data namely the LCI of the remediation, spatial data inputs of ES accounting, capital and operational costs of remediation, and monetary damage costs to ES. We also apply a modified pedigree matrix proposed by Qin et al. to qualitatively assess the uncertainty of CFs model (Qin et al., 2020) (see Table B.13). We consider a DQI of less than 3 (corresponding to moderate uncertainty) to be acceptable given the hypothetical nature of the worked case study.

An adapted version of the pedigree matrix designed for CFs, which was proposed by Qin et al. (2020), was applied to the calculated CFs. The indicators of the adapted pedigree matrix differ from the original as they were designed to assess the uncertainty of CF specifically (see Table B.13) or the output of the refined ES-LCA integration. Notwithstanding some descriptions of DQIs are not directly applicable to ES-LCA integration. In particular, the model completeness which describes the CF's coverage to the elementary LCI flows in the LCA model, e.g. "The results of the model have a moderate coverage of the characterization factors for all elementary flows in an LCI (over 60%)" such language is not suitable to our model which proposes both new elementary land use LCI flows and CFs. Despite the discussed slight deficiency, it does not negate completely the usefulness of the other indicators such as the temporal specification of CFs. Although we recognise the importance of a formal quantitative uncertainty assessment to account for the possible equifinality among the different factors that affect the framework, such an exercise is broader than the scope of this work.

3. Illustrative case study

3.1. Case description

To demonstrate the application of the refined assessment framework we employ an illustrative case study based on The London Olympic Park (LOP), also known as Queen Elizabeth Olympic Park. The site was home to a wide array of industries throughout the 19th and early 20th centuries in addition to several waste filling sites (predating engineered landfills) resulting in the presence of several contaminants of concern such as heavy metals, polyaromatic hydrocarbons, chlorinated solvents as well as ammonia. The redevelopment works took place across a site of over 200 ha between 2007 and 2010, with over 2 million m^3 of earthwork including around 700 thousand m^3 of treated contaminated soil. Soil remediation was achieved using four remediation techniques and material sorting (refer to Table B.2), 80% of the remediated soil was reused onsite (Hellings et al., 2011; Mead et al., 2013).

3.2. Modelled scenarios

Three scenarios are considered to compare and contrast the functionality of the refined ES-LCA framework. The first scenario is a business-as-usual scenario which assumes that the in-situ natural processes will degrade the pollutants within a reasonable timeframe hence no intervention is undertaken. The second scenario assumes a conventional remediation in undertaken to rehabilitate and redevelop the site within a specified time duration. The third scenario represents an NbS remediation scenario in which the site will not be in-use during the remediation period to minimise the exposure risk to pollutants.

3.2.1. No action (NA) scenario

Remediation of subsurface contamination is assumed to be achieved via monitored natural attenuation (MNA) over 30 years (Carey, 2000; Declercq et al., 2012). A total of 27 monitoring wells are installed within the project, the monitoring system is assumed to be a Continuous Multichannel Tubing(CMT) multilevel system and 21m deep (CL:AIRE, 2002). The monitoring wells are assumed to be sampled quarterly (Adamson and Newell, 2014) and sent to a 30 km away testing laboratory. The LCI of the NA scenario is presented in Table B.3.

3.2.2. London Olympic Park (LOP) scenario

This scenario represents the real timeline of the soil remediation works prior to the construction of Queen Elizabeth Olympic Park (Mead et al., 2013). Several remediation technologies were used (see Table B.2) including "Soil hospitals" which consist of multiple soil washing plants that were pioneered due to timing constraints of the London 2012 Olympic games (Hellings et al., 2011). The LCI of the LOP scenario is presented in Table B.4.

3.2.3. Nature-based solution (NbS) scenario

A large-scale hypothetical phytoremediation is assumed by planting hybrid poplar which is capable of remediating hydrocarbons and heavy metals (Gordon et al., 1998a, 1998b; Salam et al., 2020; Vangronsveld et al., 2019). A plantation density of 5000 trees per hectare is assumed over 12 years with a 30% die-out due to contamination (Dickinson, 2000). An average depth-at-breast height (DBH) of 14.7 cm and a total height of 13 m are assumed based on national biomass yield tables (Christie, 1994). Post-remediation, the phyto-biomass will be treated by anaerobic digestion (Vigil et al., 2015). The LCI of the NbS scenario is presented in Table B.5.

3.3. LCA modelling

LCA modelling has been performed using the openLCA software version 1.10.3. The LCI of foreground systems were obtained from the project document and literature (refer to SI). Ecoinvent LCI database version 3.7.1 was used with the Allocation at the point of substitution (APOS) system model for the background process (Wernet et al., 2016). The ReCiPe2016 midpoint life cycle impact assessment method (Hierarchist) was selected for impact assessment (Huijbregts et al., 2017). The functional unit (FU) is defined as the "treatment of 1 m^3 of polluted soil as per applicable standards". The project boundary is the geographical scope while the period necessary to achieve the applicable remediation standards is the temporal scope (noting the remediation period varies for each remediation alternative. The technological scope of modelled scenarios is described above in section 3.2.

3.4. ES modelling

Spatial accounting of ES within the project boundary was performed by the Integrated Valuation of Ecosystem Services (InVEST) which encompasses several models to map and value ES (Sharp et al., 2018). InVEST requires the input of spatial datasets (Table B.6) as well as biophysical tables for each model (e.g., carbon pools per land use classification for carbon sequestration modelling). The geographical scope of ES modelling is similar to LCA modelling (i.e. the project boundary) and the reference state is defined as the land use mix of 2007 (Morton et al., 2014, p. 2). The inputs of biophysical tables reflecting local conditions were sourced from the literature (refer to Tables C.1 – C.3). Due to the lack of an air filtration model in InVEST (Sharp et al., 2018), we made use of the ecosystem accounts approach developed by the UK's Centre for Ecology and Hydrology (CEH) to estimate pollution removals by different land uses from 2007 to 2030 (Jones et al., 2017) (refer to Table C.4).

4. Results & discussion

4.1. Traditional life cycle impact assessment (LCIA) results

The results of the traditional LCIA are presented in Fig B.5, and the extensive list of ReCiPe2016 midpoint indicators is available in Table C.6. The LOP scenario is seen to have the largest impact across the selected categories compared to the NA and NbS scenarios. This result is expected since the LOP scenario mostly uses resource-intensive remediation technologies namely soil washing (SW) and geotechnical stabilisation (GS) which treated 57% and 29% of the total remediation volumes respectively. The NbS scenario has a low impact relative to the LOP scenario with the harvesting phase contributing significantly to the LCIA results of NbS. The NA scenarios' results are almost negligible compared to NbS and LOP, the upstream LCIA caused by the installation of the monitoring well is contributing 59%–95% and dominating the selected LCIA indicators. Care should be taken while interpreting the LCIA results because the purpose of this hypothesised case study is to demonstrate the feasibility of the refined ES-LCA integration rather than comparing the LCIA of each scenario.

NA Scenario: Though the LCA literature on MNA is scarce (Ditor, 2010), the results of NA's LCIA were surprising as the transport of samples (TS) to testing was not significant but rather manufacturing of monitoring wells (MMW) which contradicted previous studies (Lemming et al., 2010; Lilien et al., 2014). An exception in the Global Warming Potential (GWP) and Fine Particulate Matter Formation (FMPF) categories was present, as the transport of samples contribute 41% and 33% of the indicator scores respectively arising from the use of fossil fuel (diesel). The MMW activities are causing >90% of the rest of the impact categories. The difference in the LCIA results relative to the literature could be attributed to the relatively higher number of wells (n = 27) as well as the depth of the wells (21 m below surface level).

LOP Scenario: The impacts of the LOP scenario are largely from the energy use and cement production in the SW and GS remediation technologies. A third of the GWP score is attributed to the cement production used in GS while electricity consumption in SW is responsible for 18% of the emissions. A similar trend is observed in the Water Consumption (WC) category; about 50% of the WC score is attributed to GS while another 30% comes from SW including water used in the washing process and water used in the concrete used in the platform construction of the SW plant. The fine particulate matter formation (FPMF) category score is due to the use of diesel consumed in the earthworks and energy generation. Coal burned for energy generation is responsible for 50% of the freshwater eutrophication (FE) score a further 30% comes from cement used in GS; a similar trend was present in the marine eutrophication score. In the land use (LU) impact category of background processes (upstream impacts in the supply chain), around half of the score is attributed to the production of biochar (including required wood chips) that is used in the chemical stabilisation. A caveat here, due to the proprietary right of chemical reagents, which is a common issue of chemical-based remediation (Lilien et al., 2014), the LCI of this technology was assumed based on the publicly available information of the actual case study (refer to Appendix B).

NbS Scenario: The midpoint LCIA results of the NbS scenario are low impact relative to the LOP scenario, particularly GWP and WC categories. The harvesting phase, which consists of machine-powered harvesting and anaerobic digestion of phyto-biomass, was identified as a hotspot affecting all significant impact indicators. For example, AD contributes 60% and 87% to the EF and Marine Eutrophication (ME) scores respectively, while the use of mineral fertilisers makes up a quarter of the ME score. In the GWP category, AD and diesel burned in harvesting generated collectively 60% of the GHG emissions in the NbS scenario. In the FPMF category, the diesel used in harvesting is 30% of the respective fine particulate emissions. Although this is a hypothesised scenario based on the literature, the LCIA results suggest the choice of phyto-biomass treatment method is a significant driver of the potential environmental impacts as well as the selected fuel source of the harvesting process.

4.2. Accounting of ES results

The results of spatially-explicit accounting of ES delivery over the study area (200 ha) are shown in Fig B.6 and Fig B.7. The primary purpose of ES accounting here is to investigate the role of future land uses of the redeveloped brownfield on the ecosystem capacity to deliver ES (i.e., the tertiary impacts of remediation). It can be noted that the ES results (apart from AF) of the reference scenario (land cover of 2007) and the NA scenario are similar because it is assumed that the land cover doesn't change in the latter. The nearest corresponding meteorological data were used in the AF accounting (e.g., metrological data of 2007 in the reference scenario while the metrological data of 2011 for the LOP scenario); The varying air pollutants loading each year explains the difference in AF between the reference and NA scenarios.

The NbS's scenario CSS, GR and $WP_{P\ retention}$ outperform the other scenarios corroborating previous results (Dimitriou and Mola-Yudego, 2017; Zalesny et al., 2019) while $WP_{N\ retention}$ of the NbS scenario is slightly less than the LOP scenario. The improved LOP's $WP_{P\ retention}$ result was observed in the Lee River which crosses the case study boundaries (Georges, 2018) and could be attributed to the larger grass covers which are more effective in P retention relative to tree covers (i.e. NbS) (Reddy et al., 1999). Finally, the NbS's capability to regulate air quality is better than the rest because of the higher leaf surface area (Rogers et al., 2015) but hybrid poplars were also reported to be a source of volatile organic compounds (VOCs) (Koch, 2019).

4.3. ES-LCA integration

In this section, we show the results of steps IV&V pertaining to ES-related impacts due to remediation and monetary value respectively. The ES-related CFs as a function of the spatially-differentiated land use flows are shown in Table C.6. Furthermore, the overall biophysical and monetised results of ES-LCA are presented and discussed.

CF calculations: Table C.6 shows the novel LCI flows and CFs corresponding to each modelled scenario. The calculated CFs are essentially the results of spatially-explicit ES accounting normalised to the case study area. All CFs excluding CF_{AF} are characterised by a physical unit over unit area these CFs describe a mid-point impact pathway in the terms of traditional LCIA. In the case of CF_{AF}, the unit is monetised so all 4 modelled pollutants will be characterised by a single indicator. The negative values of NbS's CFs, as in the LCIA sign convention, indicate a potential avoided damage to ES. All of the CFs above are aligned with the appropriate corresponding midpoint indicators from the ReCiPe2016 (Alejandre et al., 2019) (Refer to section 4.1).

4.3.1. Impacts on ecological process and ecosystem services (steps III & IV)

Table .1 represents the results of ES-LCA integration (as depicted by Fig. 3), the values are CFs multiplied by the area of transformed land in the case of LOP and NbS scenarios at which the transformation was assumed to be instantaneous therefore T_{occ} is equal to 1 whereas the T_{occ} of the NA scenario is the duration of MNA. Overall, the degradation of ES quality is observed in the NA and LOP scenarios excluding the LOP's

Table 1
Characterised ES-LCA results for each remediation scenario.

Scenario	CSS	GW	WP	WP	AF
NA	0	0	0	0	3.47E-02
LOP	4.02E+00	1.93E-02	2.87E-05	3.00E-05	4.48E-03
NbS	−1.26E+01	−5.20E-02	2.87E-05	3.01E-05	−9.71E-02
Unit	kg CO2e	m3	kg N	kg P	£ per annum

$WP_{N\ retention}$ which presented a potential ES benefit. The NbS scenario demonstrates an overall environmental savings trend (excluding $WP_{n\ retention}$) supporting previous results (da S Trentin et al., 2019; Espada et al., 2022; O'Connor et al., 2019).

4.3.2. *Valuation (step V)*

Fig. 4 illustrates the results of the valuation step in the framework which includes the remediation costs, the "rent value" of urban land occupation (see Table B.9 for remediation and ULO costs), and the monetised impacts on ES. The ES-LCA results of NbS indicate environmental savings while the LOP has caused £1 worth of environmental externalities per FU. As expected, the remediation of the LOP is the highest while the NA's are almost half of the NbS. It should be noted that the ES-LCA cost of LOP is less than 0.5% of the remediation costs while it is 16% in the NbS case. Though these results might be significant, they are likely to underestimate the true value of ES considering we only analysed 4 ecosystem services in addition to using conservative estimates of ES valuation. Accounting for the ULO costs presents an interesting finding, the net cost is essentially equal in NA and LOP highlighting the implicit cost of remediation temporal efficacy.

4.4. *Truncation error estimate (TEE)*

This section shows the results of the TEE analysis to demonstrate the percentage of overlooked impacts of traditional LCA results compared to the ES-LCA outcomes. Table .2 shows the ecosystem service mapped to the relevant mid-point LCIA indicator.

4.5. *Sensitivity*

ES damage costs: Direct market valuations of ES (i.e. avoided damage costs) might underestimate the value of ES. They were selected in this analysis to avoid potential bias of hedonic pricing (Bouma and Van Beukering, 2015; Irvine et al., 2020; Kuminoff et al., 2010). Fig. 4 a-e show the sensitivity results of ES damage costs across the lower, central, and higher bounds estimates respectively (also refer to Table B.10). Though AF and $WP_{P\ retention}$ ES' monetisation values are higher than CSS on a per unit basis. CSS is the most sensitive parameter. This is expected due to the larger amount of physical ES accounting as well as higher monetisation values (See Fig B.6 & B.7 for ES accounting results and Table B.10). In the NA scenario, no variation is observed due to the steady-state of ES delivery due to the assumed constant land use. The variation in the LOP and NbS scenarios is primarily driven by the CSS value; noting the negative value in the NbS case indicates a potential avoided damage suggesting increased benefits estimate of NbS with different valuation approaches.

Remediation and ULO costs: The sensitivity of the remediation and ULO costs were tested (Fig. 5). The NA scenario is most sensitive to the cost of ULO (the land being out of use), because of the longer timescale of this scenario (30 years). The higher bound estimate specifically was twice as large as the NbS because of the larger costs of land in the greater London region (*Land Value Estimates for Policy Appraisal*, 2015). The NbS remains the most cost-effective scenario across the valuation bound, with relatively lower capital and operational costs and moderate remediation time scale (12 years). While the remediation costs are the main contributor to the sensitivity of the LOP scenario, the ULO's impact on sensitivity is negligible due to the relatively shorter timescale (3 years). The LOP scenario overall cost variation is interesting as in the lower bound estimates it was closer to the NbS value whereas it is approximately equal to the LOP in the central estimate implying that a cheaper energy-intensive remediation might be more attractive than the low-impact NbS scenario based on purely cost-effectiveness perspective.

4.6. *Uncertainty*

In this section, we look at the results of the pedigree matrix of input data and the CFs of the ES-LCA integration. The data quality indicators of the input data and CFs are presented as a heatmap in Fig. 6 and discussed in sections 4.6.1 and 4.6.2 respectively.

4.6.1. *Input data*

The DQI results of input data are shown in Fig. 6 a-d. The overall DQIs of the remediation system LCI are representing moderate uncertainty. The LCI of the chemical stabilisation (CS) system has a DQI of 3 arising from the absence of publicly available information regarding the used proprietary chemical reagents (see Table B.4). On the contrary, the LCI of soil washing and material sorting both have a DQI of 1 because the LCI were collected from the actual case study (Hou et al., 2014). Despite the high DQI of the CS system, the aggregate DQI of the LOP scenario is 1.9 which reflects a low uncertainty altogether.

The spatial data pedigree matrix exhibits a lower uncertainty generally. The high uncertainty of the reliability of the NbS land cover map is attributed to the fact that NbS is a simulated scenario while the 3 DQI reliability score of the hydrologic soil group map is because of the coarser spatial scale (see Table B.6). The DQIs of the assumed remediation CAPEX and OPEX are of moderate uncertainty.

The DQI's hotspots of the NA scenario are the temporal and geographical of the OPEX and CAPEX respectively. The NA OPEX's temporal DQI was considered highly uncertain because of the longer timescale of this scenario (30 years), the similar result of NA CAPEX's geographical DQI because the assumed monitoring system is of North American origin which could present supply chain-related concerns. The remediation costs of the LOP scenario were assumed to be a lump sum factoring the depreciation costs of the remediation system therefore the

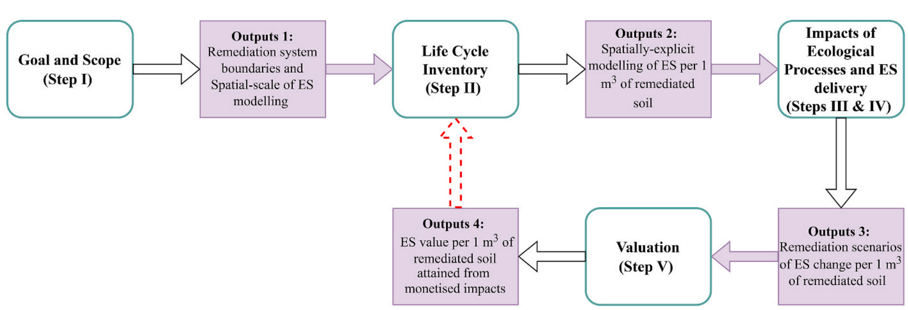

Fig. 3. Adaptation of the ES-LCA framework to the remediation context. Note that impact steps, steps III & IV corresponding to Steps II & III in Fig. 1 of Rugani et al. (Rugani et al., 2019), are combined and feedback loops were not considered (similar to Lui et al. (Liu et al., 2020)).

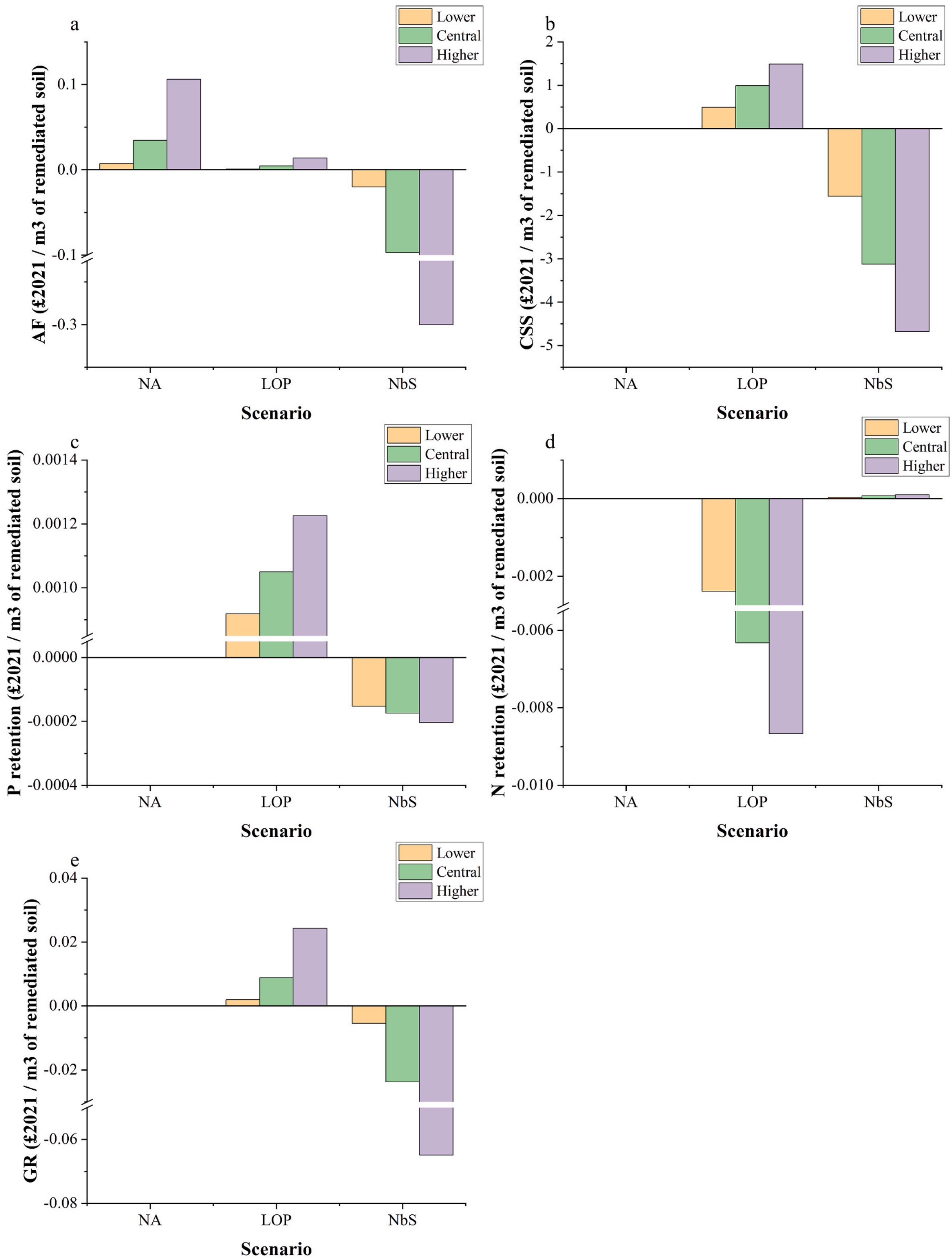

Fig. 4. Valuation (step V) outputs (i.e., monetised impacts to ES in 2021 Sterling Pounds) per FU.

corresponding reliability DQI score was highly uncertain (see Table B.7). Finally, high uncertainty is prevailing in the NbS's CAPEX and OPEX but the reliability DQIs are of moderate uncertainty because they were obtained from an industry association body (see Table B.8). The general trend of moderate uncertainty of the remediation's CAPEX and OPEX is not of concern regarding the robustness of the refined ES-LCA approach because such variation of the costs is expected.

Lastly, the DQIs of the damage costs were of low uncertainty because the estimates were obtained from public authorities in the same region rather than using international estimates (see Table B.10). The high uncertainty of the CSS's temporal correlation was due to the choice of a discounting rate of carbon, as different policy timescale corresponds to a different discounting rate (Great Britain and Treasury, 2022). The geographical DQI scores of the two components of WP (i.e., N and P

Table 2
Truncation error estimate of ES-LCA.

Ecosystem service	TEE			Mid-point LCIA	Notes
	NA	LOP	NbS		
AF	100% *	7%	95%	FPMF	*The significant difference in NA results stems from the longer period of remediation of NA (30 years) relative to the FPMF due to the manufacture and operation of the CMT system.
CSS	0%	9%	81% *	GW	* Indicates an overall negative carbon performance (i.e., carbon sink)
WP (P retention)	0%	1%	0%	ME	–
WP (N retention)	0%	81% *	14%	FE	*Improved LOP retention stems from the increased green areas post remediation
GR	0%	7%	81% *	WC	*NbS increased GR corroborates previous results showing improved water filtration in hybrid poplar plantation (Dimitriou and Mola-Yudego, 2017)

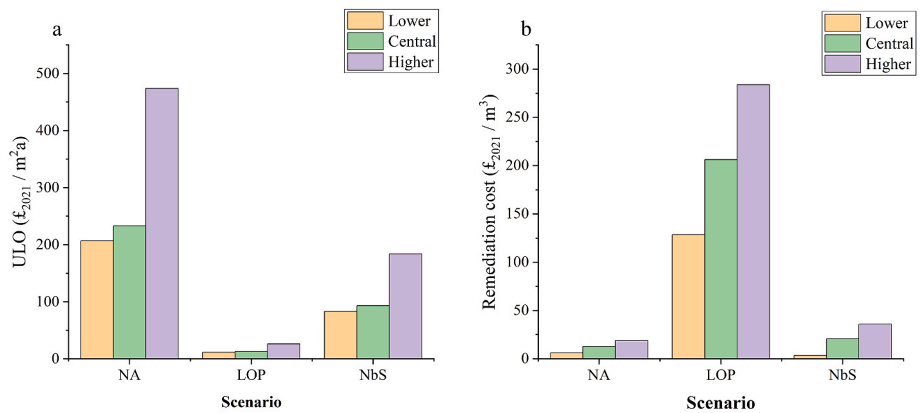

Fig. 5. Sensitivity analysis results of monetised ULO results and remediation costs.

filtration) are 3 because of the relative distance from the case study (the Greater London region). The deliberate decision to choose an avoided damage cost instead of hedonic pricing (willingness to pay) was found to have a positive impact on the DQIs of ES damage costs.

4.6.2. CFs (ES-LCA integration)

As shown in Fig. 6.e, The DQI results of CF exhibit an overall moderate uncertainty. The reliability DQIs of ES share a score of 2 (excepting AF) reflecting the spatially-explicit accounting of ES by the widely-used InVEST models (Chaplin-Kramer et al., 2017). The temporal specification DQI has a score of 3 across all the investigated ES because a lower score indicates a fully dynamic model which is not the case in this application but it might be possible with a dynamic LCA approach (Cardellini et al., 2018). The advantage of the refined approach in the remediation context (i.e., site-specific CFs) is demonstrated by the low uncertainty of the geographical specification DQIs, this was achieved using high-resolution spatial data in addition to the regional data of biophysical parameters of the ES accounting (see Appendix c). It is noteworthy that the variation of biogeographic features variations might impact the ES-LCA assessment results, therefore we recommend the use of sophisticated ES models that examine the effect of non-linear natural influences that might be missed by simple ES models as well as a stochastic approach to account for the variation across the different scenarios.

5. Overall discussion and concluding remarks

In this work, we propose a novel ecosystem services-life cycle assessment (ES-LCA) framework to allow the sustainability assessment of soil remediation and brownfield redevelopment. Additionally, guidance is provided for the transparent documentation of the assessment process. We investigate the tertiary impacts of remediation due to land use change by integrating spatially-explicit accounting of ecosystem services into the LCA methodology. Spatially-differentiated urban land use LCI flows were introduced exploiting the availability of high-resolution spatial data and well-documented site conditions of the case study. This approach enabled the use of site-specific data to characterise the performance of three remediation scenarios in an illustrative case study that is based on a historic mega remediation project (The London Olympic Park).

We also demonstrated some of the overlooked ES benefits of nature-based solutions for remediation such as carbon sequestration and storage and groundwater recharge potential. The temporal efficiency of remediation scenarios was assessed by accounting for urban land occupation and monetising the cost of the site being out of during remediation. The overall costs of remediation, ULO, and impacts on ES were aggregated into a single indicator to support the planning and decision-making of remediation projects. Although the framework is applied to an illustrative case study with some hypothetical elements, the robustness of our approach is demonstrated by DQIs and TEE results reflecting the advantages of hypothesizing alternative scenarios based on site data or well-documented technologies (i.e., phytoremediation). As discussed above, the monetary valuation of ES is likely to underestimate the full extent of impacts on ES.

Several challenges were identified while applying the ES-LCA to the case study. Firstly, operationalising the ES-LCA is a data-intensive and lengthy endeavour that exceeds the already time-consuming LCI considering that only 4 ES were modelled here.; thus, modelling additional ES will increase the complexity of ES-LCA modelling. Secondly, as noted by Rugani et al. and Liu et al. overlapping impacts on ecological processes and ecosystem services (steps III&IV) are still a challenge to disaggregate and model separately rather than in a combined manner although doing so could mask some of the hotspots of the ES cascade. Thirdly, modelling cultural ecosystem services (recreation) was attempted and partially successful (the LOP scenario) using the InVEST's Recreation model but could not be extended to the other two scenarios

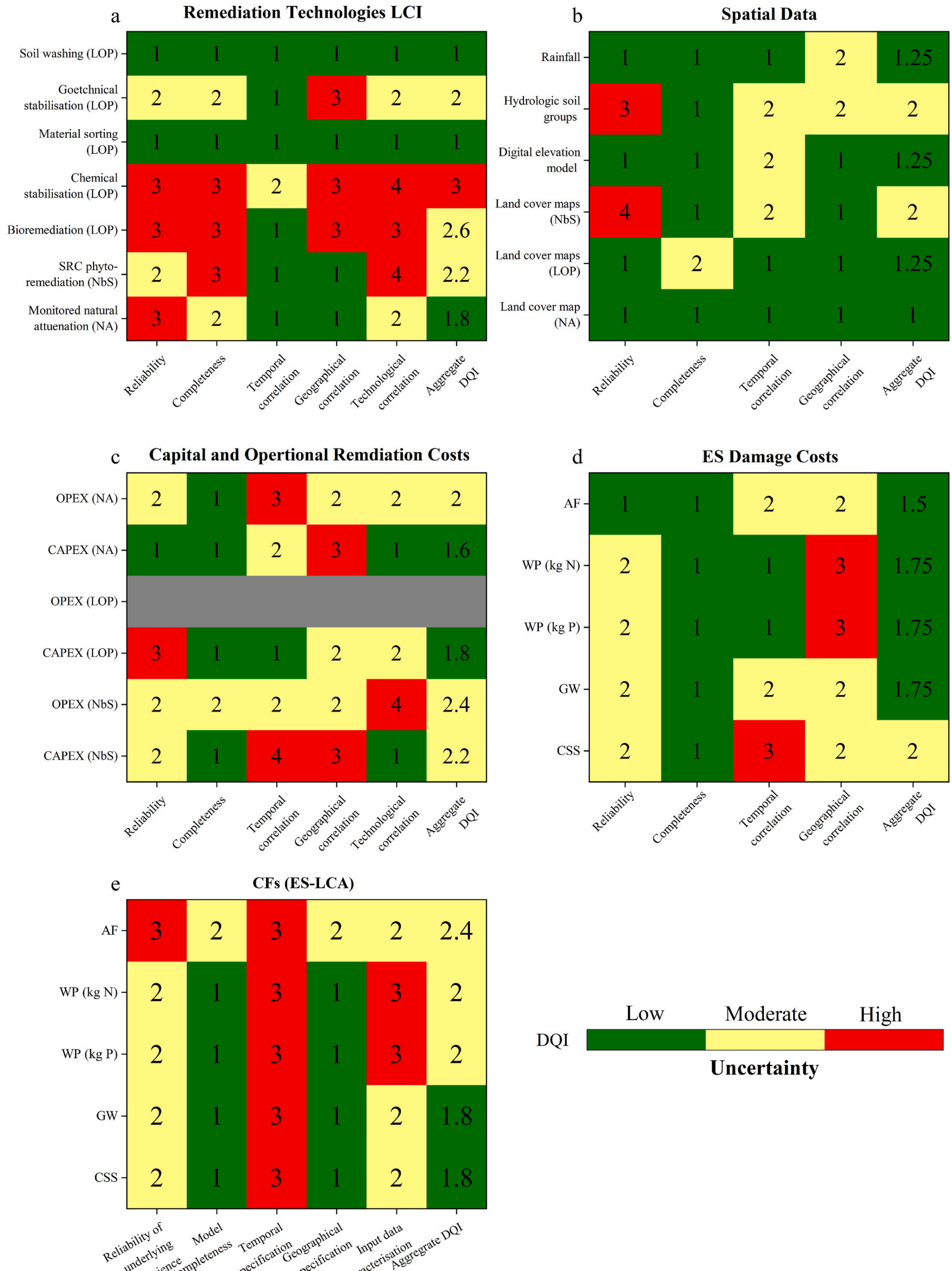

Fig. 6. Data quality indicator (DQI) results: a) LCI of remediation input data, b) spatial input data, c) CAPEX & OPEX input estimates of remediation, d) ES damage cost input estimates, e) CFs of ES-LCA integration.

because of the lack of data (geolocated photos) which does not exist for hypothetical scenarios (NA & NbS). Therefore, the decision was made to leave out cultural ES from this analysis because cultural ESs generated from a project of this scale would take generations to materialise (Neri, 2021) and would be better suited to a social life cycle assessment (sLCA) context (Alejandre et al., 2019). Fourthly, the dynamism of ecosystem services could not be fully captured here due to the simplification of InVEST models (Sharp et al., 2018) but other ES modelling suites could be useful such as the ARIES model (Villa et al., 2009). Fifthly, monetary discounting of ES benefits was not performed given the steady-state of eLCC and could potentially vary the results. Finally, though this work assessed the sensitivity and uncertainty of ES-LCA integration (which was not attempted before to our knowledge in the ES-LCA context) a quantitative uncertainty assessment of ES-LCA integration is yet to be performed.

In summary, we proposed a refined ES-LCA framework for the remediation context to improve the sustainability appraisal of remediation alternatives. The functionality of the framework is demonstrated in an illustrative case study involving a no-action scenario (monitored natural attenuation), energy-intensive scenario (LOP) scenario, and nature-based solution (NbS) scenario. High-resolution spatially-explicit accounting of ecosystem services using InVEST models was performed to calculate characterisation factors (CFs). The CFs were assigned to novel land LCI flows to characterise the tertiary impacts of remediation in terms of the ecosystem quality to deliver services. The characterised impacts and temporal efficiency of remediation were monetised and aggregated into a single indicator to support the remediation planners and decision-makers. The sensitivity and uncertainty of the framework were also investigated to ascertain the robustness of this approach. The results showed that despite the relatively longer remediation duration of the NbS scenario it would be the most cost-effective in terms of capital and operational costs, ES benefits as well as utilising the site. The overall advantages and drawbacks of the framework were also discussed to guide future efforts in this field.

CRediT authorship contribution statement

Khaled Alshehri: Conceptualization, Methodology, Software, Formal analysis, Investigation, Data curation, Visualization, Writing – original draft. **Michael Harbottle:** Conceptualization, Methodology, Writing – review & editing, Supervision. **Devin Sapsford:** Conceptualization, Methodology, Writing – review & editing, Supervision. **Alistair Beames:** Writing – review & editing. **Peter Cleall:** Conceptualization, Methodology, Writing – review & editing, Supervision.

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

The authors do not have permission to share data.

Acknowledgements

The authors are thankful to the Deanship of Scientific Research at the University of Bisha for the financial support for Khaled Alshehri through the Scholarship Program of the University. The authors wish to thank Dr. Riccardo Maddalena, Cardiff University for facilitating access to openLCA.

Supplementary data

Supplementary data (Appendices B & C) to this article can be found online at https://doi.org/10.1016/j.jclepro.2023.137352.

Appendix A: Abbreviations

Abbreviation	*Full form*
AF	Air filtration
CAPEX	Capital expenditure
CF	Characterisation factor
CICES	The Common International Classification of Ecosystem Services
CSS	Carbon storage and sequestration
DQI	Data quality indicator
ES	Ecosystem services
FE	Freshwater eutrophication
FMPF	Fine particulate matter formation
FU	Functional unit
GR	Groundwater recharge
GS	Geotechnical stabilisation
GWP	Global warming potential
InVEST	Integrated Valuation of Ecosystem Services
LANCA	Land use indicator valuation calculator
LCA	Life cycle assessment
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
LOP	London Olympic Park
LU	Land use
ME	Marine eutrophication
MRP	A mix of regional biome
NA	No Action
NbS	Nature-based solution
OPEX	Operational expenditure
PNV	Potential natural vegetation
SOC	Soil organic content
SW	Soil washing
TEE	Truncation error estimate
TESSA	Toolkit for Ecosystem Service Site-Based Assessment
TRACI	Tool for Reduction and Assessment of Chemicals and Other Environmental Impacts
UK CEH	United Kingdom's Centre for Ecology and Hydrology
ULO	Urban land occupation
WP	Water purification
UNEP	United Nations Environment Programme
SETAC	Society of Environmental Toxicology and Chemistry
TS	Transport of samples
MMW	Manufacturing of monitoring wells

References

Adamson, D., Newell, C., 2014. Frequently Asked Questions about Monitored Natural Attenuation in Groundwater: Defense Technical Information Center. https://doi.org/10.21236/ADA627131.

Adhikari, K., Hartemink, A.E., 2016. Linking soils to ecosystem services—a global review. Geoderma 262, 101–111. https://doi.org/10.1016/j.geoderma.2015.08.009.

Alejandre, E.M., van Bodegom, P.M., Guinée, J.B., 2019. Towards an optimal coverage of ecosystem services in LCA. J. Clean. Prod. 231, 714–722. https://doi.org/10.1016/j.jclepro.2019.05.284.

Alker, S., Joy, V., Roberts, P., Smith, N., 2000. The definition of brownfield. J. Environ. Plann. Manag. 43 (1), 49–69. https://doi.org/10.1080/09640560010766.

Allacker, K., Souza, D. M. de, Sala, S., 2014. Land use impact assessment in the construction sector: an analysis of LCIA models and case study application. Int. J. Life Cycle Assess. 19 (11), 1799–1809. https://doi.org/10.1007/s11367-014-0781-7.

Alves, K. de F., Andrade, E.P., Savioli, J.P., Pastor, A.V., de Figueirêdo, M.C.B., Ugaya, C.M.L., 2020. Water scarcity in Brazil: Part 2—uncertainty assessment in regionalized characterization factors. Int. J. Life Cycle Assess. 25 (12), 2359–2379. https://doi.org/10.1007/s11367-020-01739-3.

Amponsah, N.Y., Wang, J., Zhao, L., 2018. A review of life cycle greenhouse gas (GHG) emissions of commonly used ex-situ soil treatment technologies. J. Clean. Prod. 186, 514–525. https://doi.org/10.1016/j.jclepro.2018.03.164.

Andrea, F., Bini, C., Amaducci, S., 2018. Soil and ecosystem services: current knowledge and evidences from Italian case studies. Appl. Soil Ecol. 123, 693–698. https://doi.org/10.1016/j.apsoil.2017.06.031.

Annex 1, 2011. The SuRF-UK Indicator Set for Sustainable Remediation Assessment. AIRE Publications, CL.

Arbault, D., Rivière, M., Rugani, B., Benetto, E., Tiruta-Barna, L., 2014. Integrated earth system dynamic modeling for life cycle impact assessment of ecosystem services. Sci. Total Environ. 472, 262–272. https://doi.org/10.1016/j.scitotenv.2013.10.099.

Bakshi, B.R., Ziv, G., Lepech, M.D., 2015. Techno-ecological synergy: a framework for sustainable engineering. Environ. Sci. Technol. 49 (3), 1752–1760. https://doi.org/10.1021/es5041442.

Bare, J., 2011. TRACI 2.0: the tool for the reduction and assessment of chemical and other environmental impacts 2.0. Clean Technol. Environ. Policy 13 (5), 687–696. https://doi.org/10.1007/s10098-010-0338-9.

Beames, A., Broekx, S., Heijungs, R., Lookman, R., Boonen, K., Van Geert, Y., Dendoncker, K., Seuntjens, P., 2015. Accounting for land-use efficiency and temporal variations between brownfield remediation alternatives in life-cycle assessment. J. Clean. Prod. 101, 109–117. https://doi.org/10.1016/j.jclepro.2015.03.073.

Beck, T., Bos, U., Wittstock, B., Baitz, M., 2010. LANCA—Land Use Indicator Value Calculation in Life Cycle Assessment. Fraunhofer-Institut für Bauphysik.

Berghöfer, A., Mader, A., Patrickson, S., Calcaterra, E., Smit, J., Blignaut, J., de Wit, M., van Zyl, H., 2011. TEEB Manual for cities: ecosystem services in urban management. In: The Economics of Ecosystems and Biodiversity (Suiza).

Bolund, P., Hunhammar, S., 1999. Ecosystem services in urban areas. Ecol. Econ. 29 (2), 293–301. https://doi.org/10.1016/S0921-8009(99)00013-0.

Boulay, A.-M., Bare, J., Benini, L., Berger, M., Lathuillière, M.J., Manzardo, A., Margni, M., Motoshita, M., Núñez, M., Pastor, A.V., Ridoutt, B., Oki, T., Worbe, S., Pfister, S., 2018. The WULCA consensus characterization model for water scarcity footprints: assessing impacts of water consumption based on available water remaining (AWARE). Int. J. Life Cycle Assess. 23 (2), 368–378. https://doi.org/10.1007/s11367-017-1333-8.

Bouma, J.A., Van Beukering, P.J., 2015. Ecosystem Services: from Concept to Practice. Cambridge University Press.

Britain, Great, Treasury, 2022. The Green Book: Central Government Guidance on Appraisal and Evaluation.

Brown, G., Fagerholm, N., 2015. Empirical PPGIS/PGIS mapping of ecosystem services: a review and evaluation. Ecosyst. Serv. 13, 119–133. https://doi.org/10.1016/j.ecoser.2014.10.007.

Bulle, C., Margni, M., Patouillard, L., Boulay, A.-M., Bourgault, G., De Bruille, V., Cao, V., Hauschild, M., Henderson, A., Humbert, S., Kashef-Haghighi, S., Kounina, A., Laurent, A., Levasseur, A., Liard, G., Rosenbaum, R.K., Roy, P.-O., Shaked, S., Fantke, P., Jolliet, O., 2019. IMPACT World+: a globally regionalized life cycle impact assessment method. Int. J. Life Cycle Assess. 24 (9), 1653–1674. https://doi.org/10.1007/s11367-019-01583-0.

Campagin to Protect Rural England, 2018. State of Brownfield 2018: an analysis demonstrating the potential of brownfield land for housing. https://www.cpre.org.uk/wp-content/uploads/2019/11/StateZofZBrownfieldZ2018.pdf.

Cao, V., Margni, M., Favis, B.D., Deschênes, L., 2015. Aggregated indicator to assess land use impacts in life cycle assessment (LCA) based on the economic value of ecosystem services. J. Clean. Prod. 94, 56–66. https://doi.org/10.1016/j.jclepro.2015.01.041.

Cappuyns, V., 2011. Possibilities and Limitations of LCA for the Evaluation of Soil Remediation and Cleanup, pp. 213–223. https://doi.org/10.2495/CHEM110201.

Cardellini, G., Mutel, C.L., Vial, E., Muys, B., 2018. Temporalis, a generic method and tool for dynamic Life Cycle Assessment. Sci. Total Environ. 645, 585–595. https://doi.org/10.1016/j.scitotenv.2018.07.044.

Carey, M.A., 2000. Environment agency, WHO, water research Centre, & WRC. In: Guidance on the Assessment and Monitoring of Natural Attentuation of Contaminants in Groundwater.

Cervelli, E., Pindozzi, S., Capolupo, A., Okello, C., Rigillo, M., Boccia, L., 2016. Ecosystem services and bioremediation of polluted areas. Ecol. Eng. 87, 139–149. https://doi.org/10.1016/j.ecoleng.2015.09.045.

Chandra, R., Dubey, N.K., Kumar, V., 2017. Phytoremediation of Environmental Pollutants. CRC Press.

Chaplin-Kramer, R., Sim, S., Hamel, P., Bryant, B., Noe, R., Mueller, C., Rigarlsford, G., Kulak, M., Kowal, V., Sharp, R., Clavreul, J., Price, E., Polasky, S., Ruckelshaus, M., Daily, G., 2017. Life cycle assessment needs predictive spatial modelling for biodiversity and ecosystem services. Nat. Commun. 8 (1), 15065 https://doi.org/10.1038/ncomms15065.

Chen, I.-C., Chuo, Y.-Y., Ma, H., 2019. Uncertainty analysis of remediation cost and damaged land value for brownfield investment. Chemosphere 220, 371–380. https://doi.org/10.1016/j.chemosphere.2018.12.116.

Chen, H.-P., Lee, M., Chiueh, P.-T., 2021. Creating ecosystem services assessment models incorporating land use impacts based on soil quality. Sci. Total Environ. 773, 145018 https://doi.org/10.1016/j.scitotenv.2021.145018.

Christie, J.M., 1994. Great Britain, & forestry commission. In: Provisional Yield Tables for Poplar in Britain. Forestry Commission.

Ciroth, A., Fleischer, G., Steinbach, J., 2004. Uncertainty calculation in life cycle assessments: a combined model of simulation and approximation. Int. J. Life Cycle Assess. 9 (4), 216. https://doi.org/10.1007/BF02978597.

Ciroth, A., Muller, S., Weidema, B., Lesage, P., 2016. Empirically based uncertainty factors for the pedigree matrix in ecoinvent. Int. J. Life Cycle Assess. 21 (9), 1338–1348. https://doi.org/10.1007/s11367-013-0670-5.

CL:AIRE, 2002. CL:AIRE Technical Bulletin: Multilevel Sampling Systems.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M., 1997. The value of the world's ecosystem services and natural capital. Nature 387 (6630), 253–260. https://doi.org/10.1038/387253a0.

Costanza, R., de Groot, R., Braat, L., Kubiszewski, I., Fioramonti, L., Sutton, P., Farber, S., Grasso, M., 2017. Twenty years of ecosystem services: how far have we come and how far do we still need to go? Ecosyst. Serv. 28, 1–16. https://doi.org/10.1016/j.ecoser.2017.09.008.

Cundy, A.B., Bardos, R.P., Church, A., Puschenreiter, M., Friesl-Hanl, W., Müller, I., Neu, S., Mench, M., Witters, N., Vangronsveld, J., 2013. Developing principles of sustainability and stakeholder engagement for "gentle" remediation approaches: the European context. J. Environ. Manag. 129, 283–291. https://doi.org/10.1016/j.jenvman.2013.07.032.

Cundy, A.B., Bardos, R.P., Puschenreiter, M., Mench, M., Bert, V., Friesl-Hanl, W., Müller, I., Li, X.N., Weyens, N., Witters, N., Vangronsveld, J., 2016. Brownfields to green fields: realising wider benefits from practical contaminant phytomanagement strategies. J. Environ. Manag. 184, 67–77. https://doi.org/10.1016/j.jenvman.2016.03.028.

da S Trentin, A.W., Reddy, K.R., Kumar, G., Chetri, J.K., Thomé, A., 2019. Quantitative assessment of life cycle sustainability (QUALICS): framework and its application to assess electrokinetic remediation. Chemosphere 230, 92–106. https://doi.org/10.1016/j.chemosphere.2019.04.200.

De Valck, J., Beames, A., Liekens, I., Bettens, M., Seuntjens, P., Broekx, S., 2019. Valuing urban ecosystem services in sustainable brownfield redevelopment. Ecosyst. Serv. 35, 139–149. https://doi.org/10.1016/j.ecoser.2018.12.006.

Declercq, I., Cappuyns, V., Duclos, Y., 2012. Monitored natural attenuation (MNA) of contaminated soils: state of the art in Europe—a critical evaluation. Sci. Total Environ. 426, 393–405. https://doi.org/10.1016/j.scitotenv.2012.03.040.

Diamond, M.L., Page, C.A., Campbell, M., McKenna, S., Lall, R., 1999. Life-cycle framework for assessment of site remediation options: method and generic survey. Environ. Toxicol. Chem. 18 (4), 788–800. https://doi.org/10.1002/etc.5620180427.

Dickinson, N.M., 2000. Strategies for sustainable woodland on contaminated soils. Chemosphere 41 (1–2), 259–263. https://doi.org/10.1016/S0045-6535(99)00419-1.

Dimitriou, I., Mola-Yudego, B., 2017. Impact of populus plantations on water and soil quality. BioEnergy Res. 10 (3), 750–759. https://doi.org/10.1007/s12155-017-9836-5.

Ditor, M., 2010. Use of Site-specific Data to Assess Monitored Natural Attenuation and Remediation of a Contaminated Site with Life Cycle Assessment. Library and Archives Canada = Bibliothèque et Archives Canada.

Efroymson, R.A., Nicolette, J.P., Suter, G.W., 2004. A framework for net environmental benefit analysis for remediation or restoration of contaminated sites. Environ. Manag. 34 (3), 315–331. https://doi.org/10.1007/s00267-004-0089-7.

Elmqvist, T., Setälä, H., Handel, S., Van Der Ploeg, S., Aronson, J., Blignaut, J.N., Gomez-Baggethun, E., Nowak, D., Kronenberg, J., De Groot, R., 2015. Benefits of restoring ecosystem services in urban areas. Curr. Opin. Environ. Sustain. 14, 101–108.

Espada, J.J., Rodríguez, R., Gari, V., Salcedo-Abraira, P., Bautista, L.F., 2022. Coupling phytoremediation of Pb-contaminated soil and biomass energy production: a comparative Life Cycle Assessment. Sci. Total Environ. 840, 156675 https://doi.org/10.1016/j.scitotenv.2022.156675.

ESVD (n.d.). Retrieved 6 October 2022, from. https://www.esvd.net/.

European Commission, 2010a. Joint research Centre. Institute for environment and sustainability. In: International Reference Life Cycle Data System (ILCD) Handbook: Framework and Requirements for Life Cycle Impact Assessment Models and Indicators. Publications Office. https://data.europa.eu/doi/10.2788/38719.

European Commission, 2010b. Joint research Centre. Institute for environment and sustainability. In: International Reference Life Cycle Data System (ILCD) Handbook : general Guide for Life Cycle Assessment: Detailed Guidance. Publications Office. https://data.europa.eu/doi/10.2788/38479.

European Commission. Joint Research Centre, 2017. European Achievements in Soil Remediation and Brownfield Redevelopment :a Report of the European Information and Observation Network's National Reference Centres for Soil (Eionet NRC Soil). Publications Office. https://data.europa.eu/doi/10.2760/91268.

Favara, P.J., Krieger, T.M., Boughton, B., Fisher, A.S., Bhargava, M., 2011. Guidance for performing footprint analyses and life-cycle assessments for the remediation industry. Remed. J. 21 (3), 39–79. https://doi.org/10.1002/rem.20289.

Finnveden, G., Östlund, P., 1997. Exergies of natural resources in life-cycle assessment and other applications. Energy 22 (9), 923–931. https://doi.org/10.1016/S0360-5442(97)00022-4.

Frischknecht, R., 1998. Life cycle inventory analysis for decision-making: scope-dependent inventory system models and context-specific joint product allocation. Int. J. Life Cycle Assess. 3 (2) https://doi.org/10.1007/BF02978487, 67–67.

Funtowicz, S.O., Ravetz, J.R., 1990. Uncertainty and Quality in Science for Policy, vol. 15. Springer Science & Business Media.

Georges, M., 2018. Environmental Costs and Benefits at the Site of the London 2012 Olympics. Environment Agency. https://www.gov.uk/government/publications/environmental-costs-and-benefits-at-the-site-of-the-london-2012-olympics.

Gordon, M., Choe, N., Duffy, J., Ekuan, G., Heilman, P., Muiznieks, I., Ruszaj, M., Shurtleff, B.B., Strand, S., Wilmoth, J., Newman, L.A., 1998a. Phytoremediation of trichloroethylene with hybrid poplars. Environ. Health Perspect. 106, 4.

Gordon, M., Choe, N., Duffy, J., Ekuan, G., Heilman, P., Muiznieks, I., Ruszaj, M., Shurtleff, B.B., Strand, S., Wilmoth, J., Newman, L.A., 1998b. Phytoremediation of trichloroethylene with hybrid poplars. Environ. Health Perspect. 106 (Suppl. 4), 1001–1004. https://doi.org/10.1289/ehp.98106s41001.

Greenhalgh, S., Samarasinghe, O., Curran-Cournane, F., Wright, W., Brown, P., 2017. Using ecosystem services to underpin cost–benefit analysis: is it a way to protect finite soil resources? Ecosyst. Serv. 27, 1–14. https://doi.org/10.1016/j.ecoser.2017.07.005.

Grifoni, M., Franchi, E., Fusini, D., Vocciante, M., Barbafieri, M., Pedron, F., Rosellini, I., Petruzzelli, G., 2022. Soil remediation: towards a resilient and adaptive approach to deal with the ever-changing environmental challenges. Environments 9 (2), 18. https://doi.org/10.3390/environments9020018.

Hassan, R.M., Scholes, R.J., Ash, N., 2005. Millennium ecosystem assessment (Program). In: Ecosystems and Human Well-Being: Current State and Trends: Findings of the Condition and Trends Working Group of the Millennium Ecosystem Assessment. Island Press.

Hauschild, M.Z., Rosenbaum, R.K., Olsen, S.I. (Eds.), 2018. Life Cycle Assessment. Springer International Publishing. https://doi.org/10.1007/978-3-319-56475-3.

Heijungs, R., 1996. Identification of key issues for further investigation in improving the reliability of life-cycle assessments. J. Clean. Prod. 4 (3–4), 159–166. https://doi.org/10.1016/S0959-6526(96)00042-X.

Hellings, J., Lass, M., Apted, J., Mead, I., 2011. Delivering London 2012: geotechnical enabling works. Proc. Inst. Civil Eng. - Civil Eng. 164 (6), 5–10. https://doi.org/10.1680/cien.2011.164.6.5.

Hengl, T., Walsh, M.G., Sanderman, J., Wheeler, I., Harrison, S.P., Prentice, I.C., 2018. Global mapping of potential natural vegetation: an assessment of machine learning algorithms for estimating land potential. PeerJ 6, e5457. https://doi.org/10.7717/peerj.5457.

Holder, A.J., Rowe, R., McNamara, N.P., Donnison, I.S., McCalmont, J.P., 2019. Soil & Water Assessment Tool (SWAT) simulated hydrological impacts of land use change from temperate grassland to energy crops: a case study in western UK. GCB Bioenergy 11 (11), 1298–1317. https://doi.org/10.1111/gcbb.12628.

Hou, D., Al-Tabbaa, A., Guthrie, P., Hellings, J., Gu, Q., 2014. Using a hybrid LCA method to evaluate the sustainability of sediment remediation at the London Olympic Park. J. Clean. Prod. 83, 87–95. https://doi.org/10.1016/j.jclepro.2014.07.062.

Hou, D., Song, Y., Zhang, J., Hou, M., O'Connor, D., Harclerode, M., 2018. Climate change mitigation potential of contaminated land redevelopment: a city-level assessment method. J. Clean. Prod. 171, 1396–1406. https://doi.org/10.1016/j.jclepro.2017.10.071.

Hou, D., O'Connor, D., Igalavithana, A.D., Alessi, D.S., Luo, J., Tsang, D.C.W., Sparks, D.L., Yamauchi, Y., Rinklebe, J., Ok, Y.S., 2020. Metal contamination and bioremediation of agricultural soils for food safety and sustainability. Nat. Rev. Earth Environ. 1 (7), 366–381. https://doi.org/10.1038/s43017-020-0061-y.

Hou, D., Al-Tabbaa, A., O'Connor, D., Hu, Q., Zhu, Y.-G., Wang, L., Kirkwood, N., Ok, Y.S., Tsang, D.C.W., Bolan, N.S., Rinklebe, J., 2023. Sustainable remediation and redevelopment of brownfield sites. Nat. Rev. Earth Environ. https://doi.org/10.1038/s43017-023-00404-1.

Huijbregts, M.A.J., Steinmann, Z.J.N., Elshout, P.M.F., Stam, G., Verones, F., Vieira, M., Zijp, M., Hollander, A., van Zelm, R., 2017. ReCiPe2016: a harmonised life cycle impact assessment method at midpoint and endpoint level. Int. J. Life Cycle Assess. 22 (2), 138–147. https://doi.org/10.1007/s11367-016-1246-y.

Hunkeler, D.J., Hunkeler, D., Lichtenvort, K., Rebitzer, G., Ciroth, A. (Eds.), 2008. Environmental Life Cycle Costing. CRC Press.

Hutcheson, W., Hoagland, P., Jin, D., 2018. Valuing environmental education as a cultural ecosystem service at Hudson River Park. Ecosyst. Serv. 31, 387–394. https://doi.org/10.1016/j.ecoser.2018.03.005.

Igos, E., Benetto, E., Meyer, R., Baustert, P., Othoniel, B., 2019. How to treat uncertainties in life cycle assessment studies? Int. J. Life Cycle Assess. 24 (4), 794–807. https://doi.org/10.1007/s11367-018-1477-1.

Ingwersen, W.W., 2011. Emergy as a life cycle impact assessment indicator: a gold mining case study. J. Ind. Ecol. 15 (4), 550–567. https://doi.org/10.1111/j.1530-9290.2011.00333.x.

Irvine, K., Choy, B.H., Chua, L.H., Gaut, J., Ho, H.L., Tontisirin, N., 2020. Hedonic pricing to monetize ecosystem services provided by water sensitive urban design: a comparison of geelong, Australia and Singapore. Nakhara : J. Environ. Design Plann. 19, 59–78. https://doi.org/10.54028/NJ2020195978.

IS/ISO 14044, 2006. Environmental Management-Life Cycle Assessment-Requirements and Guidelines, p. 52 n.d.

ISO 14008, 2019. Monetary valuation of environmental impacts and related environmental aspects. Pub. L. No. 03/2019, 13.020.20 ISO/TC 207/SC 1 33. https://www.iso.org/standard/43243.html.

IUCN, 2020. International union for conservation of nature. In: IUCN Global Standard for Nature-Based Solutions: A User-Friendly Framework for the Verification, Design and Scaling up of NbS, first ed. IUCN, International Union for Conservation of Nature. https://doi.org/10.2305/IUCN.CH.2020.08.en.

Jones, L., Vieno, M., Morton, D., Hall, J., Carnell, E., Nemitz, E., Beck, R., Ries, S., Pritchard, N., Hayes, F., Mills, G., Cryle, P., Dickie, I., Koshy, I., Holland, M., 2017. Developing estimates for the valuation of air pollution removal in ecosystem accounts. In: Final Report for Office of National Statistics. Final report for Office of National Statistics. https://nora.nerc.ac.uk/id/eprint/524081/7/N524081RE.pdf.

Jorat, M.E., Goddard, M.A., Manning, P., Lau, H.K., Ngeow, S., Sohi, S.P., Manning, D.A.C., 2020. Passive CO2 removal in urban soils: evidence from brownfield sites. Sci. Total Environ. 703, 135573 https://doi.org/10.1016/j.scitotenv.2019.135573.

Junnila, S., Horvath, A., 2003. Life-cycle environmental effects of an office building. J. Infrastruct. Syst. 9 (4), 157–166. https://doi.org/10.1061/(ASCE)1076-0342(2003)9:4(157).

Koch, M., 2019. Forest talk: how trees communicate. https://www.overdrive.com/search?q=2266DD88-6D3F-4D3F-B4D8-3FA271712082.

Koellner, T., de Baan, L., Beck, T., Brandão, M., Civit, B., Goedkoop, M., Margni, M., i Canals, L.M., Müller-Wenk, R., Weidema, B., Wittstock, B., 2013a. Principles for life cycle inventories of land use on a global scale. Int. J. Life Cycle Assess. 18 (6) https://doi.org/10.1007/s11367-012-0392-0. Article 6.

Koellner, T., de Baan, L., Beck, T., Brandão, M., Civit, B., Margni, M., i Canals, L.M., Saad, R., de Souza, D.M., Müller-Wenk, R., 2013b. UNEP-SETAC guideline on global land use impact assessment on biodiversity and ecosystem services in LCA. Int. J. Life Cycle Assess. 18 (6) https://doi.org/10.1007/s11367-013-0579-z. Article 6.

Kuminoff, N.V., Parmeter, C.F., Pope, J.C., 2010. Which hedonic models can we trust to recover the marginal willingness to pay for environmental amenities? J. Environ. Econ. Manag. 60 (3), 145–160. https://doi.org/10.1016/j.jeem.2010.06.001.

Land value estimates for policy appraisal, 2015. Department for Communities and Local Government.

Lankia, T., Neuvonen, M., Pouta, E., Sievänen, T., Torvelainen, J., 2020. Outdoor recreation in ecosystem service accounting: pilot accounts from Finland. Scand. J. For. Res. 35 (3–4), 186–197. https://doi.org/10.1080/02827581.2020.1760342.

Larondelle, N., Lauf, S., 2016. Balancing demand and supply of multiple urban ecosystem services on different spatial scales. Ecosyst. Serv. 22, 18–31. https://doi.org/10.1016/j.ecoser.2016.09.008.

Lemming, G., Hauschild, M.Z., Chambon, J., Binning, P.J., Bulle, C., Margni, M., Bjerg, P.L., 2010. Environmental impacts of remediation of a trichloroethene-contaminated site: life cycle assessment of remediation alternatives. Environ. Sci. Technol. 44 (23), 9163–9169. https://doi.org/10.1021/es102007s.

Leroy, Y., Froelich, D., 2010. Qualitative and quantitative approaches dealing with uncertainty in life cycle assessment (LCA) of complex systems: towards a selective integration of uncertainty according to LCA objectives. Int. J. Des. Eng. 3 (2), 151. https://doi.org/10.1504/IJDE.2010.034862.

Lesage, P., Ekvall, T., Deschênes, L., Samoson, R., 2007. Environmental assessment of brownfield rehabilitation using two different life cycle inventory models: Part 1: methodological approach. Int. J. Life Cycle Assess. 12 (6), 391–398. https://doi.org/10.1065/lca2006.10.279.1.

Lewandowski, I., Schmidt, U., Londo, M., Faaij, A., 2006. The economic value of the phytoremediation function – assessed by the example of cadmium remediation by willow (Salix ssp). Agric. Syst. 89 (1), 68–89. https://doi.org/10.1016/j.agsy.2005.08.004.

Lilburne, L., Eger, A., Mudge, P., Ausseil, A.-G., Stevenson, B., Herzig, A., Beare, M., 2020. The Land Resource Circle: supporting land-use decision making with an ecosystem-service-based framework of soil functions. Geoderma 363, 114134. https://doi.org/10.1016/j.geoderma.2019.114134.

Lilien, J.P., Jin, H., Gloria, T.P., 2014. Implementation of life cycle assessment at chevron: lessons learned and good practices. Day 1 Mon, March 17, D011S013R004. https://doi.org/10.2118/168490-MS, 2014.

Lin, Y., Wu, B., Wang, D., Xiao, W., Huang, Y., Fu, S., He, Y., 2021. Comprehensive evaluation of ecosystem services for brownfield redevelopment in Changsha. Arabian J. Geosci. 14 (11), 1021. https://doi.org/10.1007/s12517-021-07261-6.

Liu, X., Ziv, G., Bakshi, B.R., 2018. Ecosystem services in life cycle assessment - Part 2: adaptations to regional and serviceshed information. J. Clean. Prod. 197, 772–780. https://doi.org/10.1016/j.jclepro.2018.05.283.

Liu, X., Bakshi, B.R., Rugani, B., de Souza, D.M., Bare, J., Johnston, J.M., Laurent, A., Verones, F., 2020. Quantification and valuation of ecosystem services in life cycle assessment: application of the cascade framework to rice farming systems. Sci. Total Environ. 747, 141278 https://doi.org/10.1016/j.scitotenv.2020.141278.

Lloyd, S.M., Ries, R., 2008. Characterizing, propagating, and analyzing uncertainty in life-cycle assessment: a survey of quantitative approaches. J. Ind. Ecol. 11 (1), 161–179. https://doi.org/10.1162/jiec.2007.1136.

Lo Piano, S., Benini, L., 2022. A critical perspective on uncertainty appraisal and sensitivity analysis in life cycle assessment. J. Ind. Ecol. 26 (3), 763–781. https://doi.org/10.1111/jiec.13237.

Maurice, B., Frischknecht, R., Coelho-Schwirtz, V., Hungerbühler, K., 2000. Uncertainty analysis in life cycle inventory. Application to the production of electricity with French coal power plants. J. Clean. Prod. 8 (2), 95–108. https://doi.org/10.1016/S0959-6526(99)00324-8.

Mead, I., Apted, J., Sharif, S., 2013. Delivering London 2012: contaminated soil treatment at the olympic park. Proc. Inst. Civil Eng. - Geotech. Eng. 166 (1), 8–17. https://doi.org/10.1680/geng.11.00109.

Metzger, M.J., Rounsevell, M.D.A., Acosta-Michlik, L., Leemans, R., Schröter, D., 2006. The vulnerability of ecosystem services to land use change. Agric. Ecosyst. Environ. 114 (1), 69–85. https://doi.org/10.1016/j.agee.2005.11.025.

Milà i Canals, L., Bauer, C., Depestele, J., Dubreuil, A., Freiermuth Knuchel, R., Gaillard, G., Michelsen, O., Müller-Wenk, R., Rydgren, B., 2007. Key elements in a framework for land use impact assessment within LCA (11 pp). Int. J. Life Cycle Assess. 12 (1), 5–15. https://doi.org/10.1065/lca2006.05.250.

Minixhofer, P., Stangl, R., 2021. Green infrastructures and the consideration of their soil-related ecosystem services in urban areas—a systematic literature review. Sustainability 13 (6), 3322. https://doi.org/10.3390/su13063322.

Morton, R.D., Rowland, C.S., Wood, C.M., Meek, L., Marston, C.G., Smith, G.M., 2014. Land Cover Map 2007 (25m Raster, GB) v1.2. NERC Environmental Information Data Centre. https://doi.org/10.5285/A1F88807-4826-44BC-994D-A902DA5119C2 [Image/tiff TIFF].

Mutel, C., Liao, X., Patouillard, L., Bare, J., Fantke, P., Frischknecht, R., Hauschild, M., Jolliet, O., Maia de Souza, D., Laurent, A., Pfister, S., Verones, F., 2019. Overview and recommendations for regionalized life cycle impact assessment. Int. J. Life Cycle Assess. 24 (5) https://doi.org/10.1007/s11367-018-1539-4. Article 5.

Neri, M.C. (Ed.), 2021. Evaluating the Local Impacts of the Rio Olympics. Routledge, Taylor & Francis Group.

Onwubuya, K., Cundy, A., Puschenreiter, M., Kumpiene, J., Bone, B., Greaves, J., Teasdale, P., Mench, M., Tlustos, P., Mikhalovsky, S., Waite, S., Friesl-Hanl, W., Marschner, B., Müller, I., 2009. Developing decision support tools for the selection of "gentle" remediation approaches. Sci. Total Environ. 407 (24), 6132–6142. https://doi.org/10.1016/j.scitotenv.2009.08.017.

Othoniel, B., Rugani, B., Heijungs, R., Beyer, M., Machwitz, M., Post, P., 2019. An improved life cycle impact assessment principle for assessing the impact of land use on ecosystem services. Sci. Total Environ. 693, 133374 https://doi.org/10.1016/j.scitotenv.2019.07.180.

O'Connor, D., Zheng, X., Hou, D., Shen, Z., Li, G., Miao, G., O'Connell, S., Guo, M., 2019. Phytoremediation: climate change resilience and sustainability assessment at a coastal brownfield redevelopment. Environ. Int. 130, 104945 https://doi.org/10.1016/j.envint.2019.104945.

O'Riordan, R., Davies, J., Stevens, C., Quinton, J.N., Boyko, C., 2021. The ecosystem services of urban soils: a review. Geoderma 395, 115076. https://doi.org/10.1016/j.geoderma.2021.115076.
Park, Y.S., Egilmez, G., Kucukvar, M., 2016. Emergy and end-point impact assessment of agricultural and food production in the United States: a supply chain-linked Ecologically-based Life Cycle Assessment. Ecol. Indicat. 62, 117–137. https://doi.org/10.1016/j.ecolind.2015.11.045.
Peh, K.S.-H., Balmford, A., Bradbury, R.B., Brown, C., Butchart, S.H.M., Hughes, F.M.R., Stattersfield, A., Thomas, D.H.L., Walpole, M., Bayliss, J., Gowing, D., Jones, J.P.G., Lewis, S.L., Mulligan, M., Pandeya, B., Stratford, C., Thompson, J.R., Turner, K., Vira, B., et al., 2013. TESSA: a toolkit for rapid assessment of ecosystem services at sites of biodiversity conservation importance. Ecosyst. Serv. 5, 51–57. https://doi.org/10.1016/j.ecoser.2013.06.003.
Potschin, M., Haines-Young, R.H., Fish, R., Turner, R.K. (Eds.), 2016. Routledge Handbook of Ecosystem Services, Routledge. Taylor & Francis Group.
Potschin-Young, M., Haines-Young, R., Görg, C., Heink, U., Jax, K., Schleyer, C., 2018. Understanding the role of conceptual frameworks: reading the ecosystem service cascade. Ecosyst. Serv. 29, 428–440. https://doi.org/10.1016/j.ecoser.2017.05.015.
Puccini, M., Seggiani, M., Vitolo, S., Iannelli, R., 2013. Life cycle assessment of remediation alternatives for dredged sediments. Chem. Eng. Trans. 35, 781–786. https://doi.org/10.3303/CET1335130.
Qin, Y., Cucurachi, S., Suh, S., 2020. Perceived uncertainties of characterization in LCA: a survey. Int. J. Life Cycle Assess. 25 (9), 1846–1858. https://doi.org/10.1007/s11367-020-01787-9.
Rate, A.W. (Ed.), 2022. Urban Soils: Principles and Practice. Springer International Publishing. https://doi.org/10.1007/978-3-030-87316-5.
Reddy, K.R., Kadlec, R.H., Flaig, E., Gale, P.M., 1999. Phosphorus retention in streams and wetlands: a review. Crit. Rev. Environ. Sci. Technol. 29 (1), 83–146. https://doi.org/10.1080/10643389991259182.
Redhead, J.W., May, L., Oliver, T.H., Hamel, P., Sharp, R., Bullock, J.M., 2018. National scale evaluation of the InVEST nutrient retention model in the United Kingdom. Sci. Total Environ. 610–611, 666–677. https://doi.org/10.1016/j.scitotenv.2017.08.092.
Robinson, D.A., Hockley, N., Dominati, E., Lebron, I., Scow, K.M., Reynolds, B., Emmett, B.A., Keith, A.M., de Jonge, L.W., Schjønning, P., Moldrup, P., Jones, S.B., Tuller, M., 2012. Natural capital, ecosystem services, and soil change: why soil science must embrace an ecosystems approach. Vadose Zone J. 11 (1) https://doi.org/10.2136/vzj2011.0051.
Rogers, K., Sacre, K., Goodenough, J., Doick, K.J., Treeconomics (Business enterprise), 2015. Valuing London's Urban Forest: Results of the London I-Tree Eco Project.
Ross, S., Evans, D., Webber, M., 2002. How LCA studies deal with uncertainty. Int. J. Life Cycle Assess. 7 (1), 47. https://doi.org/10.1007/BF02978909.
Rugani, B., Benetto, E., 2012. Improvements to emergy evaluations by using life cycle assessment. Environ. Sci. Technol. 46 (9), 4701–4712. https://doi.org/10.1021/es203440n.
Rugani, B., Maia de Souza, D., Weidema, B.P., Bare, J., Bakshi, B., Grann, B., Johnston, J. M., Pavan, A.L.R., Liu, X., Laurent, A., Verones, F., 2019. Towards integrating the ecosystem services cascade framework within the Life Cycle Assessment (LCA) cause-effect methodology. Sci. Total Environ. 690, 1284–1298. https://doi.org/10.1016/j.scitotenv.2019.07.023.
Saad, R., Margni, M., Koellner, T., Wittstock, B., Deschênes, L., 2011. Assessment of land use impacts on soil ecological functions: development of spatially differentiated characterization factors within a Canadian context. Int. J. Life Cycle Assess. 16 (3), 198–211. https://doi.org/10.1007/s11367-011-0258-x.
Salam, M.M.A., Mohsin, M., Rasheed, F., Ramzan, M., Zafar, Z., Pulkkinen, P., 2020. Assessment of European and hybrid aspen clones efficiency based on height growth and removal percentage of petroleum hydrocarbons—a field trial. Environ. Sci. Pollut. Control Ser. 27 (36), 45555–45567. https://doi.org/10.1007/s11356-020-10453-4.
Schirpke, U., Tasser, E., Ebner, M., Tappeiner, U., 2021. What can geotagged photographs tell us about cultural ecosystem services of lakes? Ecosyst. Serv. 51, 101354 https://doi.org/10.1016/j.ecoser.2021.101354.
Sharp, R., Tallis, H.T., Ricketts, T., Guerry, A.D., Wood, S.A., Chaplin-Kramer, R., Nelson, E., Ennaanay, D., Wolny, S., Olwero, N., 2018. InVEST 3.10 User's Guide; Collaborative Publication by the Natural Capital Project. Stanford University, the University of Minnesota, The Nature Conservancyy, and World Wildlife Fund.
Song, Y., Kirkwood, N., Maksimović, Č., Zheng, X., O'Connor, D., Jin, Y., Hou, D., 2019. Nature based solutions for contaminated land remediation and brownfield redevelopment in cities: a review. Sci. Total Environ. 663, 568–579. https://doi.org/10.1016/j.scitotenv.2019.01.347.
Stolte, J., Tesfai, M., Øygarden, L., Kværnø, S., Keizer, J., Verheijen, F., Panagos, P., Ballabio, C., Hessel, R. (Eds.), 2016. Soil Threats in Europe. Publications Office.
Tan, R.R., 2008. Using fuzzy numbers to propagate uncertainty in matrix-based LCI. Int. J. Life Cycle Assess. 13 (7), 585–592. https://doi.org/10.1007/s11367-008-0032-x.
Tang, Y.-T., Nathanail, C.P., 2012. Sticks and stones: the impact of the definitions of brownfield in policies on socio-economic sustainability. Sustainability 4 (5). https://doi.org/10.3390/su4050840. Article 5.
Tratalos, J., Fuller, R.A., Warren, P.H., Davies, R.G., Gaston, K.J., 2007. Urban form, biodiversity potential and ecosystem services. Landsc. Urban Plann. 83 (4), 308–317.
VanderWilde, C.P., Newell, J.P., 2021. Ecosystem services and life cycle assessment: a bibliometric review. Resour. Conserv. Recycl. 169, 105461 https://doi.org/10.1016/j.resconrec.2021.105461.
Vangronsveld, J., Weyens, N., Thijs, S., Dubin, D., Clemmens, M., Van Geert, K., Van Den Eeckhaut, M., Van den bossche, P., Gestel, G.V., Bruneel, N., Crauwels, L., Lemmens, C., 2019. Phytoremediation—Code of Good Practice. OVAM, p. 131. https://ovam.vlaanderen.be/.
Verones, F., Bare, J., Bulle, C., Frischknecht, R., Hauschild, M., Hellweg, S., Henderson, A., Jolliet, O., Laurent, A., Liao, X., Lindner, J.P., de Souza, D.M., Michelsen, O., Patouillard, L., Pfister, S., Posthuma, L., Prado, V., Ridoutt, B., Rosenbaum, R.K., et al., 2017. LCIA framework and cross-cutting issues guidance within the UNEP-SETAC Life Cycle Initiative. J. Clean. Prod. 161, 957–967. https://doi.org/10.1016/j.jclepro.2017.05.206.
Vigil, M., Marey-Pérez, M.F., Martinez Huerta, G., Álvarez Cabal, V., 2015. Is phytoremediation without biomass valorization sustainable? — Comparative LCA of landfilling vs. anaerobic co-digestion. Sci. Total Environ. 505, 844–850. https://doi.org/10.1016/j.scitotenv.2014.10.047.
Villa, F., Ceroni, M., Bagstad, K., Johnson, G., Krivov, S., 2009. ARIES (Artificial Intelligence for Ecosystem Services): A New Tool for Ecosystem Services Assessment, Planning, and Valuation, pp. 21–22.
Visentin, C., da Silva Trentin, A.W., Braun, A.B., Thomé, A., 2019. Application of life cycle assessment as a tool for evaluating the sustainability of contaminated sites remediation: a systematic and bibliographic analysis. Sci. Total Environ. 672, 893–905. https://doi.org/10.1016/j.scitotenv.2019.04.034.
Volchko, Y., Norrman, J., Bergknut, M., Rosén, L., Söderqvist, T., 2013. Incorporating the soil function concept into sustainability appraisal of remediation alternatives. J. Environ. Manag. 129, 367–376. https://doi.org/10.1016/j.jenvman.2013.07.025.
Volchko, Y., Berggren Kleja, D., Back, P.-E., Tiberg, C., Enell, A., Larsson, M., Jones, C.M., Taylor, A., Viketoft, M., Åberg, A., Dahlberg, A.-K., Weiss, J., Wiberg, K., Rosén, L., 2020. Assessing costs and benefits of improved soil quality management in remediation projects: a study of an urban site contaminated with PAH and metals. Sci. Total Environ. 707, 135582 https://doi.org/10.1016/j.scitotenv.2019.135582.
Ward, H., Wenz, L., Steckel, J.C., Minx, J.C., 2018. Truncation error estimates in process life cycle assessment using input-output analysis: truncation error estimates in life cycle assessment. J. Ind. Ecol. 22 (5), 1080–1091. https://doi.org/10.1111/jiec.12655.
Weidema, B.P., Wesnæs, M.S., 1996. Data quality management for life cycle inventories—an example of using data quality indicators. J. Clean. Prod. 4 (3–4), 167–174. https://doi.org/10.1016/S0959-6526(96)00043-1.
Weidema, B.P., Bauer, C., Hischier, R., Mutel, C., Nemecek, T., Reinhard, J., Vadenbo, C., Wernet, G., 2013. Overview and Methodology: Data Quality Guideline for the Ecoinvent Database Version 3.
Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., Weidema, B., 2016. The ecoinvent database version 3 (part I): overview and methodology. Int. J. Life Cycle Assess. 21 (9), 1218–1230. https://doi.org/10.1007/s11367-016-1087-8.
Witters, N., Mendelsohn, R.O., Van Slycken, S., Weyens, N., Schreurs, E., Meers, E., Tack, F., Carleer, R., Vangronsveld, J., 2012. Phytoremediation, a sustainable remediation technology? Conclusions from a case study. I: energy production and carbon dioxide abatement. Biomass Bioenergy 39, 454–469. https://doi.org/10.1016/j.biombioe.2011.08.016.
Woo, A., Lee, S., 2016. Illuminating the impacts of brownfield redevelopments on neighboring housing prices: case of Cuyahoga County, Ohio in the US. Environ. Plann.: Econ. Space 48 (6), 1107–1132. https://doi.org/10.1177/0308518X16636380.
Zalesny, R.S., Headlee, W.L., Gopalakrishnan, G., Bauer, E.O., Hall, R.B., Hazel, D.W., Isebrands, J.G., Licht, L.A., Negri, M.C., Nichols, E.G., Rockwood, D.L., Wiese, A.H., 2019. Ecosystem services of poplar at long-term phytoremediation sites in the Midwest and Southeast, United States. WIREs Energy Environ. 8 (6) https://doi.org/10.1002/wene.349.
Zhang, Y., Baral, A., Bakshi, B.R., 2010. Accounting for ecosystem services in life cycle assessment, Part II: toward an ecologically based LCA. Environ. Sci. Technol. 44 (7), 2624–2631. https://doi.org/10.1021/es900548a.
Zhang, J., Cao, X., Yao, Z., Lin, Q., Yan, B., Cui, X., He, Z., Yang, X., Wang, C.-H., Chen, G., 2021. Phytoremediation of Cd-contaminated farmland soil via various Sedum alfredii-oilseed rape cropping systems: efficiency comparison and cost-benefit analysis. J. Hazard Mater. 419 https://doi.org/10.1016/j.jhazmat.2021.126489. Scopus.
Zhong, Q., Zhang, L., Zhu, Y., Konijnendijk van den Bosch, C., Han, J., Zhang, G., Li, Y., 2020a. A conceptual framework for ex ante valuation of ecosystem services of brownfield greening from a systematic perspective. Ecosys. Health Sustain. 6 (1), 1743206 https://doi.org/10.1080/20964129.2020.1743206.
Zhong, Q., Zhang, L., Zhu, Y., Konijnendijk van den Bosch, C., Han, J., Zhang, G., Li, Y., 2020b. A conceptual framework for ex ante valuation of ecosystem services of brownfield greening from a systematic perspective. Ecosys. Health Sustain. 6 (1), 1743206 https://doi.org/10.1080/20964129.2020.1743206.