THESIS FOR THE DEGREE OF DOCTOR OF PHILOSOPHY

Opportunity Windows and Added Value of Gentle Remediation Options for Contaminated Land Management

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Cover:
GRO applications for sustainable and risk-based land management and providing wider values, also Figure 2-2 (Paul Drenning, 2024).

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ABSTRACT

Well-functioning, healthy soils are increasingly recognized as vital to human well-being, but soil contamination impairs the capacity of soils to perform their essential functions and provide humans with ecosystem services (ES). Contaminated land poses risks to human health and the environment, which must be managed, but also constitute an important and underutilized land and soil resource for providing ES in urban areas through phytomanagement with gentle remediation options (GRO) – nature-based solutions using plants, fungi, bacteria, and soil amendments to manage risks at contaminated sites while also improving soil functionality. The overall aim of this thesis is to explore the opportunity windows and added value of using GRO for contaminated land management and develop applied knowledge and methods for practitioners to support a broader use of GRO in practice. Five studies were carried out to achieve the overall aim and specific objectives, containing both conceptual and empirical work, with a field experiment performed at the Kolleberga tree nursery. Specific contributions from the studies include: considering GRO applications for sustainable remediation and development (Paper I); developing a risk management framework for GRO (Paper II); investigating the costs and benefits and social profitability of GRO compared to conventional alternatives (Paper III); estimating time requirements for phytoextraction (Paper IV); and evaluating the effects of GRO on soil health (Paper V). Results from the studies in this Ph.D.-thesis are considered within the unifying concept of opportunity windows to explore the wider application and added value of GRO for contaminated land management and are connected to the generic workflow for contaminated land management in Sweden to facilitate communication with stakeholders and inclusion in the decision-making process. The applied knowledge and methods developed in this Ph.D.-thesis support the wider application of GRO for contaminated land management by exploring the opportunity windows for feasible use of GRO and demonstrating their added value.

Keywords: Gentle remediation options (GRO); Phytomanagement; Ecosystem services; Soil functions; Soil health; Cost-benefit analysis; Sustainable and Risk-Based Land Management (SRBLM)

SAMMANFATTNING

Välfungerande, friska jordar erkänns alltmer som avgörande för människors välbefinnande men markföroreningar försämrar markens funktioner och förse människor med ekosystemtjänster (ES). Förorenade områden kan innebära risker för människors hälsa och miljön vilket måste hanteras, men de utgör också en viktig och underutnyttjad mark- och jordresurs. Genom så kallad fytomanagement med skonsamma efterbehandlingsmetoder (GRO) – naturbaserade lösningar som använder växter, svampar, bakterier och jordförbättringsmedel – kan risker hanteras samtidigt som markfunktionalitet förbättras och får ökad förmåga att tillhandahålla ES i urbana områden. Det övergripande syftet med den här avhandlingen är att utforska möjligheterna och mervärdet av att använda GRO för hantering av förorenad jord samt att utveckla kunskap och metoder som kan stödja en bredare användning av GRO i praktiken. Fem studier har genomförts och inkluderar både konceptuellt och empiriskt arbete, det senare genom ett fältförsök vid Kolleberga skogsplantskola. Specifika bidrag från dessa studier är: ett underlag för att tydliggöra mervärdet av tillämpning av GRO för hållbar efterbehandling samt stadsutveckling (Studie I); ett ramverk för att tydliggöra hur olika GRO-strategier kan användas för att hantera risker (Studie II); en undersökning av kostnader och nyttor samt samhällelig lönsamhet för GRO jämfört med konventionella saneringsalternativ (Studie III); modeller för att kunna uppskatta den tid som krävs för att efterbehandling genom fytoextraktion skall nå uppsatt mål (Studie IV); en metod för att utvärdera GROs effekter på jordhälsa (markens förmåga att leverera ekosystemtjänster, Studie V). Resultaten från studierna kopplas till ett generellt arbetsflöde för hantering av förorenad mark i Sverige för att tydliggöra hur dessa kan integreras i en beslutsprocess samt hur de kan underlätta kommunikation mellan olika aktörer och intressenter. Den tillämpade kunskap och de metoder som utvecklats inom ramen för denna doktorsavhandling stödjer en bredare tillämpning av GRO för hantering av förorenad mark genom att utforska möjlighetsfönstren för GRO och demonstrera mervärdet av dem.

Nyckelord: Skonsamma saneringsmetoder (GRO); Fytomanagement; Ekosystemtjänster; Markfunktioner; Jordhälsa; Kostnadsnyttoanalys; Hållbar och riskbaserad markförvaltning (SRBLM)

LIST OF PUBLICATIONS

This thesis contains the following publications appended to the thesis:

I. **Drenning, P.**, Norrman, J., Chowdhury, S., Rosén, L., Volchko, Y., Andersson-Sköld, Y. (2020). Enhancing ecosystem services at urban brownfield sites - What value does contaminated soil have in the built environment? IOP Conf. Ser. Earth Environ. Sci. 588. https://doi.org/10.1088/1755-1315/588/5/052008

II. **Drenning, P.**, Chowdhury, S., Volchko, Y., Rosén, L., Andersson-Sköld, Y., Norrman, J. (2022). A risk management framework for Gentle Remediation Options (GRO). Sci. Total Environ. 802. https://doi.org/10.1016/j.scitotenv.2021.149880

III. **Drenning, P.**, Volchko, Y., Ahrens, L., Rosén, L., Söderqvist, T., Norrman, J. (2023). Comparison of PFAS soil remediation alternatives at a civilian airport using cost-benefit analysis. Sci. Total Environ. 882, 163664. https://doi.org/10.1016/j.scitotenv.2023.163664

IV. **Drenning, P.**, Enell, A., Berggren Kleja, D., Volchko, Y., Norrman, J. Probabilistic models to estimate the time required for phytoextraction. (Revised manuscript preliminarily accepted for publication in *Environ. Sci. Pollut. Res.)*

V. **Drenning, P.**, Volchko, Y., Enell, A., Berggren Kleja, D., Larsson, M., Norrman, J. Evaluating the effects of gentle remediation options (GRO) on soil health. (Submitted to *Sci. Total Environ.*)

Division of work between authors

Paper I: The authors contributed as follows: Conceptualization: P.D., S.C., Y.V., J.N.; Methodology: P.D., S.C., Y.V., J.N.; Investigation: P.D.; Visualization: P.D., S.C., Y.V., J.N.; Writing—original draft preparation: P.D., S.C., Y.V., J.N.; Writing—review and editing: P.D., S.C., Y.V., L.R., Y.A-S., J.N.; Supervision: Y.V., L.R., Y.A-S., J.N.; Project administration: J.N.; Funding acquisition, Y.V., L.R., J.N. All authors have read and agreed to the published version of the manuscript.

Paper II: The authors contributed as follows: Conceptualization: P.D., J.N., S.C.; Methodology: P.D., J.N., S.C.; Investigation: P.D.; Writing—original draft preparation: P.D., J.N., S.C.; Writing—review and editing: P.D., J.N., S.C., L.R., Y.V., Y.A-S.; Supervision: Y.V., L.R., Y.A-S., J.N.; Project administration: J.N.; Funding acquisition, Y.V., L.R., J.N.

All authors have read and agreed to the published version of the manuscript.

Paper III: The authors contributed as follows: Conceptualization: P.D., Y.V., T.S., L.R., J.N.; Methodology: P.D., Y.V., T.S., L.R., J.N.; Validation: L.A.; Formal analysis: P.D., Y.V.; Investigation: P.D., Y.V.; Visualization: P.D., Y.V.; Writing—original draft preparation: P.D., Y.V., L.A., J.N.; Writing—review and editing: P.D., Y.V., L.A., T.S., L.R., J.N.; Supervision: Y.V., L.R., J.N.; Project administration: J.N.; Funding acquisition, Y.V., L.R., J.N. All authors have read and agreed to the published version of the manuscript.

Paper IV: All authors contributed to the study conception and design. Material preparation, data collection and analysis were performed by Paul Drenning. The first draft of the manuscript was written by Paul Drenning. Anja Enell, Dan Berggren Kleja, Yevheniya Volchko and Jenny Norrman reviewed and commented on previous versions of the manuscript. All authors read and approved the final manuscript.

Paper V: The authors contributed as follows: Conceptualization: P.D., Y.V., A.E., D.B.K., J.N.; Methodology: P.D.; Formal analysis: P.D.; Investigation: P.D., Y.V., A.E., D.B.K., M.L., J.N.; Writing—original draft preparation: P.D.; Writing—review and editing: P.D., Y.V., A.E., D.B.K., M.L., J.N.; Supervision: Y.V., J.N.; Project administration: Y.V., J.N.; Funding acquisition: P.D., Y.V., A.E., D.B.K., J.N.

Other work and publications not appended to the thesis:

Drenning, P. (2021a). Gentle Remediation Options (GRO) for Managing Risks and Providing Ecosystem Services at Contaminated Sites [Licentiate thesis, Technical report 2021:8]. Chalmers University of Technology, Department of Architecture and Civil Engineering. https://doi.org/10.13140/RG.2.2.32637.69604

Drenning, P. (2021b). Gentle Remediation Options (GRO): A Literature Review (Part 1/2). Chalmers University of Technology, Department of Architecture and Civil Engineering, Gothenburg, Sweden. https://doi.org/10.13140/RG.2.2.36086.11849

Drenning, P. (2021c). Soil Functions and Ecosystem Services: A Literature Review (Part 2/2). Chalmers University of Technology, Department of Architecture and Civil Engineering, Gothenburg, Sweden. https://doi.org/10.13140/RG.2.2.23922.63685

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Finally, I would like to say a big thank you to the family, friends and co-workers who kept me sane throughout this whole period. This has been a long road (and as they say in the Shire, "if you don't keep your feet, there's no knowing where you might be swept off to"), but I was able to make it this far thanks to the support I received from friends and family. Thank you to my family and friends (mellons) back in the USA and here in Gothenburg who make this place feel like home. And, of course, I am thankful every day for the love and support from my fiancée, Annija (and our best boy, Karums) and looking forward to many happy years together.

Gothenburg, March 2024
Paul Drenning

TABLE OF CONTENTS

1 INTRODUCTION

This chapter provides a brief background of the research, presents the research aim and the main objectives of this thesis, as well as the scope of work, followed by clarifying limitations.

1.1 Background

Well-functioning, healthy soils are increasingly recognized as vital to human well-being by supporting not only crop production for food security but also providing other essential ecosystem services (ES) such as water purification, carbon sequestration, nutrient cycling, and habitat for biodiversity (Adhikari and Hartemink, 2016; Bünemann et al., 2018; EEA, 2022; El Mujtar et al., 2019; Greiner et al., 2017; Kibblewhite et al., 2008). The recently issued *Proposal for a Directive on Soil Monitoring and Resilience* (EC, 2023) further emphasizes the importance of soils and aims to set a high-level policy and research agenda as well as a harmonized, legally-binding path forward to manage soils as a common resource and ensure that they are in healthy condition. The definition of 'soil health' can vary considerably, but the *Soil Directive* defines healthy soils as *soils that are in good chemical, biological and physical conditions and are able to continuously provide as many ecosystem services as possible* (EC, 2023). Approximately 60-70% of the soils in the EU are considered unhealthy due to degradation (Veerman et al., 2020), which reduces their capability for food production and other services essential for achieving environmental objectives such as the Sustainable Development Goals (SDGs) (FAO et al., 2020; Keesstra et al., 2016). Soil contamination (used here as synonymous to pollution) is one of the main causes of soil degradation and the effects of soil contamination on soil biota are manifold but can ultimately deteriorate the health of the soil ecosystem thus inhibiting the soil's ability to provide key ES for human well-being (Bünemann et al., 2018; FAO et al., 2020; FAO and UNEP, 2021; Orgiazzi et al., 2016; Turbé et al., 2010), as shown in Figure 1-1.

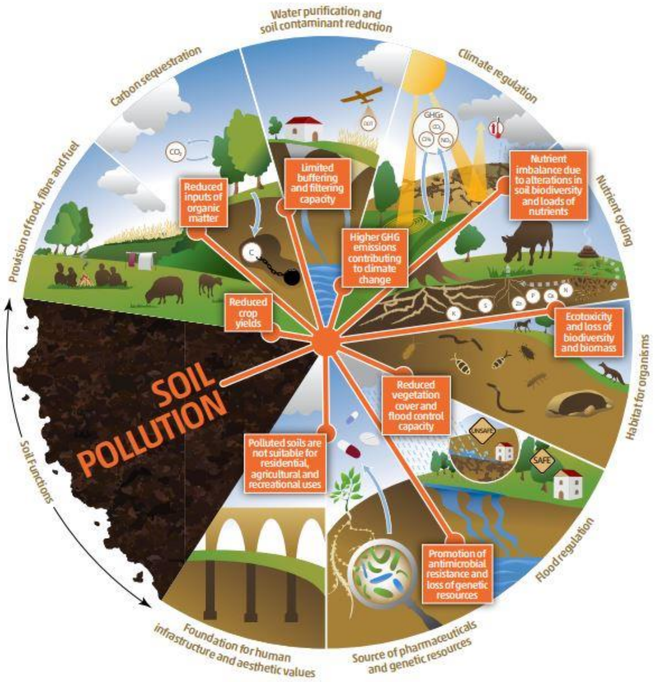

Figure 1-1. Soil pollution causes a cycle of degradation processes that leads to the reduction and ultimately to the loss of ecosystem services, from (FAO and UNEP, 2021).

In Europe, there are an estimated 2.8 million potentially contaminated sites resulting from human activity (Pérez and Eugenio, 2018), of which approximately 85,000 are in Sweden (SEPA, 2021a). Many of these are called 'brownfields', which are underused or derelict areas with real or perceived soil and groundwater contamination that require intervention to bring them back to beneficial use, and often face many barriers to redevelopment such as investment risks, ownership constraints, risk of future liability claims and public stigma (Ferber et al., 2006; ISO, 2017; Norrman et al., 2016; Vegter et al., 2002). The current paradigm for contaminated land management is based on managing risks through the use of conventional soil remediation techniques to, most often, remove the source of the soil contamination (Kuppusamy et al., 2016b, 2016a; Swartjes, 2011). Remediation by soil excavation and landfilling ('dig-and-dump') or ex-situ treatment is still the most commonly used method in practice (both in Sweden and abroad) since it is fast and effective for source removal, thus gaining regulatory approval, but is often the result of oversimplified, generic risk assessments and conservatively applied legislative guidelines (SEPA, 2018a; SGI, 2018; White arkitekter AB, 2021). Remediation, however, is not intrinsically sustainable (Bardos et al., 2020a; Cundy et al., 2016), and many remediation techniques can have considerable negative environmental impacts such as high carbon emissions, waste production and significant degradation or even destruction of the soil ecosystem and its essential functions (FAO et al., 2020; Gerhardt et al., 2017; Rosén et al., 2015; Swartjes, 2011; Volchko et al., 2014).

In general, contaminated soil has long been viewed as waste to be disposed of rather than as a valuable resource to be treated and reused (Gerhardt et al., 2017; Mench et al., 2010). Yet, contaminated land can still have good soil quality and retain the capacity to perform their functions and provide valuable ES, particularly for 'soft' end uses like green spaces (Bardos et al., 2016; FAO et al., 2020; Gerhardt et al., 2017; Swartjes, 2011; Volchko et al., 2014). There is thus a need to integrate additional soil parameters and a broader soil health assessment as part of contaminated land management to improve decision-making and prioritize areas for preservation or development according to their capability and condition (Blanchart et al., 2018; Volchko et al., 2014, 2019). Furthermore, there is a demand for new practices to facilitate sustainable remediation and brownfield regeneration, with fully 78% of surveyed practitioners in Sweden indicated a large need for alternative remediation methods to prevent 'over-remediation' by overuse of dig-and-dump (SEPA, 2018a; SGI, 2018). Indeed, a significant amount of brownfield land area remains derelict or underutilized due to rehabilitation being uneconomic or unsustainable using conventional methods (Bardos, 2014; Bardos et al., 2016).

To meet this need, nature-based solutions (NBS) are increasingly being recognized for their potential to manage different types of societal challenges (EEA, 2023; IUCN, 2020; McQuaid et al., 2022), including both improving soil health and managing contamination in soil and groundwater (Bardos et al., 2020a; Hou et al., 2023; Song et al., 2019). Gentle remediation options (GRO) are a subset of NBS that utilise plants, fungi, bacteria, and soil amendments to manage risks while at the same time improving (or at least not reducing) soil functionality and may be viable alternatives to conventional techniques, in particular for large areas and contaminated sites that pose low to medium risks to human health and the environment (Cundy et al., 2016; GREENLAND, 2014a; Jones et al., 2014; OVAM, 2019).

Despite the well-recognized and pressing need for alternative remediation methods, there has been slow progress in both remediating contaminated sites and adopting new practices in Sweden (SEPA, 2018a; SGI, 2018). The selection of non-conventional remediation options has often been constrained by the lack of available information regarding their performance under different site conditions and other uncertainties regarding effectiveness (Scott and Nathanail, 2004). As previously noted in Drenning (2021a), and in other studies interviewing experts in Sweden (Berghel et al., 2021; White arkitekter AB, 2021), there is a growing interest to consider GRO, ES, and NBS in contaminated land management. Nevertheless, widespread adoption is still lacking due to perceived and actual limitations (e.g., time requirements), uncertainties and a general lack of knowledge and awareness surrounding GRO. Indeed, there are many aspects aside from strict technical performance that can promote or inhibit the use of GRO and there is a clear need to improve accessibility, address knowledge gaps and further spread knowledge about GRO. Some of these broader aspects are here addressed within the unifying concept of **opportunity windows**[1] to provide accessible scientific information to stakeholders regarding the wider application and **added value** of GRO for contaminated land management.

1.2 Aim and objectives

The overall aim of this thesis is *to explore the opportunity windows and added value of using GRO for contaminated land management and develop applied knowledge and methods for practitioners to support a broader use of GRO in practice.*

To achieve the overall aim, this thesis has the following specific objectives:

i. To explore the wider values of contaminated land and the potential of GRO as an alternative remediation strategy to rehabilitate soils in urban environments.
ii. To investigate how different GRO strategies can be used for risk management by identifying and finding support for relevant risk mitigation mechanisms and their associated risk reduction times.
iii. To investigate and demonstrate the potential costs and benefits in society of GRO compared to conventional remediation technologies.
iv. To develop simplified probabilistic models to estimate and clarify expectations regarding the time requirements for phytoextraction.
v. To evaluate and demonstrate the potential impacts, positive or negative, of GRO on soil functioning and the delivery of ecosystem services.
vi. To develop a framework to effectively communicate about risk management using GRO and identifying feasible alternatives.

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[1] 'Opportunity windows' is derived from the concept of remediation-option 'operating windows', originally introduced by Scott and Nathanail (2004) for the context of remediation. The concept was developed further and adapted within both the FP7 HOMBRE and Greenland projects as 'high-level operating windows (HLOW)' together with the 'Brownfields Opportunity Matrix (BOM)' to aggregate key information for use more broadly as decision-support guidance for brownfield soft re-use and 'detailed operating windows (DOW)' to identify the optimal site or soil conditions for applying GRO, respectively (GREENLAND, 2014a; Jones et al., 2014). Opportunity windows as used here are comparable to HLOW as developed in HOMBRE.

1.3 Scope of work

The work carried out in the Ph.D.-project was both **conceptual** and **empirical**, with a field experiment initiated about half-way into the PhD project, beginning in June 2021, to test GRO in real-world conditions at the former Kolleberga tree nursery in Sweden. The materials and methods differ between the studies to accomplish the above-listed specific objectives (the non-sequential relationship between project aspects is shown in Figure 1-2) and has resulted in five distinct publications that are appended to the thesis:

Paper I: Enhancing ecosystem services at urban brownfield sites – what value does contaminated soil have in the built environment?

Paper II: A risk management framework for Gentle Remediation Options (GRO)

Paper III: Comparison of PFAS soil remediation alternatives at a civilian airport using cost-benefit analysis

Paper IV: Probabilistic models to estimate the time required for phytoextraction

Paper V: Evaluating the effects of Gentle Remediation Options (GRO) on soil health

This thesis contains material that has been published previously in the author's licentiate thesis: Drenning, P. (2021a). Gentle Remediation Options (GRO) for Managing Risks and Providing Ecosystem Services at Contaminated Sites [Licentiate thesis, Technical report 2021:8]. Chalmers University of Technology, Department of Architecture and Civil Engineering. https://doi.org/10.13140/RG.2.2.32637.69604

To achieve the aim and fulfil the specific objectives of this thesis, a multi-disciplinary approach was required. Identifying and exploring the intersection between related (yet often disconnected) fields as part of a ‘thematic exploration’ – including contamination and associated risks, remediation of contaminated sites (both gentle and conventional), sustainability in remediation, soil science (requiring a study of soil biota, soil functioning and soil quality assessment) and associated fields – forms the groundwork for this Ph.D.-thesis. To establish the context within which this work has been carried out, the thesis begins with a theoretical background (Chapter 2) to briefly present the relevant topics.

The methodology section (Chapter 3) describes the process to achieve the aim of this Ph.D.-thesis and lists the main methods followed to fulfil the research objectives. The field experiment at Kolleberga tree nursery, which was essential for fulfil specific objectives iv and v, is described in this section. A brief case study description for Paper III is also provided in this section.

Results (Chapter 0) are provided in the subsequent section and include key results in brief relating to Papers II-IV as they pertain to the overall aim of this thesis.

Following the results, a discussion (Chapter 5) of the work carried out is provided in the next section to discuss broader implications, including how it relates GRO opportunity windows and added values and connections to the generic workflow for contaminated land management in Sweden. Then, main conclusions (Chapter 6) that can be drawn from this Ph.D.-thesis are briefly summarized. In the final section, an outline of ongoing and future work (Chapter 7) is given including reflections about the relevance of this Ph.D.-thesis for practitioners and what still remains to investigate and develop.

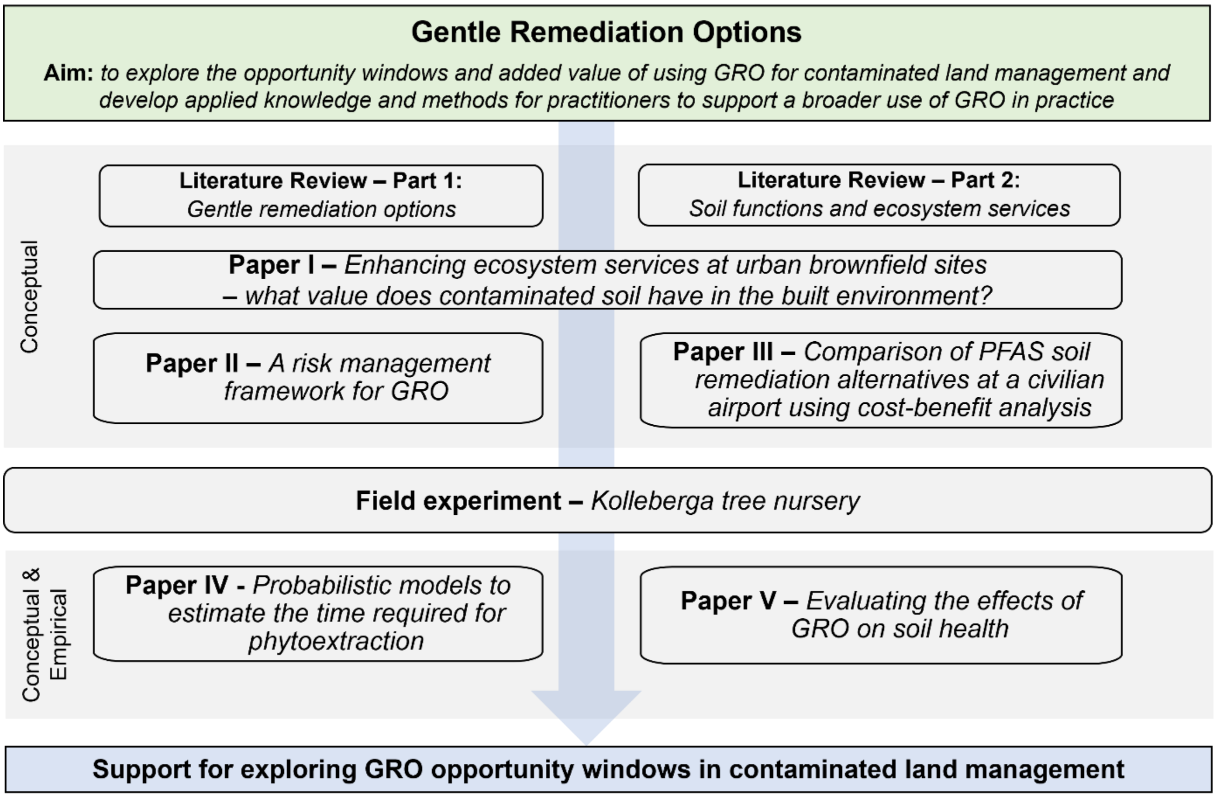

Figure 1-2. Scope of work showing the conceptual and empirical designation of publications included in the thesis and progress towards achieving the overall aim.

1.4 Limitations

Some limitations of this Ph.D.-thesis are as follows:

- The field work for the field experiment was carried out by the authors and collaborators. However, the laboratory analyses were performed by external labs and not overseen by the author so there may be errors or inconsistencies in analyses that have not been taken into account. Such variance was minimized to the greatest extent possible when performing the field work and designing the field experiment.
- As it is multidisciplinary research, the focus has been put in linking different fields of interest rather than an in-depth exploration of each topic. Thus, the thesis provides a necessarily limited investigation into each of these, and some important information or context may be missing.

- GRO, phytomanagement, ecosystem services, soil science and brownfield redevelopment and regeneration are concepts with a solid scientific foundation but are developing quickly. Some important information may have been missed in writing this thesis and new material is being published regularly that may be concurrent with the writing of this thesis and not included here.
- Generalizing the essential knowledge pertaining to GRO that fed into the development of the different applied tools and methods developed in this Ph.D.-thesis inevitably led to some oversimplifications, some of which are noted in the Discussion and Conclusions section. It is acknowledged that actual field application of GRO is a site-specific process that requires a more detailed risk assessment and in-depth knowledge of the site conditions to effectively manage the exposure risks to receptors at a contaminated site using GRO.

2 THEORETICAL BACKGROUND

This chapter briefly presents different concepts related to the research, connects them to support the proposal herein, and builds on the findings to elaborate upon the scope of the research.

2.1 Contaminated land management

The generic workflow for contaminated land management (CLM) is generally a stepwise process from goal formulation, site investigations, risk assessment and options appraisal to implementation and monitoring of a remediation option for risk management (SEPA, 2021b; Swartjes, 2011). Risk assessment at contaminated sites is based on the source-pathway-receptor (S-P-R) concept, also referred to as 'contaminant linkages' (UK Environment Agency, 2021), Figure 2-1. In this risk assessment framework, the mere presence of a hazard (e.g., soil contamination) does not necessarily mean that it constitutes a risk (Swartjes, 2011). For a risk to occur, there must be a source (hazard), a receptor (something that could be adversely affected) and an exposure pathway linking the source to the receptor (Bardos et al., 2020a, 2020b; Cundy et al., 2016; Swartjes, 2011). A receptor might be a human, an ecologically sensitive site, species or ecosystem, surface or groundwater resource, archaeological resource, property such as a building, crops or fisheries (Swartjes, 2011). Receptors can potentially be exposed to soil contaminants through several exposure pathways, and if the risk assessment has determined a viable exposure risk, an (eco)toxicological assessment can then be carried out to determine what adverse effects may arise depending on the estimated dosage (Swartjes, 2011).

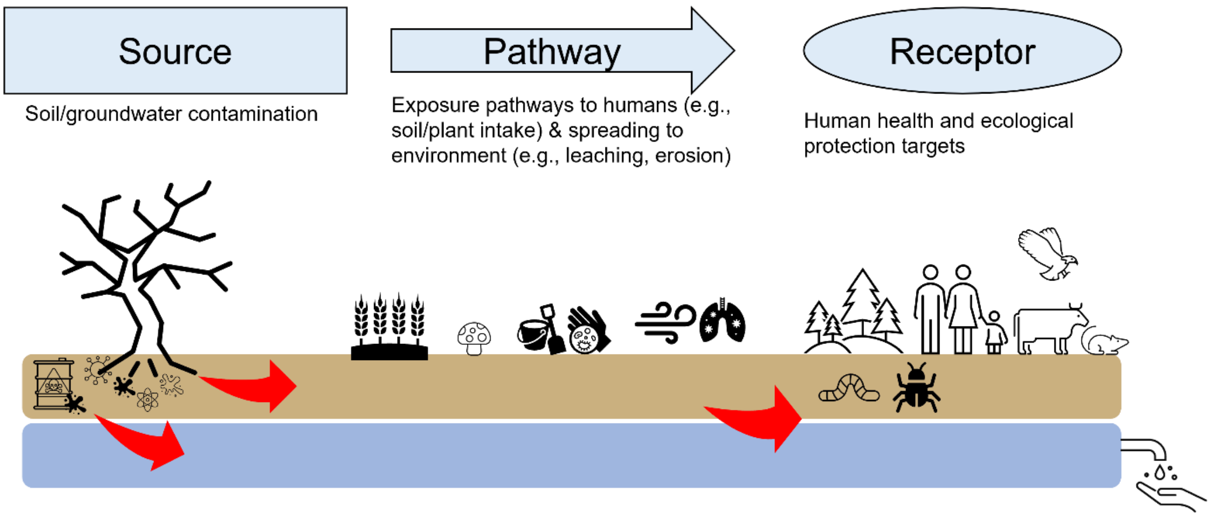

Figure 2-1. Source-pathway-receptor model for risk assessment at contaminated sites.

Human health is always a protection target of vital importance when assessing risks at contaminated sites. In the Swedish EPA's (SEPA) soil guideline value model (SEPA, 2021b, 2016, 2009), the following main exposure pathways are accounted for in human health risk assessment: ingestion of contaminated soil, ingestion of plants (grown on contaminated sites that may have elevated concentrations), inhalation of dust, inhalation of vapour, dermal contact and intake of drinking water (if taken from a well on site). Regarding the environment, the main protection targets accounted for in Sweden are the soil ecosystem, groundwater as a resource and surface water ecosystems. These receptors can be exposed via spreading of contaminants in free phase, porewater, etc. which is largely dependent on the specific contaminant's bioavailability (i.e., the readily available fraction of a contaminant that can cross cell membranes to enter the organism) and solubility, which in turn is heavily influenced by site-specific conditions (Naidu et al., 2015; SEPA, 2016; Swartjes, 2011). Fully understanding the actual risks posed by contaminants to sensitive receptors requires a more complex, site-specific risk assessment wherein a critical factor is the bioavailability of contaminants (Naidu et al., 2015; Swartjes, 2011). Increasingly, a risk-based approach to monitoring soil degradation is recommended by accounting for soil health and/or valuable ecosystem goods and services as protection targets (Bardos et al., 2020a, 2020b; EEA, 2022).

Risk management interventions to mitigate/reduce the risks can take place at any point in the S-P-R chain as long as it breaks the contaminant linkage, which could involve removing the source, disrupting the pathway or managing the receptor to reduce the risk of unacceptable harm (Bardos et al., 2020a, 2020b; Cundy et al., 2016; Swartjes, 2011). A variety of remediation options are available that target different points across the various contaminant linkages. Conventional soil remediation techniques are those that utilise physical, chemical, biological or a combination of methods to, most often, address the source of contamination ex-situ (entailing soil excavation and subsequent treatment on- or off-site via soil washing, thermal treatment, etc.) or in-situ to degrade, transform, extract or stabilise (in)organic contaminants at the site or utilise barriers like clay liners and permeable reactive barriers to isolate the site from its surroundings (Kuppusamy et al., 2016b, 2016a; Swartjes, 2011). The current international consensus is that land contamination decision making should be made on the basis of risks to human health and the wider environment, according to S-P-R linkages, a paradigm often referred to as risk-based land management (RBLM) (Bardos et al., 2011b, 2020b, 2018; Vegter et al., 2002). RBLM provides an objective way to link actions to the prevention of harm, a rationale for how to intervene (i.e., managing contaminant linkages), and a rationale to prioritise the dispensation of limited resources at sites according to risk evaluation (Bardos et al., 2011b, 2020a, 2020b, 2018; Common Forum and NICOLE, 2013; Swartjes, 2011; Vegter et al., 2002).

RBLM is predicated on the reduction of risks to human health and the environment to the degree necessary to ensure a safe, beneficial reuse of site (i.e., fitness for use) while protecting the environment over the long-term (Bardos et al., 2020a, 2018; ISO, 2017; Swartjes, 2011). Sustainable remediation and sustainable brownfield regeneration can be seen as overlapping domains in the wider context of sustainable land development (ISO, 2017; Rizzo et al., 2016). Therefore, risk management should also meet sustainable development principles as a core project objective, and this integrated approach constitutes sustainable risk-based land management (SRBLM) (Bardos et al., 2020a, 2020b; Common Forum and NICOLE, 2013; Rizzo et al., 2016). SRBLM has emerged as the optimal approach for balanced contaminated land decision-making, which combines a risk-based framework for determining when the risk (or potential risk) is unacceptable and where/when action is necessary with ensuring that sustainability is a part of deciding how such unacceptable risks are to be managed (Bardos et al., 2020b).

2.2 Gentle remediation options

This section will provide a brief overview of gentle remediation options (GRO). For more information and compilations of field studies the reader is referred to (Drenning, 2021b) and the summary table compiling GRO mechanisms, contaminants and media for which they are applicable and possible plant species for 'situational risk management' in Drenning (2021a).

Gentle remediation options (GRO) are nature-based solutions that can be applied to manage risks at brownfields and provide or maintain vital ecosystem services through revegetation (Bardos et al., 2020a, 2016; Cundy et al., 2016; Song et al., 2019). GRO are defined as *risk management strategies or technologies that result in a net gain (or at least no gross reduction) in soil function as well as achieving effective risk management* (Cundy et al. 2016). GRO is an umbrella term covering a set of remediation technologies based upon the use of plant (phyto-), fungi (myco-), and/or bacteria-based (bio-) methods with or without the use of chemical additives or soil amendments, Table 2-1 (Cundy et al., 2016; GREENLAND, 2014a). Soil invertebrates such as earthworms (vermi-) have also been shown to improve decontamination of organic (e.g. pesticides) and inorganic contaminants (metals) by plants and microorganisms (FAO et al., 2020; Lacalle et al., 2020; Orgiazzi et al., 2016; Rodriguez-Campos et al., 2014; Turbé et al., 2010), and could also be considered a GRO.

Table 2-1. List of definitions for GROs used to remediate soils contaminated by either trace elements or mixed contamination, adapted from (Bardos et al., 2020a; Cundy et al., 2016; GREENLAND, 2014a; OVAM, 2019).

GRO	Definition
Phytoextraction	Process in which plants and their associated microorganisms absorb contaminants and fix them in above-ground plant tissue that can then be removed from the site during harvesting.
Phytodegradation/ phytotransformation	The use of plants (and associated microorganisms like endophytic bacteria) to uptake, store and degrade contaminants.
Rhizodegradation	The use of plant enzymes and rhizospheric (in root zone) microorganisms to degrade organic contaminants.
Phytostabilisation	Reduction in the bioavailability and mobility of contaminants by immobilisation in root systems and/or living dead biomass in the rhizosphere soil.
Phytovolatilisation	The use of plants to remove contaminants from the growth matrix, transform them to less toxic forms and disperse them (or their degradation products) into the atmosphere.
In-situ immobilisation	Reduction in the bioavailability of contaminants by immobilisation or binding them to the soil matrix through the incorporation into the soil of organic or inorganic compounds to prevent excessive uptake and transfer into the food chain.
Phytoexclusion	The implementation of a stable vegetation cover using excluder plants which do not accumulate contaminants in the harvestable biomass, often combined with in-situ immobilisation.
Rhizofiltration	The removal of contaminants from aqueous sources by plant roots and associated microorganisms.
Phytohydraulics	Process in which plants and their microorganisms take up and evaporate water and thereby influence the groundwater level, the direction and velocity of the groundwater flow.
Bioremediation	Generic term applied to a range of remediation and risk management technologies which utilise soil microorganisms to degrade, stabilise or reduce the bioavailability of contaminants.
Mycoremediation	A form of bioremediation in which fungi-based methods are used to degrade, extract, stabilise or reduce the bioavailability or contaminants.
Vermiremediation	A remediation technique which utilises earthworms to remove or stabilise soil contaminants.

GRO have emerged as alternatives to conventional physicochemical methods, which may be unsuitable or unnecessary in many cases, and are multifunctional strategies for: i) effective risk management, ii) a reduction of soil ecotoxicity, iii) the legal and ethically required reduction of contamination risks for both human health and the environment; and, concurrently, a recovery of iv) soil health and v) associated ecosystem services (Burges et al., 2018; Cundy et al., 2016; GREENLAND, 2014a; Lacalle et al., 2020). Substantial economic (e.g., profitable biomass generation), socio-cultural (e.g., leisure and recreation), and environmental (e.g., ecosystem services and restoration of plant and microbial and animal communities) wider values are also possible through GRO application when intelligently applied (Bardos et al., 2016; Conesa et al., 2012; Cundy et al., 2013, 2016; Evangelou et al., 2012; GREENLAND, 2014a).

In terms of risk management, GRO are primarily applied on contaminated soils to reduce contaminant transfer to local receptors by gradually removing the bioavailable pool of inorganic contaminants (*phytoextraction*), removing or degrading organic contaminants (*phyto- and rhizodegradation*), filtering contaminants from surface water and waste water (*rhizofiltration*) or groundwater (*phytohydraulics*), and stabilising or immobilising contaminants in the soil matrix (*phytostabilisation, in-situ immobilisation*) often in combination with vegetation cover using excluder plants (*phytoexclusion*) (Table 2-1). If well-designed, GRO can be customised to provide risk management along S-P-R contaminant linkages via i) gradual removal or immobilisation (i.e. reducing bioavailability/solubility) of the contaminant source, ii) managing the flux of contaminants along exposure pathways and breaking connections to receptors through containment and stabilisation, and iii) managing the receptor's access to the contaminated medium thus preventing exposure (Bardos et al., 2020a; Cundy et al., 2016; GREENLAND, 2014a), Figure 2-2.

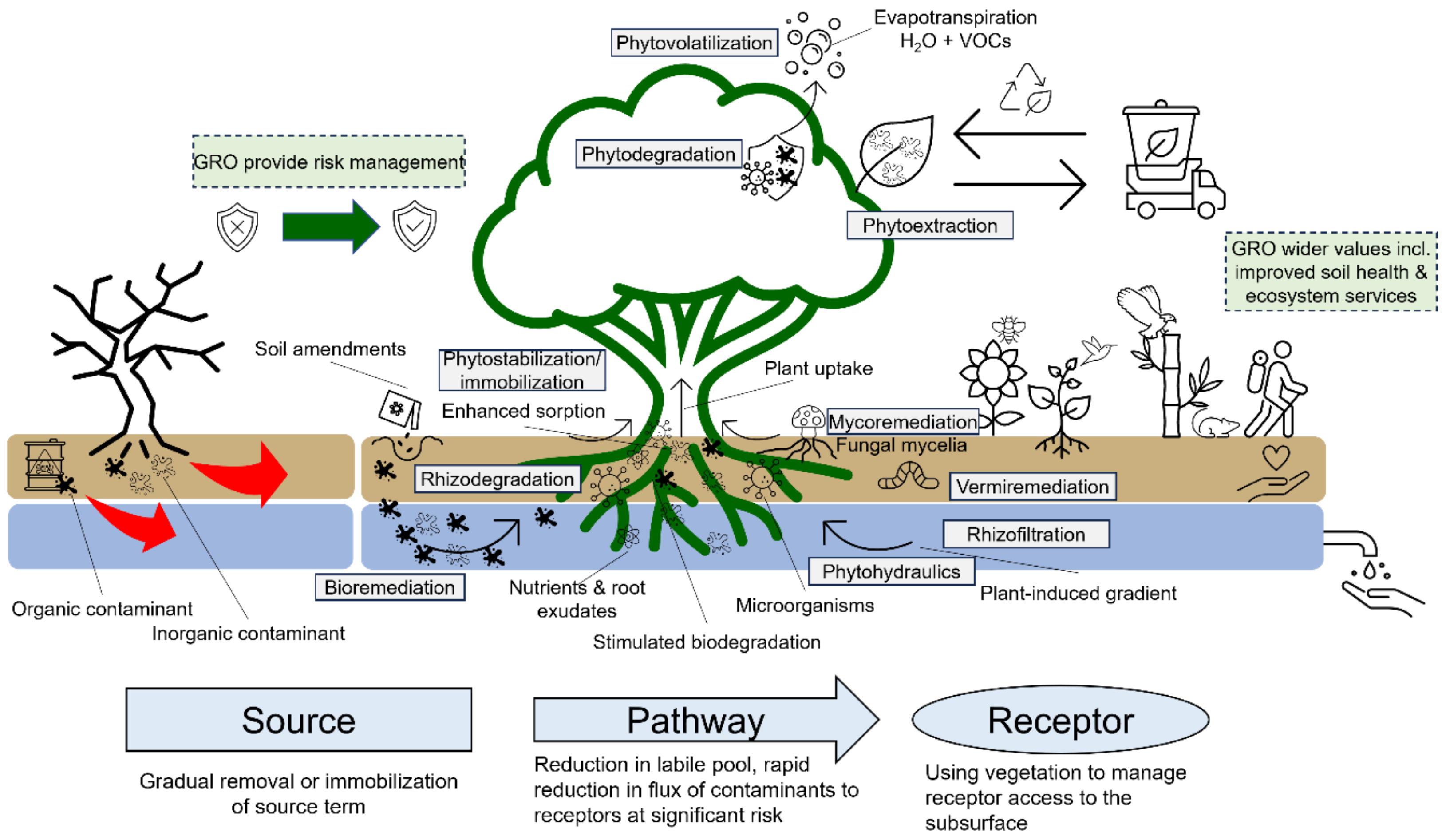

Figure 2-2. GRO applications for sustainable and risk-based land management and providing wider values.

While GRO may not be well-suited to highly contaminated sites, hotspots or point source terms such as buried tanks or oil spills, they are particularly suitable for large areas and contaminated sites that pose low to medium risks to human health and the environment (Andersson-Sköld et al., 2014; Cundy et al., 2016; Enell et al., 2016; GREENLAND, 2014a). GRO are useful as 'primary prevention strategies' in various applications to reduce or eliminate human (and non-human) exposure to contaminants (Henry et al., 2013). GRO can also be used for source removal of inorganic and organic contaminants though the timeframe for remediation can differ significantly between the contaminants and the mechanisms involved (Kennen and Kirkwood, 2015; OVAM, 2019). An important note is that the estimated time for full source removal (e.g., via extraction or degradation) can vary significantly depending on if total or bioavailable concentrations are used to measure success.

To improve the effectiveness of GRO, phytoremediation can be enhanced (or 'aided' or 'microorganism-assisted') through enriching the microbes in the rhizosphere or within the plant itself by bioaugmentation (i.e., introducing external species to the site that may be better suited for degrading specific contaminants) or biostimulation (i.e., enhancing the already existing microbes by the use of soil amendments) that can promote plant growth and tolerance and increase degradation and extraction rates (Mench et al., 2010; OVAM, 2019; Thijs et al., 2017, 2016; Vangronsveld et al., 2009). Soil amendments are frequently used to enhance the effectiveness of phytoremediation by reducing (or increasing) the bioavailability of metals in soil and uptake in plants as well improve soil quality, particularly when using organic amendments, to enable the establishment of vegetation in poor soils by, for example, improving soil physical properties like bulk density and pore structure, improving water infiltration and holding capacity, improving soil fertility by adding essential micro- and macronutrients, balancing soil pH, re-establishing microbial communities and increasing soil organic matter (Burges et al., 2018; Epelde et al., 2009b; Gómez-Sagasti et al., 2018; GREENLAND, 2014b; Kidd et al., 2015; Kumpiene et al., 2019; Mench et al., 2010; Vangronsveld et al., 2009).

A promising new direction in the application of GRO is phytomanagement; commonly defined as "*the long-term combination of profitable crop production with gentle remediation options (GRO) leading gradually to the reduction of contaminant linkages due to metal(loid) excess and restoration of ecosystem services*" (Cundy et al., 2016; GREENLAND, 2014a, 2014b; Robinson et al., 2009). Best practices for successful phytomanagement have been developed and optimised in large-scale European projects (e.g., GREENLAND and PhytoSUDOE); including through i) enhancing standard phytoremediation strategies with soil amendments and/or bacterial inoculates and mycorrhizal fungi, ii) creating tree plantations based on short-rotation coppicing of woody plants such as poplar and willow, iii) using high-biomass annual or perennial herbaceous species (e.g., rapeseed, sunflower, tobacco, bioenergy grasses, maize, etc.), and iv) applying best practice agronomic techniques like crop rotations, intercropping with legumes, agroforestry, cover crops, etc. to improve phytoremediation effectiveness (Garbisu et al., 2019; Gómez-Sagasti et al., 2018; GREENLAND, 2014a, 2014b; Kidd et al., 2015; Mench et al., 2019; Moreira et al., 2021, 2019).

2.3 Biochar

This section will briefly describe biochar and its uses to manage contaminated soil and improve soil health, see e.g., (Gul et al., 2015; Guo et al., 2020; Lehmann et al., 2011; Verheijen et al., 2009) for more information.

Biochar is the carbon-rich solid product remaining after residual biomass (e.g., wood, crop residues, manures) is thermally degraded under oxygen-limited conditions (pyrolysis) at temperatures greater than 350°C, and is used as a soil amendment (Lehmann et al., 2011; Verheijen et al., 2009). The resulting charcoal-like material often has a very high organic carbon content, neutral to basic pH, high nutrient content (e.g., P, K, Ca, etc.), high porosity and surface area, and other properties that can greatly alter the soil environment (Kookana et al., 2011; Verheijen et al., 2009). However, biochar properties vary widely and profoundly (Lehmann et al., 2011), and an important caveat is that the pyrolysis temperature and type of biomass (feedstock) greatly influence the resulting physico-chemical properties of the biochar and for which applications it is most suitable as well as potential toxins (e.g., PAHs, metals) that may remain in the material (Kookana et al., 2011; Lehmann et al., 2011; Verheijen et al., 2009). Biochar effects also vary depending on the type of soil and are typically greater in less fertile, marginal soils with low organic matter content and pH and sandy soils compared to soils with a higher clay content (Bekchanova et al., 2021; Enell et al., 2020; Tang et al., 2013). In general, biochar amendment to soils is considered a promising strategy for long-term carbon storage in soils since much of the carbon is very stable with a long residence time, meaning that it can sustainably sequester carbon in the soil without being degraded for hundreds to thousands of years (Gul et al., 2015; Kuzyakov et al., 2009; Mašek et al., 2013; Spokas, 2010; Verheijen et al., 2009). Biochar is also considered to have additional multifunctional values beyond carbon storage such as improving soil health, nutrient and microbial carrier, immobilising agent for remediation of toxic metals and organic contaminants in soil and water, and many others (Bolan et al., 2021). In-situ soil remediation with biochar amendment is considered to be an environmentally sustainable alternative remediation technique to conventional methods (Guo, 2020), which has recently been investigated in Sweden (Enell et al., 2020; Papageorgiou et al., 2021).

Indeed, there is a growing body of research showing the successful application of biochar to manage both inorganic and organic contaminants in soils (Beesley et al., 2011; Enell et al., 2020; Guo et al., 2020; Hou, 2021; Lin et al., 2022; Mierzwa-Hersztek et al., 2018; O'Connor et al., 2018; Tang et al., 2013). Biochar amendment does not remove contaminants from soil but instead transforms the water-soluble and bioavailable fractions of contaminants into immobilized forms, thus reducing the bioavailability and the ecotoxicity of the toxic elements, which is achieved primarily through elevating the soil pH, introducing carbonates and phosphates, and enhancing surface sorption (Guo et al., 2020). The mechanisms by which biochar stabilizes contaminants differs between inorganic and organic contaminants (Figure 2-3).

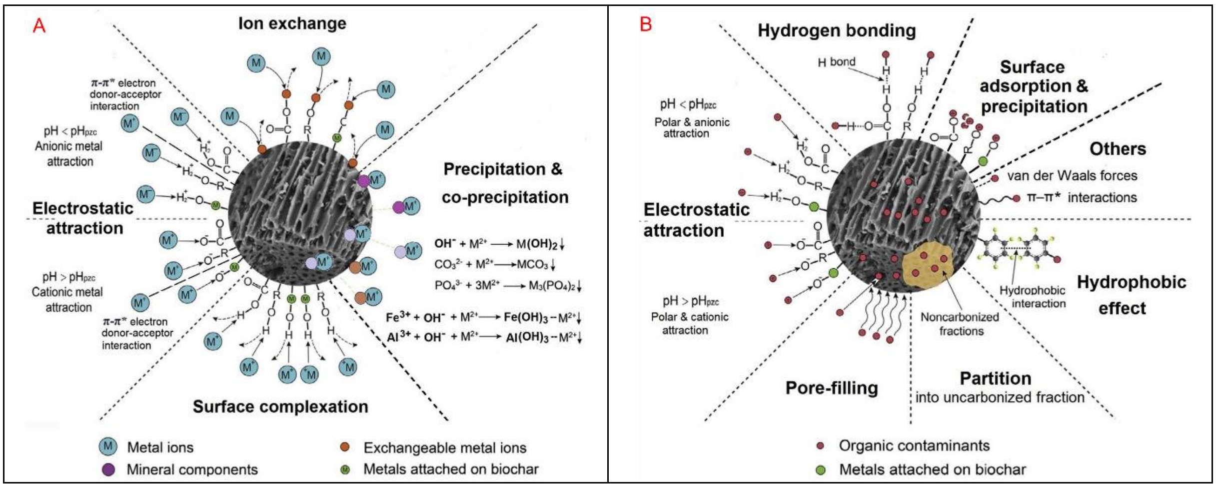

Figure 2-3. Major mechanisms by which biochar stabilizes inorganic (A) and organic (B) contaminants in soil, images modified from (Guo et al., 2020).

For inorganic contaminants such as metal(loid)s, biochar stabilizes cationic metals through enhanced sorption (electrostatic attraction, ion-exchange-based surface adsorption, surface complexation) and precipitation by soil pH elevation and ash addition of carbonates and phosphates (Beesley et al., 2011; Guo et al., 2020; Tang et al., 2013; Wang and Hou, 2024). However, for anionic metals, such as isomers of Cr, As and Sb, biochar amendment can actually increase their mobility in soil due to pH elevation (Beesley et al., 2014; Guo et al., 2020), which could increase the risk of leaching but may also facilitate phytoextraction-based remediation of these anions (Guo et al., 2020; Wang and Hou, 2024). For organic contaminants, biochar promotes stabilization through surface adsorption of non-polar organic compounds (via pore filling, partition and hydrophobic effect) and polar organic compounds (via hydrogen bonding, electrostatic attraction, specific surface interaction, and surface precipitation or chemisorption) (Guo et al., 2020; Tang et al., 2013), and may also promote biological degradation by improving the soil environment and enhancing microbial activity (Gregory et al., 2015; Guo et al., 2020; Lin et al., 2022; Zheng et al., 2022). In general, adsorption effectiveness varies between biochars and depends on the surface properties of the biochar, application rates, type of soil, mixing in soil and different types of biochar may be better suited to stabilizing different types of contaminants (Guo et al., 2020). For example, biochars produced at higher pyrolysis temperatures often have higher surface area and carbonized fraction, which can lead to high sorption capacity, and properties of surface area, porosity and the amount of functional groups can differ depending on the feedstock (Guo et al., 2020; Tang et al., 2013). According to (Guo et al., 2020), manure-derived biochars at lower pyrolysis temperature were more efficient than plant residue-derived biochars in stabilizing soil heavy metals while plant-residue-derived biochars at higher pyrolysis temperature mostly outperformed manure-derived biochars in adsorbing organic contaminants. Also, due to aging effects, biochar effectiveness may fade over time but this is debatable and varies considerably between studies and few long-term studies of field applications have been conducted to investigate this issue (Guo et al., 2020; O'Connor et al., 2018; Wang et al., 2020).

Biochar as an agricultural soil amendment is also well-demonstrated to have positive effects on many different soil properties, processes and functions relating to soil health and crop production (Agegnehu et al., 2017; Guo, 2020; He et al., 2021; Kookana et al., 2011; Kuppusamy et al., 2016c; Lehmann et al., 2011; Schröder et al., 2018; Verheijen et al., 2009). Physical and chemical soil properties are widely reported to improve with the addition of biochar such as an increase in water retention, porosity, aeration and aggregate stability with the addition of biochar (Blanco-Canqui, 2021; Gul et al., 2015; He et al., 2021; Hou, 2021; Hou et al., 2023; Zhang et al., 2021). However, biochar can also immobilize plant-available forms of essential nutrients such as nitrogen which could potentially inhibit biomass production (El-Naggar et al., 2019; Enell et al., 2020; Kookana et al., 2011; Zhao et al., 2023), but could be effectively mitigated by applying fertilizer or compost together with biochar (Beesley et al., 2011).

Biochar effects on soil biota can vary considerably (Lehmann et al., 2011). In general, microbial abundance should increase with the addition of biochar due to e.g., the porous, highly sorbent biochar providing a habitat and refuge for microorganisms from grazers or predators, addition of small pool of labile C providing a boost in biomass production, promoting the growth of mycorrhizal fungi, and better growth conditions due to biochar improving soil physico-chemical properties such as pH, nutrient and C availability, water retention, aeration, and reduced contaminant toxicity (Beesley et al., 2011; Domene et al., 2014; Gul et al., 2015; Jeffery et al., 2022; Lehmann et al., 2011; Verheijen et al., 2009; Zhou et al., 2017). Microbial activity may also increase in the short-term due to a positive 'priming' effect with the addition of a small labile C pool (Zimmerman et al., 2011). A general decrease in C mineralization, due to a less stressed microbial community converting organic C to biomass more efficiently without producing as many enzymes and releasing as much CO_2, would be expected over the long term especially with biochars produced at higher pyrolysis temperatures that are more stable and not as readily mineralizable by microbes (Ameloot et al., 2013; Bruun et al., 2008; Budai et al., 2016; Domene et al., 2015, 2014; Lehmann et al., 2011; Zhou et al., 2017; Zimmerman et al., 2011). These effects appear to be strongest for low ligno-cellulosic biochars derived from crop residue and manure, biochars slow pyrolyzed at high temperature over 500°C, and soils with higher clay content and neutral pH (Gul et al., 2015; Zhou et al., 2017). Biochar can also cause shifts in soil microbial community dynamics and composition (Brtnicky et al., 2021; Gul et al., 2015; Zhao et al., 2023; Zheng et al., 2016).

The effects of biochar amendment on soil fauna (e.g., earthworms, Collembola, enchytraeids, nematodes) are more mixed and can even be negative according to numerous studies (Brtnicky et al., 2021; Marks et al., 2016; Prodana et al., 2021; Zhao et al., 2023). However, other studies have shown contradictory results with no clear negative effects (or even positive effects) from biochar on soil fauna (Bamminger et al., 2014; Brtnicky et al., 2021; Domene et al., 2014, 2015; Gruss et al., 2019; Honvault et al., 2023; Jeffery et al., 2022; Verheijen et al., 2009). In general, the effects from biochar vary and the type of biochar (temperature, ash content, particle size, feedstock, etc.) and application rate strongly influence its effects on soil fauna and their ability to perform their essential functions (Brtnicky et al., 2021; Lehmann et al., 2011; Marks et al., 2016; Prodana et al., 2019; Verheijen et al., 2009).

2.4 Soil health

In the broader field of soil science, there is not yet a consensus regarding terminology and methods so this section will provide a brief overview of soil health and terminology according to the current state-of-the-art. The field of soil biology, function and ecosystem services is vast and many concepts will be covered here in limited depth; for more information the reader is referred to Soil Functions and Ecosystem Services – A Literature Review (Part 2/2) (Drenning, 2021c) and other more extensive, in-depth reports, e.g., (Faber et al., 2022; FAO et al., 2020; Orgiazzi et al., 2016; Turbé et al., 2010).

While *soil health* and *soil quality* have tended to be used interchangeably (Bünemann et al., 2018), soil health is preferred in this PhD.-work in accordance with the terminology and conceptual framework established in the recent EJP SOIL SIREN project (Faber et al., 2022). Historically, *soil quality* was used to refer broadly to the capacity of a soil to perform its functions necessary for its intended end use (Garbisu et al., 2011; Karlen et al., 2003, 1997; USDA Natural Resource Conservation Service, 2015; Volchko et al., 2013), which was primarily based on nutrient availability for crop production. Doran and Zeiss (2000) introduced the term *soil health* to better account for the biological component of soils, which was originally defined as *the capacity of soil to function as a vital living system, within ecosystem and land-use boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and promote plant and animal health*. Additional distinctions have recently emerged between the terms that are worth taking into consideration (as shown in Figure 2-4).

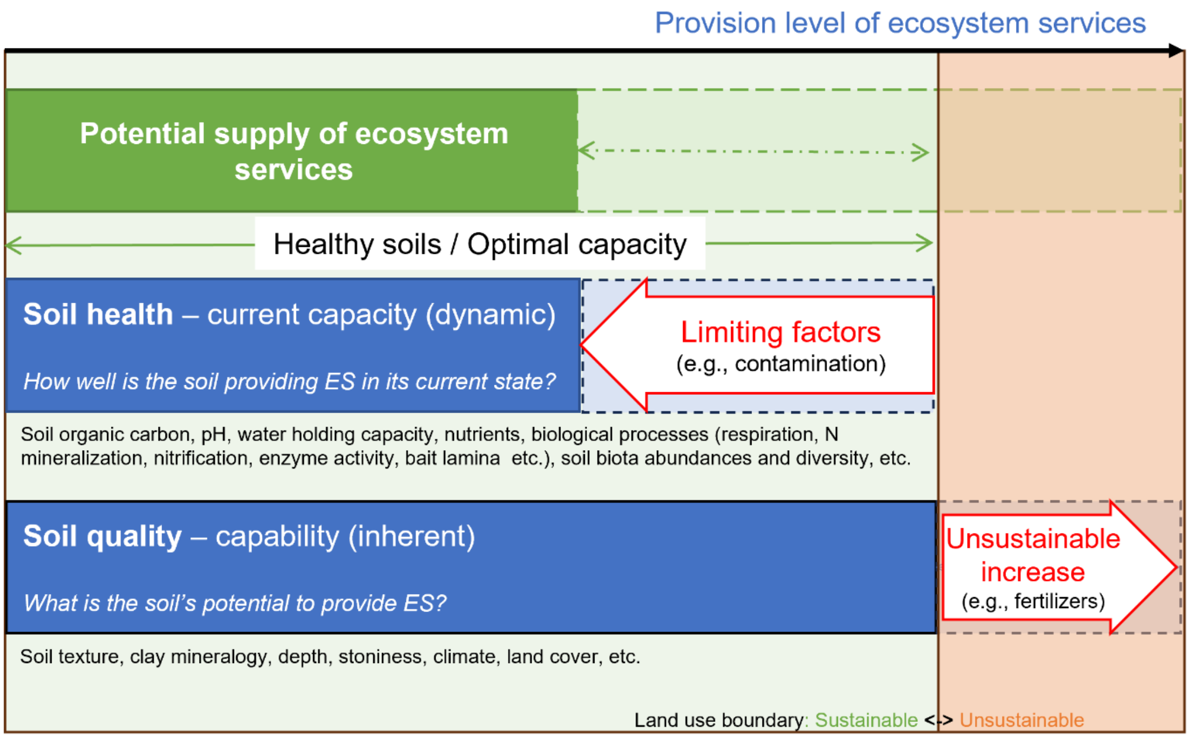

Figure 2-4. Graphical representation of the relationships between soil health, soil quality (with relevant indicators listed underneath) and the potential supply of ecosystem services against a theoretical boundary for sustainable land use, modified from (Faber et al., 2022).

As shown in Figure 2-4, soil quality refers to the ‘capability’ or ‘potential’ of a soil to deliver ecosystem goods and services, which is largely dependent on inherent or intrinsic properties of the soil that are not easily influenced by management such as soil depth, texture, type, climate, land use, etc., and is usually directed towards the quality of a soil for agricultural production of food, fiber and fuel. Soil health, on the other hand, refers to a soil’s ‘actual (current) capacity’ or status for performing its functions to deliver ecosystem goods on services under current management practices or degradation levels, which is often based on dynamic, management-influenced properties (Bünemann et al., 2018; EEA, 2022; Faber et al., 2022; Hein et al., 2016; Lehmann et al., 2020). Factors of scale and scope (e.g., local site versus regional level) can also differ between uses (Faber et al., 2022; Lehmann et al., 2020).

Ecosystem services (ES) are usually defined as *the direct and indirect contributions of an ecosystem to human well-being* (TEEB, 2010). The provision of ES for human well-being is generally understood as a stepwise process (Haines-Young and Potschin, 2010; Potschin and Haines-Young, 2016), and is typically depicted as a cascade model showing a production pathway from biophysical structures and processes contributing to ecosystem functions (supporting or intermediate services) and ecosystem services (final services) that provide benefits to humans that could then be economically valued (shown in Figure 2-5 with examples for soils). Soils have traditionally been neglected in ES classification and assessment methods such as the Common International Classification of Ecosystem Services, CICES (Haines-Young and Potschin, 2018), so many similar frameworks and typologies have been developed for soil-based ES (e.g., Adhikari and Hartemink, 2016; Dominati et al., 2010; Faber et al., 2022; Greiner et al., 2017; Jónsson and Davídsdóttir, 2016; Kibblewhite et al., 2008).

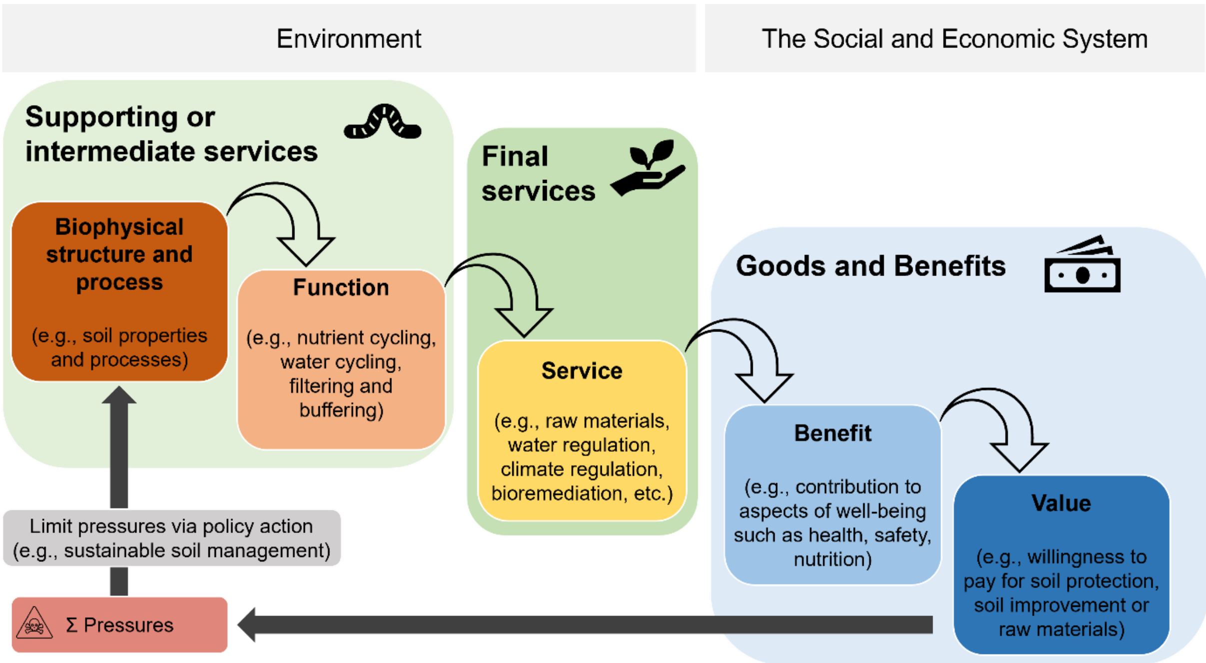

Figure 2-5. The cascade model, adapted from (Haines-Young and Potschin, 2010), *and modified for the soil environment based on* (Greiner et al., 2017).

In the context of soils, ES are soil related if their supply is directly and quantifiably controlled by soils and their properties, processes and functions (Paul et al., 2021). *Soil function* is a confusing term that has been used inconsistently (Baveye et al., 2016; Bünemann et al., 2018; Faber et al., 2022), but is defined by Bünemann et al. (2018) as *(bundles of) soil processes that underpin the delivery of ecosystem services*, which can be considered equivalent to 'supporting' ecosystem services (ES) that underpin the delivery of provisioning, regulating and cultural ES (Potschin and Haines-Young, 2016). Indeed, many soil-based ES are considered supporting services (e.g., nutrient cycling and provisioning, soil formation, structure and maintenance, habitat and biodiversity) or regulatory services (e.g., climate regulation and carbon sequestration, water purification, regulation and cycling, soil bioremediation, pest and disease control and others) (Adhikari and Hartemink, 2016; Brussaard, 2013; Dominati et al., 2010; FAO et al., 2020; Haygarth and Ritz, 2009; Jónsson and Davídsdóttir, 2016; Orgiazzi et al., 2016; Robinson et al., 2013; Wall et al., 2004)). Soils are also critical for provisioning services such as providing a foundation for biomass production (food, fibre and fuel) and structures, water supply, and genetic resources (Adhikari and Hartemink, 2016; Jónsson and Davídsdóttir, 2016; Paul et al., 2021; Turbé et al., 2010).

The soil-based subset of ES can thus be generally understood as functional outputs of biophysical, biochemical, and physicochemical processes resulting from highly complex interactions between functional assemblages (i.e., specific groups of functionally-related soil organisms such as biological regulators, ecosystem engineers and chemical engineers) of soil biota and the abiotic physical and chemical soil environment (Birgé et al., 2016; Brussaard, 2013; Faber et al., 2022; Fox et al., 2020; Kibblewhite et al., 2008; Turbé et al., 2010). Some experts propose that ES are outputs of the biological component of the soil ecosystem, i.e., soil biota as 'service providing units' (SPU), which provide ES through well-defined 'ecological production functions' (EPF) that can mathematically relate characteristics of the SPU (e.g., abundance, biomass, function) to the ecological outputs that drive ES delivery (Faber et al., 2022, 2021; Munns et al., 2015). However, this terminology is not yet widespread and other authors maintain that the dichotomy between biotic and abiotic soil services is confusing and that abiotic flows should be an inherent part of ES (Fox et al., 2020; Meulen et al., 2016; Meulen and Maring, 2018).

There are many published methodologies for soil quality, soil health and soil function assessments (e.g., Andrews et al., 2004; Epelde et al., 2014; Gugino et al., 2009; Moebius-Clune et al., 2016; Pulleman et al., 2012; Rutgers et al., 2012; Thomsen et al., 2012; Velasquez et al., 2007; Volchko et al., 2014, 2019). The methodologies vary but are often based on a multi-parametric approach using a wide range of soil quality indicators (SQI), or 'descriptors' – *a parameter describing a physical, chemical, or biological characteristic of soil health* (EC, 2023), which are correlated (or at least thought to be) with relevant soil functions and ES (Baveye et al., 2016; Bünemann et al., 2018; Kibblewhite et al., 2008). SQI thus serve as proxies of these key soil properties and processes relating to important soil functions (Bünemann et al., 2018; EC, 2023; EEA, 2022; Faber et al., 2022; Lehmann et al., 2020; Meulen and Maring, 2018). Assessment and monitoring of soil health has historically focused on abiotic, physico-chemical parameters (e.g. pH, organic matter content, CEC, nutrient availability, water capacity, soil texture, etc.), but biological parameters are becoming increasingly used in soil health assessments as indicators since they can provide a more direct measure of soil functioning (Alkorta et al., 2003; Bonilla-Bedoya et al., 2023; Bünemann et al., 2018; Epelde et al., 2009a; Faber et al., 2013; Garbisu et al., 2011; Gómez-sagasti et al., 2012; Griffiths et al., 2016; Orgiazzi et al., 2016; Ritz et al., 2009; Stone et al., 2016; Wall et al., 2012). Typically, a 'biological indicator' refers to measuring the biomass, abundance, activity and/or biodiversity of common/representative species performing important functions in the ecosystem, such as earthworms, bacteria and fungi, collembola and nematodes. Thus, these biological indicators (i.e., *bioindicators)* can be used to assess the status and changes in ecological soil properties and processes within a given physico-chemical context, and increasingly are valued for inclusion in soil health assessment, site-specific management strategies, measuring the state of ecosystems and for monitoring the progress of ecosystem recovery or restoration (Bonilla-Bedoya et al., 2023; Doran and Zeiss, 2000; Gómez-sagasti et al., 2012; Orgiazzi et al., 2016; Ritz et al., 2009). There are, however, many indicators that could be potentially applied to assess soil health, so methods such as the 'logical sieve', where potential SQI are filtered and scored according to multiple criteria (e.g., meaningfulness, standardisation, accessibility, availability, cost-efficiency, sensitivity, repeatability, etc.), are commonly used to select a minimum dataset of suitable indicators for a particular application (Faber et al., 2013; Griffiths et al., 2016; Ritz et al., 2009; Stone et al., 2016).

Raw data from laboratory measurements can be difficult to understand, especially for non-experts. To interpret the measured values of SQI, an approach would entail using scoring functions (i.e., 'more is better', 'less is better', or 'optimum') to define ranges for when the value of a SQI is 'good' for a particular soil type and context and derive normalized indicator scores (Andrews et al., 2004; Lehmann et al., 2020; Obriot et al., 2016; Rinot et al., 2019; Thoumazeau et al., 2019). In many proposed methods for soil health assessment, scoring and weighing the relative importance of *SQI* for specific soil functions are considered important parts of calculating the soil health indices and, importantly, can substantially differ depending on the soil type (e.g., (Andrews et al., 2004; Gelaw et al., 2015; Lehmann et al., 2020; Obriot et al., 2016; Rinot et al., 2019). The resulting normalised scores could then be used to calculate an aggregated soil quality/health index, which is *the integration of multiple soil indicators into one single score to estimate soil health* (Lehmann et al., 2020). The soil health index can be a single, overall value or multiple indices connected to specific categories such as ES (Blanchart et al., 2018; Hyun et al., 2022; Rinot et al., 2019; Su et al., 2018; Thoumazeau et al., 2019). Several studies have successfully utilised various quantitative soil quality indices based on a combination of physical, chemical and biological indicators to evaluate the effects on soil health of soil remediation techniques such as phytoremediation (Burges et al., 2017, 2016; Epelde et al., 2014b; Mench et al., 2022) or various soil amendments such as lime (Mijangos et al., 2010) and biochar (Bera et al., 2016). For example, the 'treated soil quality index', originally proposed in Mijangos et al. (2010), was modified from the soil quality index presented in Bloem et al. (2006) to better fit the aim of evaluating the changes resulting from in-situ gentle remediation in comparison to an untreated, control soil.

2.5 Cost-benefit analysis

Remediation is not inherently sustainable (Anderson et al., 2018; Bardos et al., 2020a; Cundy et al., 2016) and poorly planned projects, or high-impact remediation options, can have substantial negative effects, e.g., impacts on provisioning of ecosystem services (Rosén et al., 2015; Söderqvist et al., 2015). Selecting the most preferable remediation alternative for a particular site from a sustainability point of view or allocating limited resources between sites, therefore, is a non-trivial decision for decision-makers (Söderqvist et al., 2015). Cost-benefit analysis (CBA) is a decision-support tool for economic analysis where the cost-benefit rule is based on utilitarian, welfare economics for expressing positive (benefits) and negative (costs) effects on human well-being (Johansson and Kriström, 2018; Romjin and Renes, 2013), including both financial costs and benefits as well as positive and negative externalities (i.e., positive or negative effects on health and the environment in terms of provisioning of ecosystem services, carbon emissions, noise, traffic etc.) that fall outside the scope of traditional financial analysis (Figure 2-6.)

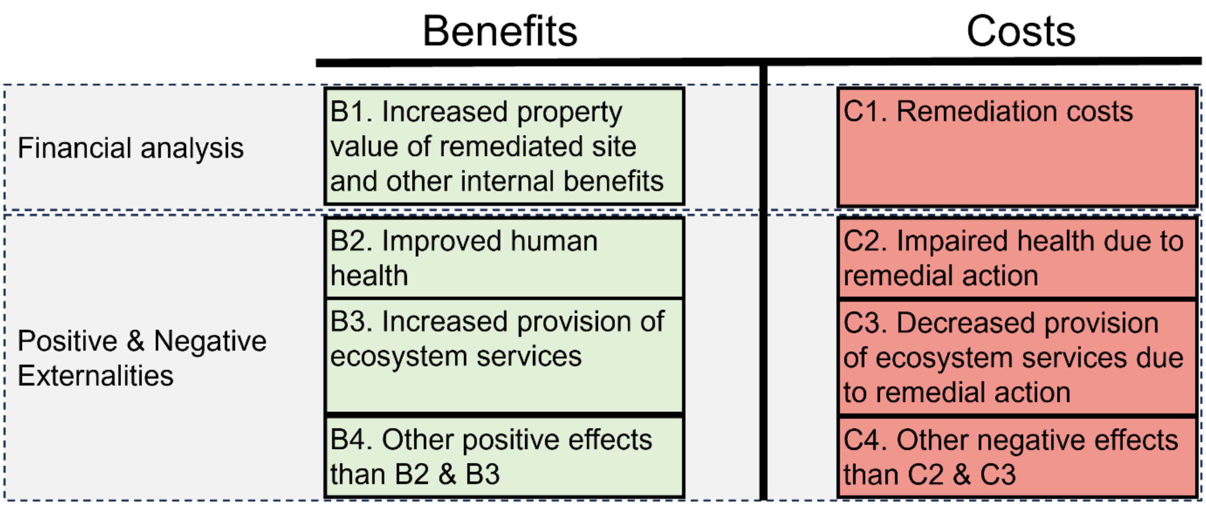

Figure 2-6. Cost and benefit items relevant for remediation projects to include in a CBA, indicating the scope of financial analysis and externalities, after (Söderqvist et al., 2015).

Using monetary units makes it possible to weigh the costs of a remedial action against associated benefits over a certain time horizon, with all costs and benefits over this time horizon calculated as present values (PV) by discounting using a social discount rate, and in relation to a reference alternative. A positive net sum of discounted costs and benefits, i.e., net present value (NPV) > 0, means that the remedial action entails a social profitability, whereas a negative net sum indicates social loss (Johansson and Kriström, 2018; Rosén et al., 2015; Söderqvist et al., 2015). The calculation of the NPV can be described as follows (Söderqvist et al., 2015):

$$NPV = \sum_{t=0}^{T} \frac{1}{(1+r_t)^t}(B_t - C_t)$$

where B_t ($B1_t$ + $B2_t$ + $B3_t$ + $B4_t$) and C_t ($C1_t$ + $C2_t$ + $C3_t$ + $C4_t$) are the sum of discounted benefits and costs at time t (usually years), r_t is the social discount rate at t, and T is the time horizon associated with the benefits and costs.

There are many important considerations that influence the outcome of a CBA. For instance, the choice of social discount rate can greatly impact the resulting NPV, especially in applications with long time horizons of e.g., 200 years (Söderqvist et al., 2015). In essence, the choice of discount rate reflects the emphasis placed on future values: the higher the discount rate the lower the present value of the future benefits and costs, other things being equal (Johansson and Kriström, 2018). The choice of social discount rate is important when valuing, for example, the expected positive or negative externalities of a remediation alternative occurring long into the future and the choice of discount rate can become an issue of inter-generational equity. Placing a value on future costs and benefits is difficult but recent guidance from the United States (OMB, 2023) and the UK (HM Treasury, 2022) recommend a low, long-term discount rate or even a declining discount rate over the time horizon, which can make projects with long-term benefits appear more profitable and account for uncertainties in the long term. Distributional analysis is also an important part of an economic analysis where the expected costs and benefits as effects of a remedial action are distributed across stakeholders as potential beneficiaries or payers (EC, 2014; Johansson and Kriström, 2018; Romjin and Renes, 2013); where, for example, a developer may benefit from increased property values while society may benefit/pay from an increase/decrease in the provision of ecosystem services. Also, since there are often significant uncertainties associated with cost and benefit items and assumptions, a sensitivity analysis to assess the sensitivity of the model to changes in key parameters, such as the social discount rate, is important to include. Probabilistic approaches are also recommended to account for uncertainties in a CBA (EC, 2014; Johansson and Kriström, 2018; Söderqvist et al., 2015).

CBA has been highlighted as a decision-support method with great potential for incorporating sustainability measures in an understandable, easy-to-use approach and account for the value of restoring or preserving soil functionality and ecosystem services (ES) (Onwubuya et al., 2009). Based on an anthropocentric ethics, a CBA investigates consequences for human well-being from environmental change, such as those caused by changes in the supply of ecosystem services due to remedial action, and, whenever possible, monetizes them by expressing their value in terms of monetary units. Economic valuation of ES contributes to the decision-making process by integrating ES into decision-support and engaging potentially responsible parties to participate in both the remediation process and funding of risk mitigation measures (Harwell et al., 2021). Indeed, the necessity of ecosystem service assessment to demonstrate the value of ES for society at large has been illustrated well in the Swedish context and highlights both the necessity of clear valuation of ES for sound decision-making and ensuring that the long-term needs of society for functioning ecosystems are met (SEPA, 2018b; SOU, 2013). Considering soils, economic estimates of the value of ecosystem services delivered by soil biodiversity needs to be provided in order to perform cost-benefit analyses for soil remediation or other measures to protect soil biodiversity, and the consequences of soil biodiversity mismanagement have been estimated to be in excess of 1 trillion dollars per year worldwide (Turbé et al., 2010).

However, many ES may not be easily monetizable and there might be other values that cannot be captured by a CBA, e.g., the intrinsic value of soil health or ecosystems. For example, many wider benefits (or costs) that may result from remediation such as the loss or gain of soil functionality are difficult to account for in a CBA but are an important aspect of soil remediation (Chen and Li, 2018). Despite the potential limitations, there are a number of studies that have investigated economic valuation methods such as CBA for the application of NBS (Alshehri et al., 2023a, 2023b; Masiero et al., 2022; Quaranta et al., 2021; Valck et al., 2019), GRO (Cervelli et al., 2016; Chen and Li, 2018; Compernolle et al., 2013, 2012; Dudai et al., 2018; Guo et al., 2022; Lewandowski et al., 2006; Thewys and Kuppens, 2008; Wan et al., 2020, 2016; Witters et al., 2012), or brownfield remediation and redevelopment more generally (Huysegoms et al., 2019, 2018; Söderqvist et al., 2015; Volchko et al., 2020).

Regarding valuation of soil biodiversity, there are existing approaches for valuing ecosystem services which have been also applied to value soil-based ES (Dominati et al., 2014; Franceschinis et al., 2023; Greiner et al., 2017; Jónsson and Davídsdóttir, 2016; Meulen and Maring, 2018; Pascual et al., 2015; Robinson et al., 2013; Turbé et al., 2010). Soil ES valuation is strongly linked to land management (Pereira et al., 2018). Most studies focus on agricultural contexts and a limited number of more easily valuable soil-based ES (e.g., carbon storage, nutrient retention) as well as ES loss due to soil erosion and cost-based valuation methods (Jónsson and Davídsdóttir, 2016; Meulen and Maring, 2018). One approach to economically value soil biodiversity utilizes the terminology of ‘production functions’ to describe the contribution of individual ‘service providing units’ (SPU) to the delivery of ES as ‘ecological production functions’ (EPF), which can in term become an input to economic production functions (Faber et al., 2022, 2021; Munns et al., 2015). These stepwise relationships could be used for economic valuation of the marginal product of soil biodiversity in terms of their incremental contribution to ES and monetizing the consequences of a reduction in the provisioning of ES (due to e.g., soil contamination) for human well-being (Faber et al., 2021). Such economic valuations of soil biodiversity would facilitate their inclusion in a CBA for improved decision-support and land management (Faber et al., 2022, 2021).

The challenge is that much of the value of natural capital cannot be effectively expressed in monetary terms, and ‘use’ values may be captured while ‘non-use’, ‘indirect use’, or ‘option’ values of ecosystems and biodiversity may be neglected leading to their continuing loss and degradation (TEEB, 2010). It should also be noted that there are many different worldviews and ethical or knowledge systems that inform how humans derive layers of value from nature that preclude economic valuation (Pascual et al., 2023). Aggregating all these types of value that humans derive from natural capital is often referred to as the ‘total economic value’ (TEV) (Baveye et al., 2016; Pascual et al., 2015). In economic terms, soil biodiversity can be viewed as an economic asset or portfolio of resources that build up soil natural capital, which in turn can be economically valued, and the flow of ecosystem services derived from soil biodiversity is the accrued interest or return (positive or negative) from managing the asset (Hein et al., 2016; Pascual et al., 2015). Valuation methods and the total economic value of soil biodiversity and ES are described in more detail in (Drenning, 2021c), and more extensive discussions in (Baveye et al., 2016; Pascual et al., 2015; Pereira et al., 2018; Robinson et al., 2013).

3 METHODOLOGY

This chapter provides the methodology followed to achieve the overall aim and specific research objectives.

In this chapter, a brief overview of the methodology (see Figure 3-1) to fulfil the specific research objectives as followed in the five appended papers (I-V) is provided in chronological order from publication date. The cases studies for Papers II and III are briefly described under the respective headings. The field experiment at Kolleberga tree nursery is also briefly described and relates to Papers IV and V.

3.1 Literature reviews

Multiple literature reviews contributed to fulfilling the overall aim and the specific objectives of this thesis through thematic exploration. Extensive, semi-systematic reviews were carried out and targeted towards synthesizing need-to-know, practical information for 1) successful and effective application of GRO (Drenning, 2021b) and 2) understanding and assessing soil functions and ecosystem services primarily in the context of contaminated sites (Drenning, 2021c). These reviews form the theoretical foundation of the Ph.D.-project and supported the development of the risk management framework and establishment of the field experiment at Kolleberga. Further details about the methods for reviewing literature and writing can be read in each report.

The review in Drenning et al. (2020) (Paper I) aimed to identify key intersecting themes between related fields that can be used to support decision-makers by reinforcing the connections to sustainable remediation and development. Additional literature reviews were carried out as part of all the scientific papers and were more targeted towards exploring and understanding the fundamental concepts that were directly applied in each study. The subjects of interest are shown in Figure 3-1 and more information can be found in the respective paper.

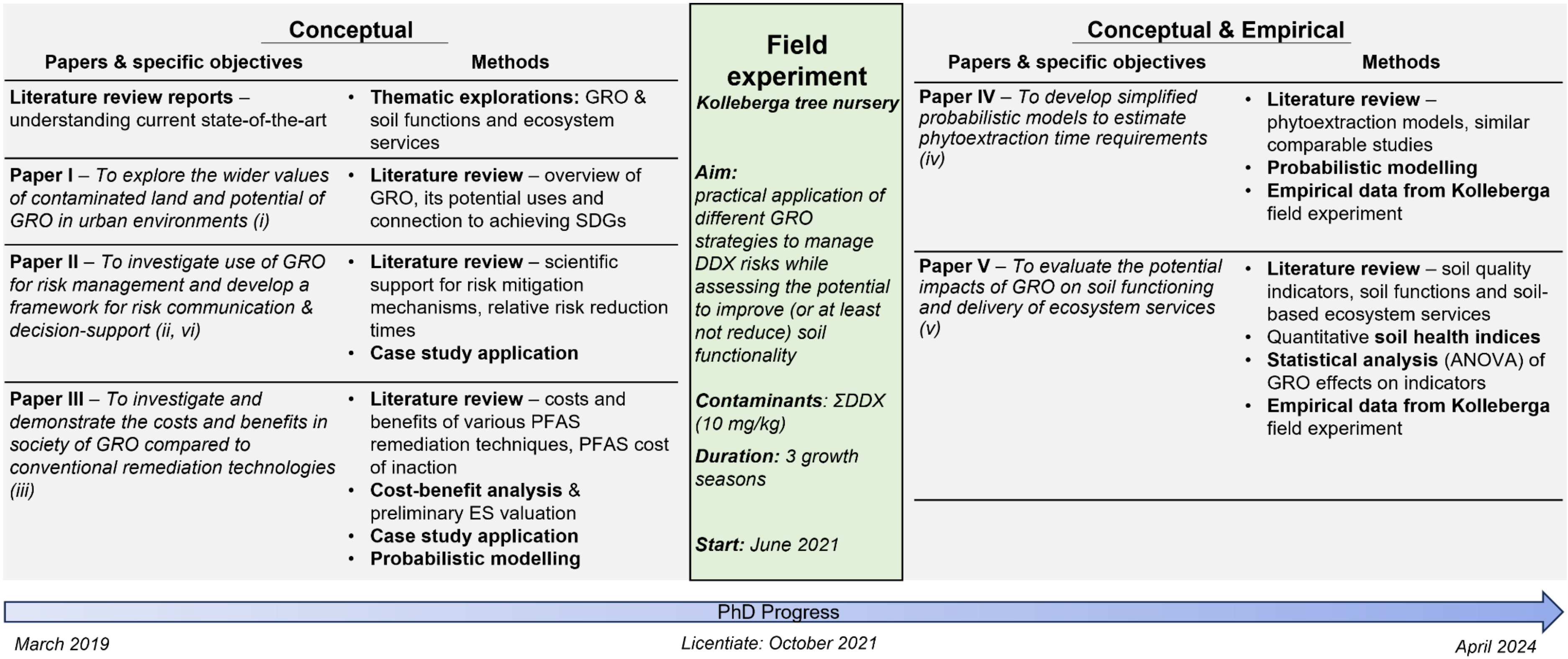

Figure 3-1. Overview of the methods used in the Ph.D.-thesis for each paper to achieve the specific objectives (in parenthesis).

3.2 Development and application of GRO risk management framework

The GRO risk management framework was developed and applied in Drenning et al. (2022) (Paper II), which supported the fulfilment of research objective ii) t*o investigate how different GRO strategies can be used for risk management by identifying and finding support for relevant risk mitigation mechanisms and their associated risk reduction times*, and vi) *to develop a framework to effectively communicate about risk management using GRO and identifying feasible alternatives*. The general process followed to develop and apply the GRO risk management framework is briefly described in this section, see Paper II for more details.

As reported in Drenning et al. (2022) (Paper II), the working process for **development** of the risk management framework for GRO proceeded by first conceptualizing connections between GRO, risk mitigation mechanisms and their impact on ecological and human health risks. An extensive literature review was undertaken to identify studies that can support the hypothesized risk mitigation mechanisms. The conceptualization is illustrated in a conceptual diagram and forms the basis for the generic framework. Mapping of the expected timeframes for effective risk reduction of different GRO and contaminant groups was based on existing literature. The time perspectives for different GRO and groups of contaminants were added to the figure which altogether forms the generic risk management framework.

The generic framework can be adapted to account for certain site-specific considerations and envisioned future land use for **application** of the framework at a particular site. To account for the varying contamination levels and provide an indication of the relative risk, the risk quotients (RQ) for each contaminant were calculated by dividing the mean (total) concentration in the soil by either the corresponding health-based SGV or the lowest environmental SGV determined in the land-use specific SEPA model. Drenning et al. (2022) (Paper II) presents more information on this part of the working process and preliminary results concerning the varying risk assessment per modelled green land use exposure scenario and corresponding SGVs. The GRO risk management framework has been applied for a case study site, Polstjärnegatan in Gothenburg, Sweden (see Paper II for more details), but is demonstrated here only for the Kolleberga site.

Since its original publication, the risk management framework for GRO has been **updated** based on feedback from various experts and stakeholders gained during presentations and formal workshops (Table 3-1). Direct questions were posed to the participants regarding: how they think they could use the framework and if not, what is missing and how could it be improved? The participants varied in the different forums but there was representation from regulatory agencies, landowners, research, consultants, communication, contractors, municipalities, and county level agencies, which provided a broad range of perspectives on the framework and the potential for improvement.

Table 3-1. Summary of meetings and workshops where the GRO risk management framework was discussed to feedback provided to improve it. BAPR = Baltic Phytoremediation project (https://hassleholmmiljo.se/hassleholm-miljo---privatperson/kundservice/bapr-projektet.html)

Date	Context	Purpose	Representation
August 2021	PhD project reference group meeting	Review the ongoing PhD work and provide feedback	Group of experts in Sweden working with contaminated sites
April 2022	BAPR webinar	Present the ongoing PhD work and organize small group workshops in break-out rooms discussing with participants the utility of the framework and potential improvements	Listening and participation in workshop participation from wide range of experts in Sweden working with contaminated sites
June 2022	BAPR phytoremediation seminar	Present the ongoing PhD work to participants in the BAPR project in person and discussing with participants the utility of the framework and potential improvements	Participants in the BAPR project representing multiple countries around the Baltic seas
September 2022	BAPR workshop	Invited participants to a workshop in Gothenburg for presentations relating to phytoremediation as well as an in-person workshop to use the GRO framework for a current project as a case study to both come up with viable GRO strategies and provide feedback on using the GRO framework	A small group of ca. 20 experts in Sweden interested in phytoremediation
September 2022	PhD project reference group meeting	Review the ongoing PhD work and provide feedback	Group of experts in Sweden working with contaminated sites

3.3 Cost-benefit analysis

Cost-benefit analysis was carried out in Drenning et al. (2023) (Paper III), which supported the fulfilment of research objective iii) *to investigate and demonstrate the potential costs and benefits in society of GRO compared to conventional remediation technologies*. The general process followed to perform the cost-benefit analysis and the case study site, Stockholm Arlanda Airport, is briefly described in the following section, see Paper III for more details.

In a CBA, cost and benefit items of remediation alternatives are monetized in comparison with a reference alternative. The cost and benefit items are discounted over a time horizon of 120 years using a real social discount rate of 3.5%, as recommended for CBA in Sweden (STA, 2020). Present values (*PV*) for each alternative and the net present value (*NPV*) are calculated using as follows (Eq. 1,2) (Söderqvist et al., 2015):

$$NPV = \sum_{i=1}^{N} PV(B_i) - \sum_{j=1}^{M} PV(C_j), \qquad \text{(Eq. 1)}$$

$$PV(B_i) = \sum_{t=0}^{T} \frac{1}{(1+r)^t} B_{it} \text{ and } PV(C_j) = \sum_{j=0}^{T} \frac{1}{(1+r)^t} C_{jt}\ , \qquad \text{(Eq. 2)}$$

where T is the time horizon, r is the social discount rate, and t is the time when benefits and costs occur for each benefit item (B_i, $i = 1 \ldots N$) and cost item (C_j, $j = 1 \ldots M$).

The most profitable remediation alternative for society is that with the highest positive *NPV*. If all the *NPV*s are <0, then the remediation alternative with the lowest negative *NPV* results in the least social loss in economic terms.

The CBA was carried out by adapting the method presented in Söderqvist et al. (2015) and Volchko et al. (2020), according to the following steps:

1. Identification of remediation alternatives, a reference alternative, the social discount rate and a relevant time horizon associated with the alternatives.
2. Identification of costs and benefits associated with each remediation alternative and defining scenarios to account for model uncertainties.
3. Quantification and monetization of costs and benefits by defining a minimum, maximum, and most likely value based on literature studies and personal contact with contractors and assigning probability distributions to input variables and cost and benefit items to represent the uncertainties in these input variables.
4. Calculating the *NPV* and associated uncertainties of each alternative by using Monte-Carlo simulations and discounting the cost and benefit items using a social discount rate and a relevant time horizon, simulating the CBA for the different defined scenarios, and investigating the results to evaluate the uncertainties in *NPV*s of the remediation alternatives and performing sensitivity analyses.
5. Concluding about the social profitability and ranking of remediation alternatives to provide recommendations as decision-support.

See Paper III and the accompanying supplementary material (SM) for more details regarding the cost and benefit items relevant for CBA in the remediation project (and Figure 2-6) and the methods used to quantify them, sensitivity analysis of input variables, the assumptions made to form the base model scenario as well as the multiple scenarios created to test model uncertainties, including accounting for different a) reference alternatives ('do nothing' or total excavation and disposal, Alt 0); b) social discount rates (%); c) spreading scenarios – large and small spreading; and d) magnitude of annual avoided cost of inaction (AACOI; MSEK).

3.3.1 Case study description: Stockholm Arlanda Airport

The firefighting training site is situated at Stockholm Arlanda Airport outside of Stockholm, Sweden, where aqueous film-forming foam (AFFF) containing PFAS was used until 2011 (Gobelius et al., 2017).

The geology consists primarily of surface layers of glacial clay underlain by sandy glacial till which varies between a depth of 1.5-8 m below the surface depending on the thickness of the clay (Rosenqvist et al., 2017). The firefighting training site is located within an an area with a top layer of beach sand and silt, varying between 0.3-2 m in thickness, that thins out and disappears altogether closer to the landing strips southwest of the training site. Filling material of sand and gravel form the immediate surface layer of 0.5 m in the built area above the natural geological soil layers. Hydrogeological investigations have determined that there are two distinct aquifers: an unconfined aquifer in the upper layer of sand and silt above the clay and a confined aquifer in the sandy glacial till below. The upper aquifer is contaminated with PFAS and constitutes an important spreading pathway for PFAS off-site to nearby surface water systems. At the training site, the groundwater depth ranges between 1.1-1.8 m across the site, but the water table is at or near the surface layer in some areas. It has also been determined that the groundwater flows in a south-westerly direction, towards a nearby open ditch that is in hydraulic contact with nearby surface water but away from and not in contact with the glacio-fluvial sand deposits to the northeast of Stockholm Arlanda Airport with high hydraulic conductivity (Rosenqvist et al., 2017).

Sampling campaigns at the site have extensively investigated PFAS concentrations in soil, sediment, groundwater, surface water and aquatic organisms. An extensive soil sampling campaign by Rosenqvist et al. (2017) analysed 40 soil samples and reported high maximum values with significant variation between the different types of PFAS compounds in concentration as well as spreading distance from the source (see Paper III SM for more details). The sum total of the 13 analysed PFAS compounds ranged from 0.63 ng g^{-1} to 2 700 ng g^{-1} dw. They found that PFOS (a subset of PFAS compounds) made up 88% of the PFAS compounds measured in soil with an average value of 234 ng g^{-1} dw across the site. An important note is the median value of 34 ng g^{-1} dw, indicating large differences in measured concentrations closer to the source (the training site hotspot) versus further downstream away from the immediate source. The depth to which the soil is contaminated with PFAS varies considerably between the immediate hotspot and soil layers throughout the rest of the site. For comparison, preliminary guidelines have been established by the Swedish Geotechnical Institute (SGI) which provide a soil guideline value of 20 ng g^{-1} dw for PFOS for "less sensitive land use" and 3 ng g^{-1} dw for "sensitive land use" to protect human health and the environment (Pettersson et al., 2015). The guideline value for groundwater is 45 ng L^{-1}. The tested concentrations in both soil and groundwater greatly exceed the guideline values in many sampling locations.

3.3.2 PFAS remediation alternatives for the site

As emerging contaminants have gained widespread attention only in recent years, remediation technologies to immobilize, remove or destroy PFAS and its associate compounds are not yet well-established (Held and Reinhard, 2020; ITRC, 2018; Ok et al., 2020; Smith et al., 2016). Indeed, a combination of multiple technologies (i.e., treatment chains (Lu et al., 2020)) is often required to remediate a site effectively (ITRC, 2018; Merino et al., 2016). Based on literature review, five remediation alternatives were developed, where each alternative is a combination of several technologies for managing the risks posed by PFAS contamination in soils, summarized in Table 3-2 and described in detail in SM of Paper III.

Table 3-2. Overview of the remediation alternatives for PFAS-contaminated soils at the Stockholm Arlanda Airport site. REF indicates the reference alternative used in the CBA.

CBA of Alt 1-5 compared to Alt 0, i.e., 'total excavation' (base scenario)		REF	Remediation alternatives evaluated against Alt 0				
		Alt 0	Alt 1	Alt 2	Alt 3	Alt 4	Alt 5
CBA of Alt 0-5 compared to the 'do nothing' case	REF	Remediation alternatives evaluated against 'Do nothing'					
	Do nothing	Alt 0	Alt 1	Alt 2	Alt 3	Alt 4	Alt 5
Remedial actions at the hotspot							
Excavation (before treatment)		X	X	X	X	X	X
Ex-situ stabilization/solidification (S/S) with cement and activated carbon on-site		X	X	X	X		
Ex-situ thermal treatment off-site						X	
Ex-situ soil washing On-site							X
Backfilling with the treated masses		X	X	X	X		X
Backfilling with pristine soils						X	
Remedial actions at the rest of the site							
Excavation (before treatment or disposal)		X					
In-situ stabilization/solidification (S/S) with cement and activated carbon			X				
In-situ immobilisation/stabilization with activated carbon without cement				X			
Phytoremediation with birches and spruces					X	X	X
Landfilling at a disposal site		X					
Backfilling with pristine soils		X					
Achievement of risk reduction targets (years required to manage risks)							
Hotspot	-	2	2	2	2	2	2
Rest of site	-	2	2	2	20	20	20
Long-term project management and monitoring	-	0	0	20[b]	20[b]	20[b]	20[b]

[a] CBA: cost-benefit analysis. [b] It is assumed that risk reduction can take a shorter time, but the site may not be left without monitoring and adaptive management when using gentle remediation options (Drenning et al., 2021).

3.4 Field experiment: Kolleberga tree nursery (Ljungbyhed, Sweden)

3.4.1 Site description

The field experiment site is the Kolleberga former tree nursery in the Scania region of Southern Sweden (Ljungbyhed) encompassing fenced agricultural fields of ca. 23 hectares. The site was previously operational to cultivate pine and spruce plants to serve the forest industry. Since its initial usage in 1950s, technical DDT was used to control different types of pests, both by dipping the plants in barrels of dissolved DDT as well as spraying across the field by hand and with tractors. Despite the Swedish ban on DDT in 1969, DDT and its metabolites (including both *p,p'* and *o,p'* isomers) dichlorodiphenyldichloroethylene (DDE) and dichlorodiphenyldichloroethane (DDD), hereafter collectively referred to as ΣDDX, are still detected in the agricultural fields. The ΣDDX composition in the field soil is approximately 77% *p,p'*-DDT, 9% *o,p'*-DDT, 4% *p,p'*-DDD, 2% *o,p'*-DDD, 8% *p,p'*-DDE, and <1% *o,p'*-DDE, which is similar to the makeup of technical DDT (ATSDR, 2022) with marginally increased degradation products indicating that little degradation has occurred since its usage. Soil concentrations of ΣDDX, (C_{soil}), at Kolleberga have been found to be in the range between 5-15 mg/kg $_{dw}$, with a maximum value of ca. 23 mg/kg $_{dw}$, to a depth of approximately 0.35 m below ground level due to repeated ploughing and mixing of the soil in the fields (Nilsson, 2019). These concentrations exceed the Swedish generic soil guideline value of acceptable levels for a less sensitive land use of 1 mg kg $_{dw}$ for the combined sum of *p,p'*- and *o,p'*- isomers of DDT, DDD, and DDE, i.e., ΣDDX. No DDX has been detected in groundwater.

The agricultural fields are no longer used for productive forestry but are managed by sowing a mixture of grasses, periodically cutting, and ploughing the grass back into the soil. The site geology is loamy glacio-fluvial sand consisting of 87% fine-medium sand, 4% silt, 7% clay, and 2% gravel and larger stones, with a bulk density of approximately 1 500 kg/m^3. The soil is well-drained and has moderate levels of organic carbon (Table 3-4). The depth to the groundwater table is ca. 4-5 m.

3.4.2 Experimental set-up

A 3-year pilot-scale field experiment was established at the Kolleberga tree nursery site according to the following steps (corresponding with Figure 3-2):

1. A transect of 50x5m was excavated to ca. 35cm depth below ground level (depth of contamination/plough depth) and moved to a soil pile.
2. The soil pile was mixed to homogenize the soil and separated into two piles of equal volume. One pile was mixed with biochar (produced by pyrolysis of wood chips and bark using a floating bed reactor at 750°C for 20 min, Table 3-3) as a soil amendment at a 3% w/w ratio.
3. In the trial area, 24 experimental plots of 2x2m and 35cm depth were dug and a fiber cloth was put into the bottom to contain the soil and roots within the soil volume.
4. The soil was randomly distributed into the plots – half with and half without biochar – corresponding to a randomized block design, i.e., in triplicate but separated into 3 blocks that contained each of the 8 treatments. Four different plants mixes were

established in the 24 plots, including pumpkin, grass mix, legume mix, and willow (Figure 3-3).

5. The remaining soil in the pile and from digging the experimental plots was put back into the excavation area to restore the excavation.

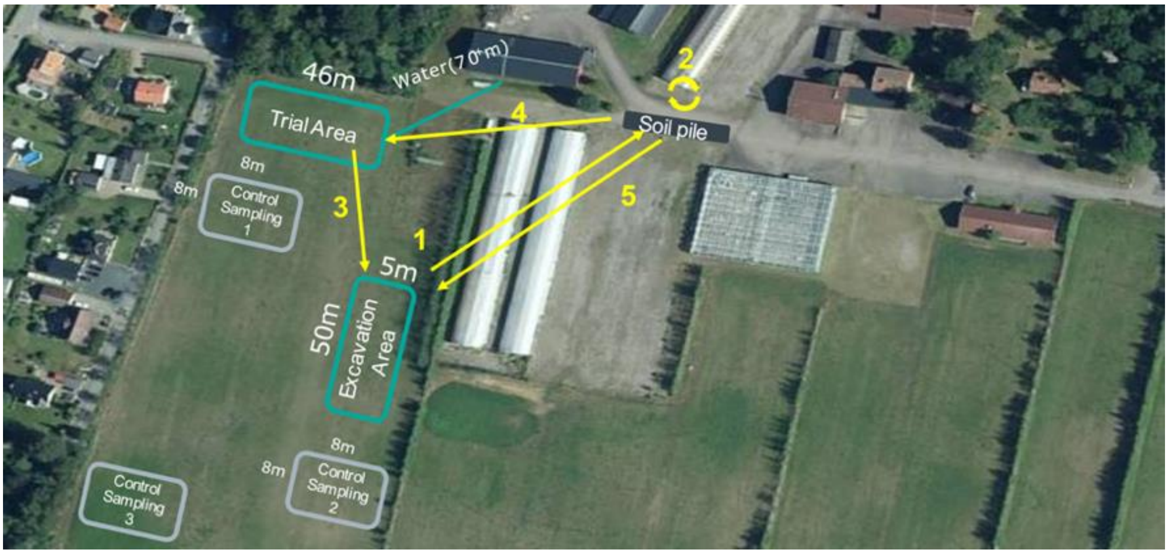

Figure 3-2. Steps followed to set up the field experiment.

Table 3-3. Biochar summary characteristics, values reported as measured in dry basis. SSA = specific surface area.

	Source Material	**Pyrolysis Temperature °C**	**Carbon content %**	**Ash content (550 c) %**	**SSA m^2/g**	**Bulk density (<3mm) kg/m^3**
Biochar[1]	Wood chips	750	82.1	14.9	247	291

[1]Sourced from NSR AB

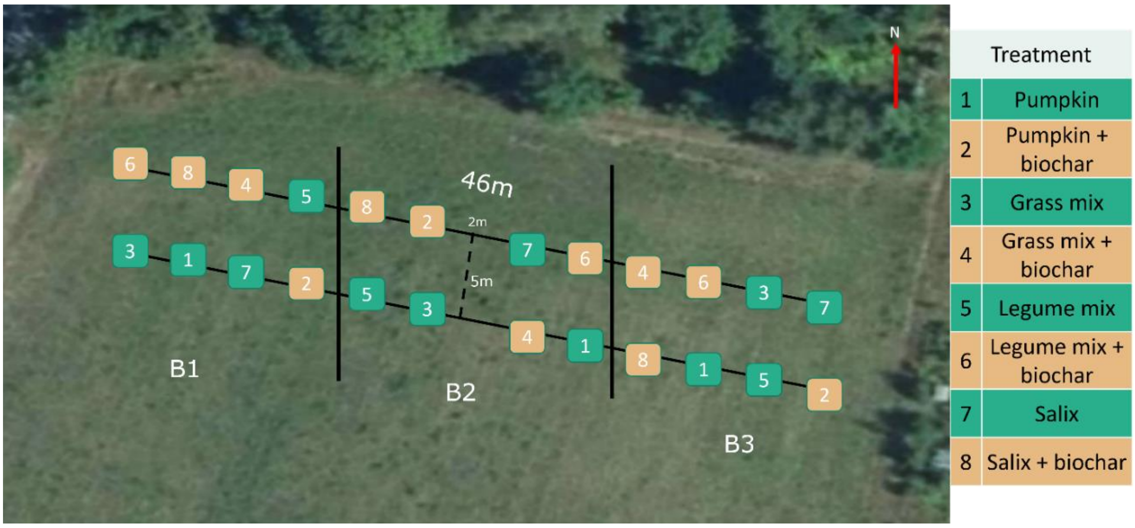

Figure 3-3. Overview map of the experimental area and treatments in a block design. Treatment numbers 1-8 correspond to the table, orange boxes are where biochar was mixed into the soil.

Three different GRO strategies were selected by applying the GRO risk management framework for Kolleberga and tested using specific plant species or a mix of species (treatment numbers corresponding to Figure 3-3): 1) phytoextraction (pumpkin, *Cucurbita pepo* ssp. *pepo* cv. Howden), 2) aided phytostabilisation (willow, *Salix viminalis* cv. Emma and Ester, and a grass-mixture of *Festuca rubra, Festuca pratensis, Phleum pratense, Poa pratensis*, and *Lolium perenne*) with biochar as a soil amendment, and 3) phyto/rhizodegradation (a mix of leguminous plants including clover, *Trifolum repens*, and alfalfa, *Medicago sativa*). The purpose of adding biochar to the soil was primarily to aid/improve the stabilization of DDX (i.e., to decrease the soil porewater concentration and bioavailability of the DDX). Amendment with biochar can also improve the physical, chemical, and biological properties of soil, and increase the soil microbial biomass and the plant biomass production, potentially having positive effects on phytoremediation as well as improving soil health, so it is evaluated in combination with all plants. The plants were watered regularly using an automated irrigation system, with each plot receiving the same amount of water, and the pumpkin and willow plants were chemically fertilized according to each plant's need. The field control was not irrigated or fertilized. The grass mixture without biochar (T3) serves as an experimental control (reference soil) that resembles the field vegetation while controlling for irrigation effects.

3.4.3 Soil and plant sampling and analysis

At the start of the experiment, 3 field control sampling areas, 3 plots with biochar and 3 plots without biochar were sampled and analyzed to determine the initial soil parameters (see Table 3-4). Soil samples were collected by digging multiple test pits in the plots with a garden shovel to a depth of 25cm, collecting soil using a hand shovel across the soil profile, and thoroughly mixing the soil in a bucket to homogenize. After each subsequent growth season, sampling points within the plots were randomized to select sampling locations, the plant biomass harvested by cutting the stems close to the ground surface level, and soil samples were collected using a small (Φ 2cm) core sampler and extracting 20 soil cores in 4 randomized locations, to a depth of 20-25cm, which were then homogenized, collected in diffusion-tight plastic bags, labelled for each individual plot and stored in cooling boxes before sending to the labs. The samples were sieved through a Φ 2mm sieve either by the responsible laboratory (first year) or directly in the field (second year).

Table 3-4. Field control soil parameters at start of experiment (n = 3), mean values ± standard deviation. ΣDDX: total DDX concentration (mg/$kg_{soil\ dw}$), POM: DDX porewater concentration (ng/L); TOC: total organic carbon ($\%_{Carbon}$); N-tot: total organic nitrogen (g/ $kg_{soil\ dw}$); P-AL, K-AL: available phosphorous and potassium (mg/100 $g_{soil\ dw}$); BR: basal respiration (mg CO_2/hour-$kg_{soil\ dw}$); MBC: microbial biomass carbon (mg C/$kg_{soil\ dw}$).

ΣDDX	POM	pH	TOC	N-tot	P-AL	K-AL	BR	MBC
7.52 ± 1.84	266 ± 175	6.3 ± 0.1	1.66% ± 0.51	0.678 ± 0.088	18.7 ± 3.1	3.53 ± 0.76	0.657 ± 0.253	70.2 ± 19.3

Accredited commercial and university labs were contracted to perform soil analysis for the field experiment and standardized methods followed whenever possible (overview of the methods is provided in Table S3). Total ΣDDX concentrations in the soil were measured using GC-MS according to (Rashid et al., 2010) and moisture content by thermogravimetry according to SS-EN 12880:2000. The following soil physicochemical parameters were measured: determination of pH in water (ISO 10390:2021); extraction and determination of Ca, K, Mg, and P content in soil using the AL-method (SS 028310:1993); determination of soil particle size distribution using the sieving and sedimentation method (ISO 11277); determination of total nitrogen (N-tot) after dry combustion (ISO 13878); determination of total organic carbon (TOC) after dry combustion (ISO 10694); determination of plant-available nitrate (NO_3) and nitrite (NO_2) content by flow analysis and spectrometric detection (ISO 13395); determination of maximum water holding capacity, WHC_{max} (ISO 11268-2:2012, Annex C). Microbial biomass was determined as microbial biomass carbon (MBC) according to the fumigation-extraction method (ISO 14240-2:1997). Microbial basal respiration (BR) was determined as O_2 consumption over five days by the use of the OxiTop® method (WTW GmbH; Platen and Wirtz, 1999), according to (ISO 16072:2002). A conversion factor of 0.9 was assumed for the respiratory quotient to convert O_2 consumption to release of CO_2 (Ben-Noah and Friedman, 2018). Potential nitrification was determined according to (ISO 15685:2012). The earthworm bioassay was conducted with the test species *Eisenia fetida* according to ISO 11268-1 (acute toxicity) and –2 (reproduction). Five of the living adult earthworms from each sample were randomly selected and removed on day 28, weighed individually, held on filter paper for 24 hours to purge guts, weighed after purging, and then placed in glass vials and frozen to use for measuring the DDX uptake into their biomass. The ΣDDX concentration in earthworms was analysed by modifying the method described in Henriksson et al. (2017) (complete method described in Paper V SM). The bait lamina (filled strips purchased from TerraProtecta Gmbh) field assessment was carried out according to ISO 18311:2016. See Paper V for more details.

3.5 Phytoextraction models

Phytoextraction modelling was carried out in Drenning et al. (2024) (Paper IV – *preliminarily accepted for publication*), which supported the fulfilment of research objective iv) *to develop simplified probabilistic models to estimate and clarify expectations regarding the time requirements for phytoextraction.* The general process followed to develop the simplified probabilistic models for phytoextraction is briefly described in this section, see Paper IV for more details.

Two probabilistic phytoextraction models for estimating time requirements for phytoextraction were developed in this study. They are based on existing analytical models that use simplified equations (Robinson et al., 2015, 2009, 2006, 2003) combined with empirical data derived from literature and the field experiment at Kolleberga. The two tested models considered either a) a linear steady-state extraction over time, or b) a first-order exponential decay function that would, theoretically, account for more complex soil chemistry and a decreasing pool of contaminants in the soil over time. Probabilistic modelling and Monte Carlo simulations are described in more detail in section 3.6.

See Paper IV for more details regarding the aggregation and comparison between the site-specific (Kolleberga) and literature-derived dataset as well as the different types of model output and scenario analysis by defining different model scenarios to test phytoextraction feasibility for different model assumptions, including different a) initial concentrations ($C_{soil,i}$); b) pumpkin stem bioaccumulation factors (BAF_{stem}); and c) pumpkin biomass production (*BMP*, $kg_{biomass,dw}$ /year).

3.5.1 Linear analytical model

The linear analytical models that are commonly used to provide an initial estimate of the time required for phytoextraction are based on a set of equations using empirical, easily acquired data. The standard equations vary somewhat between studies but in general are built on the assumption that the input variables are steady-state, i.e., plant uptake of contaminants and biomass production held constant over time. This results in a constant contaminant extraction potential (*E*), i.e., contaminant mass taken up per year, which can be used to calculate a mass balance for a certain amount of soil and estimate how many years are required to reduce the initial soil concentration to a final target level (Algreen et al., 2014; Grignet et al., 2020; Herzig et al., 2014; Robinson et al., 2015; Thijs et al., 2018). Robinson et al., (2015, 2009, 2006, 2003b) provided a series equations using a limited number of variables for calculating the total metal uptake over time, which are used here as a starting point and slightly modified to assess DDX phytoextraction.

The contaminant extraction potential, *E* (mg_{DDX}/year) is calculated based on the concentration of DDX in dry weight (dw) harvestable plant parts (i.e., stems and leaves), C_{plant} ($mg_{DDX}/kg_{biomass\ dw}$) and the dry weight biomass production per harvest, year in this case, *BMP* ($kg_{biomass\ dw}$/year):

$$E = C_{plant} * BMP \qquad \text{(Eq. 3)}$$

The concentration in the different harvestable pumpkin parts is calculated using the initial concentration of DDX in soil, $C_{soil,i}$ ($mg_{DDX}/kg_{soil\ dw}$), and their respective bioaccumulation factors, *BAF* ($mg_{DDX}/kg_{plant\ dw}$ / $mg_{DDX}/kg_{soil\ dw}$):

$$BAF = \frac{C_{plant}}{C_{soil}} \qquad \text{(Eq. 4)}$$

Consequently, the extraction potential *E* is:

$$E = \left(BAF_{stem} * C_{soil,i}\right) * BMP_{stem} + \left(BAF_{leaves} * C_{soil,i}\right) * BMP_{leaves} \qquad \text{(Eq. 5)}$$

Assuming *E* to be constant over time, the corresponding remediation time, t_{final} (years) required to reduce the initial contaminant concentration in soil to a final target level is calculated as a constant (linear) decrease over time:

$$t_{final} = \frac{m_{soil,i} - m_{soil,f}}{E} \qquad \text{(Eq. 6).}$$

where $m_{soil,i}$ and $m_{soil,f}$ are the total DDX mass in the soil (mg_{DDX}) at the starting point of the phytoextraction (initial mass, *i*) and at the point when reaching the final target concentration (final mass, *f*). The mass of DDX, m_{soil}, is calculated as:

$$m_{soil} = \rho * V * C_{soil} \qquad \text{(Eq. 7)}$$

where ρ is the soil bulk density (kg_{soil}/m^3) and V is the volume of soil (m^3) undergoing phytoextraction.

The removal rate, k (removal percentage/year) is then calculated as:

$$k = \frac{E}{m_{soil,i}} * 100 \qquad \text{(Eq. 8)}$$

The *BMP*, the treated soil volume (V), and the resulting mass of contaminants (m), and thus E, are in this case calculated for a unit area of 1 m^2, a depth of 0.35 m and a soil bulk density of 1 500 kg/m^3.

This linear analytical model with a constant extraction potential, E, and consequently a constant removal rate, k, does not account for variability or potential decreases in effectiveness and bioavailability over time (Robinson et al., 2015), and thus provides the theoretically shortest possible time for phytoextraction.

3.5.2 First-order exponential decay analytical model

Contaminant uptake is a function of the extractable contaminant mass in the soil, which would decrease over time thereby reducing effectiveness. This can be mathematically described using a first-order exponential decay function that could, at least theoretically, account for more complex soil chemistry and a decreasing pool of contaminants over time. Although the first-order decay model is commonly used to model biological degradation, it is important to note that the mechanism of reduction simulated here is only the removal by phytoextraction, not biological degradation. The analytical model becomes:

$$m_{soil}(t) = m_{soil,i} * e^{-k*t} \qquad \text{(Eq. 9)}$$

Where $m_{soil}(t)$ is the contaminant mass (mg_{DDX}) in soil at time t (years), $m_{soil,i}$ is the initial contaminant mass (mg_{DDX}) in soil, and k is the constant removal rate (%/year, Eq. 8). By rearranging Eq. 9, the corresponding remediation time, t_{final} (years), required to reduce the initial contaminant mass in soil to a final target mass, $m_{soil,f}$ (mg_{DDX}), is calculated accordingly:

$$t_{final} = \frac{ln\left(\frac{m_{soil,i}}{m_{soil,f}}\right)}{k} \qquad \text{(Eq. 10)}$$

3.6 Probabilistic modelling

Probability distributions and Monte Carlo simulations were used in both Drenning et al. (2023) (Paper III) and Drenning et al. (2024) (Paper IV – *preliminarily accepted for publication*) to account for uncertainties in relatively simple analytical models. The approach taken to incorporate probabilistic modelling in the different studies is described briefly in this section, see Paper III and IV for more details.

Both probabilistic models were set up in MS Excel using the Palisade add-in software @Risk 8.2 for defining probability distributions that represent uncertainties in the input variables (see respective papers for more details on specific input variables). Probability distributions were created for each variable using the aggregated (literature or empirical) datasets by assigning a Beta-PERT or (Log-)Normal distribution to the input data in the @Risk software based on minimum, maximum and most likely values, or mean and standard deviation for each input variable, respectively.

Monte Carlo simulations were run 10 000 times by repeatedly picking random values from the probability distributions of input variables as described by Bedford and Cooke (2001), which was used to calculate the resulting model output probability distributions: *NPVs* in the CBA and probable extraction potential, removal rates and time requirements to reach set soil target values in the phytoextraction model.

For investigating the sensitivity of the input variables in the probabilistic models, Spearman rank correlation coefficients were calculated by @Risk 8.2 to identify the variables that contributed most to the uncertainty in the model results.

3.7 Soil health assessment

Soil health assessment was carried out in Drenning et al. (2024) (Paper V – *submitted manuscript*), which supported the fulfilment of research objective v) t*o evaluate and demonstrate the potential impacts, positive or negative, of GRO on soil functioning and the delivery of ecosystem services.* Figure 3-4 gives an overview of the general process followed to perform the soil health assessment and the steps are briefly described in this section, see Paper IV for more details.

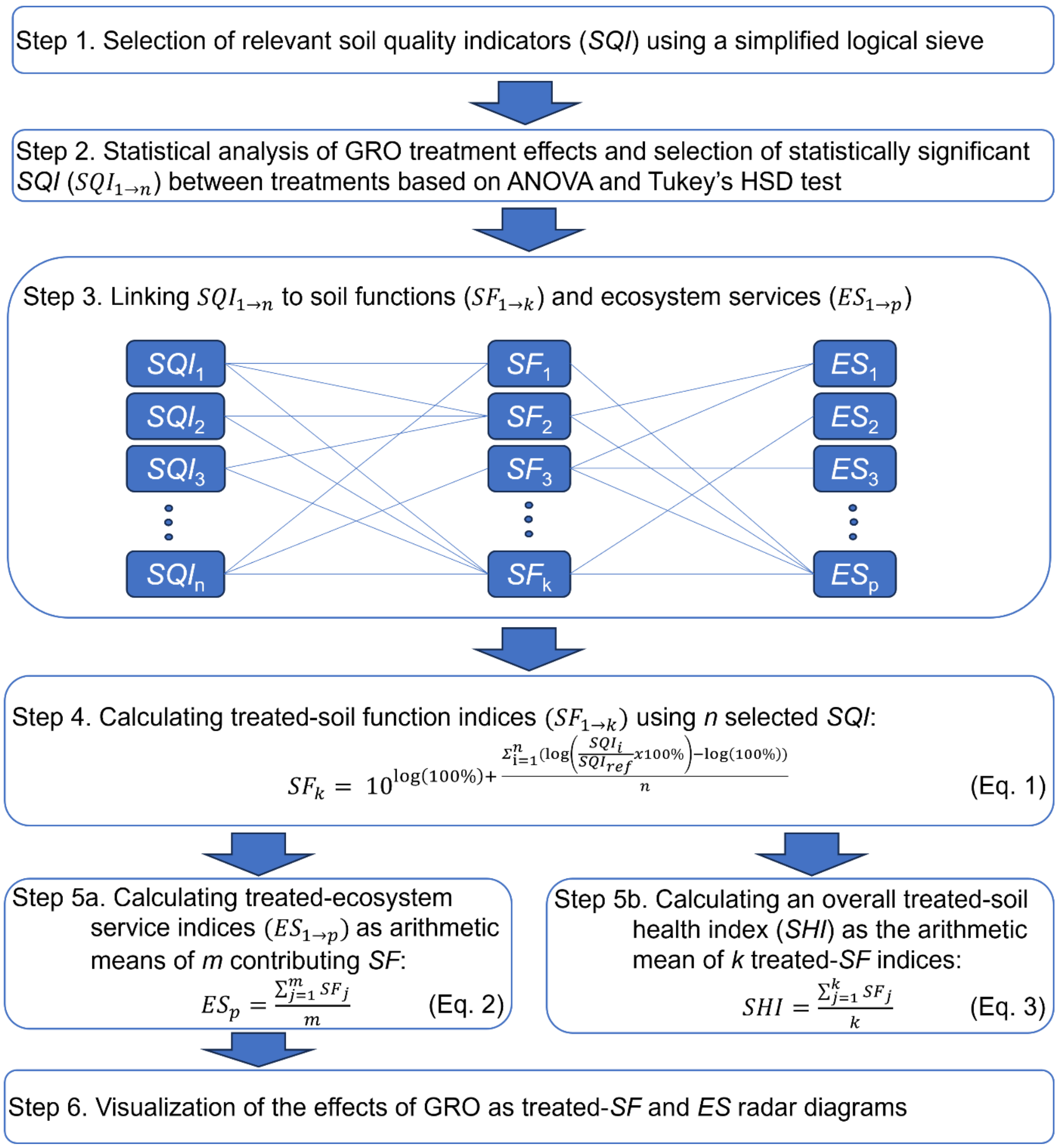

Figure 3-4. Schematic illustration of the methodology followed for evaluating the effects of GRO on soil health.

3.7.1 Selection of soil quality indicators via logical sieve

A gross list of soil quality indicators (*SQI*) was compiled through a non-exhaustive, narrative literature review, including findings from other major reviews and reports compiling the most commonly used indicators for soil monitoring programs (e.g., (Bünemann et al., 2018; Faber et al., 2013; Griffiths et al., 2016; Moebius-Clune et al., 2016; Pulleman et al., 2012; Ritz et al., 2009; Schindelbeck et al., 2008; Stone et al., 2016; Turbé et al., 2010). To filter the gross list of potential *SQI*, a simplified 'logical sieve', modified from Faber et al. (2013); Griffiths et al. (2016); Ritz et al. (2009); and Stone et al. (2016), was used whereby indicators are filtered according to criteria of accessibility, market availability, cost-effectiveness, ease of field sampling and standardization to determine those which are most suitable for the context to provide soils that are fit for purpose. *Availability in commercial laboratories* was particularly emphasized in this study as applied research with no closely connected research laboratory. A test battery of highest scoring indicators was then selected for use and complemented with other non-standard measurements for this particular case. See Paper V for more details and results of the simplified logical sieve.

3.7.2 Statistical data analysis

Analysis of variance (ANOVA) was used to statistically analyze the data and determine if the treatment effects of GRO were statistically significant ($p<0.05$) for individual *SQI*, where only those that were significantly different were included in calculating the treated index value for each soil function (*SF*). A correlation matrix was used to determine if there were significant correlations (Pearson's $r > 0.8$) between measured indicators to remove any redundancies (Obriot et al., 2016; Rinot et al., 2019). Using data from the second year of the experiment (Y2), the differences between GRO treatments' effects on the *SQI*, the treated-*SF* indices, and overall *SHI* were assessed by means of two-way ANOVA, with Plants (P) and Amendment (A, with or without biochar) as fixed factors and random effects of blocks in a split-plot model design. When significant results were obtained using ANOVA ($p<0.05$), Tukey's HSD test was used to make multiple pairwise comparisons and differences between groups deemed significant if $p < 0.05$. All statistical tests were performed using R statistical software v. 4.2.1 (R Core Team, 2022). See Paper V for more details.

3.7.3 Connecting indicators to soil functions and soil-based ecosystem services

A non-exhaustive narrative literature review was carried out to compile frequently proposed soil functions (*SF*), soil-based ecosystem services (*ES*), sub-functions, and processes. Redundancies, repetition, and overlap between definitions, processes, and terminology were aggregated and an overarching term and definition proposed if there is no clear consensus. This synthetic list of *SF* (Appendix A) and soil-based *ES* (Appendix B) was also tentatively matched with classes of the Common International Classification of Ecosystem Services CICES v5.1 (Haines-Young and Potschin, 2018) where there existed an equivalent service and gaps highlighted where there was not, as proposed by Paul et al. (2021). Using the EJP SIREN conceptual framework (Faber et al., 2022) as the theoretical basis, the *SQI* used in this study were grouped within the higher-level categories of *SF* based on correlations/associations between indicators and specific soil functions/processes. Based on prevailing literature, the *SF* were then linked to the specific soil-based *ES* to which the particular function contributes, which could be to one or several different services. The resulting hierarchical connections between the selected *SQI*, *SF* and soil-based *ES* (Figure 4-9) are used in the proposed soil health assessment.

3.7.4 Calculation of soil health indices

The treated-soil function indices to evaluate the effects of GRO on soil health were calculated using the index proposed by Epelde et al. (2014b), see Paper V for more details.

First, the effects of GRO on each *SF* were calculated as the log-transformed difference in each *SQI* for a specific treatment compared to a reference (experimental control) soil and an index value was calculated for each individual *SF*, based on the following equation (Epelde et al., 2014b):

$$SF_{1\to k} = 10^{\log(100\%)+\frac{\Sigma_{i=1}^{n}\left(\log\left(\frac{SQI_i}{SQI_{ref}}x100\%\right)-\log(100\%)\right)}{n}} \qquad \text{(Eq. 11)}$$

Where, SQI_{ref} corresponds to the mean value of the 'control' or reference (*ref*) soil for each indicator; SQI_i is the measured value for each treatment and replicate; *n* is the total number of *SQI* grouped within each *SF* used to calculate the index ($SF_{1\to k}$); the mean values of the reference soil are set to 100% to calculate the factorial deviation. See Paper V SM for more details and an example calculation.

Further, the relative change in soil health as the 'current capacity' of the soil to perform its functions were then used to derive trends regarding the eventual delivery of specific *ES* for human benefit. The arithmetic mean of the set of contributing *SF* for each soil-based *ES* ($ES_{1\to p}$) was calculated using Eq. 12 as treated-*ES* indices to give an indication of the expected change (positive, negative or no effect) compared to the reference soil:

$$ES_p = \frac{\sum_{j=1}^{m} SF_j}{m} \qquad \text{(Eq. 12)}$$

An overall treated-soil health index (*SHI*), which provides an integrated score of the effects of each GRO treatment on soil health compared to the reference soil, was calculated as the arithmetic mean of the treated-*SF* indices using Eq. 13:

$$SHI = \frac{\sum_{j=1}^{k} SF_j}{k} \qquad \text{(Eq. 13)}$$

The current study evaluates GRO treatment effects on specific SQI from the first two years of the experiment (Y1 & Y2). However, the *SHI* is calculated just using data from the Y2, since the dataset for Y1 was incomplete for the treatment soils.

A sensitivity analysis of the effects of GRO on *ES* was also carried out by using both the arithmetic mean and geometric mean, which prevents extreme values from overly influencing the resulting mean, and comparing the results to determine if there were changes.

4 RESULTS

This chapter summarizes the main results of the Ph.D.-thesis, including the added benefits of GRO for sustainable remediation and development (4.1); the GRO risk management framework, updates made since publication and application for the Kolleberga site (4.2); the cost-benefit analysis for PFAS remediation alternatives at the Stockholm Arlanda Airport case study site (4.3); probabilistic phytoextraction modelling applied for the Kolleberga site (4.4); and the effects of GRO on soil health based on the field experiment at Kolleberga (4.5).

4.1 Added benefits of GRO and connection to SDGs

As discussed in Drenning et al. (2020) (Paper I), brownfield sites represent important land and soil resources and provide significant opportunities in urban, peri-urban and even rural areas to meet national and international environmental goals. GRO as innovative remediation techniques and alternative land management strategies offer many direct and co-benefits in relation to sustainable remediation and development, some of which are summarized in Table 4-1.

Table 4-1. GRO strategies for sustainable remediation and development – summary of benefits. Icons are shown for SDGs identified in Drenning et al. (2020) and additional related SDGs are listed below these. Updated and modified from (Drenning, 2021a).

	Environmental	Economic	Social
GRO benefits	• Low-impact in-situ remediation of many contaminants • Mitigated environmental impacts of remediation and urban land use • Preserving/improving soil and land resources • Elements of green infrastructure • Provide/restore/maintain ecosystem services and soil function • Nature-based solutions • Resilience to climate change impacts and natural disasters • Carbon sequestration	• Significant cost savings (ca. 50%) compared to conventional remediation alternatives • Enables remediation of low- and moderate-risk contaminated sites at lower cost • Highly useful for sites of low land value or where conventional remediation techniques are unsuitable • Useful as part of a treatment train • Potential for profitable biomass production (e.g. bioenergy)	• Improved urban liveability • Allow sites to be used as formal greenspace and other 'soft' uses • Vegetated brownfields also provide informal recreational space and other citizen benefits • Aesthetically pleasing remediation • Potential part of landscape architecture toolkit
Related SDGs	3 GOOD HEALTH AND WELL-BEING; 11 SUSTAINABLE CITIES AND COMMUNITIES; 13 CLIMATE ACTION; 15 LIFE ON LAND		
	poverty elimination (1), zero hunger (2), good health and well-being (3), protecting children and women (5), supplying safe drinking water (6), increasing security and resilience of cities (11), mitigating climate change (13), preventing loss of aquatic biodiversity (14), preventing land degradation and loss of terrestrial biodiversity (15)		
Relevant political goals & EU directives	European Green New Deal, Biodiversity Plan to 2030, Soil Thematic Strategy, Circular Economy, Zero Pollution Action Plan, Directive for Soil Monitoring and Resilience		

4.2 A risk management framework for gentle remediation options (GRO)

As reported in Drenning et al. (2022) (Paper II), a risk management and communication framework was developed to clarify the connections between GRO, risk mitigation mechanisms and their impact on ecological and human health risks. The results are reported here in brief by presenting the resulting generic risk management framework for GRO, which has since been updated after its publication (Drenning et al., 2022), and inclusion in Drenning (2021a), based on feedback from various stakeholders during workshops and presentations. The site-specific application of the GRO framework is then presented for the Kolleberga field experiment site.

4.2.1 Generic risk management framework

A few updates have been made to the first version of the generic risk management framework based on comments and feedback received from stakeholders:

- The top box for human health exposure pathways now reads "Human health/animals". It was pointed out that many of the included exposure pathways can also be relevant for wild and grazing animals and were thus included to potentially account for these ecological receptors when relevant for a site.
- "Drinking water" as a potentially relevant human health exposure pathway was added.
- A dotted line connecting "Bioavailability/solubility reduction" to "Vapor inhalation" indicates that reducing bioavailability can also potentially mitigate volatilization of certain contaminants by altering the soil chemistry.
- "Surface water (and wetlands)" is now included in the bottom box for risk objects to include wetlands as a potential ecological receptor from contaminants at a site.
- In "Relative risk reduction time", all three contaminant group bars beside "Phytoextraction" have been modified to have a dotted outline and different shaded color to indicate that the necessary time for phytoextraction is highly variable and dependent on the contaminant (bioavailable) concentration and will not necessarily always "potentially take decades", or the longest possible time.
- The box for "Secondary effects by Vegetation cover" has been slightly modified to make it easier to read and "Managing receptor access (barrier)" was changed to clarify the meaning.

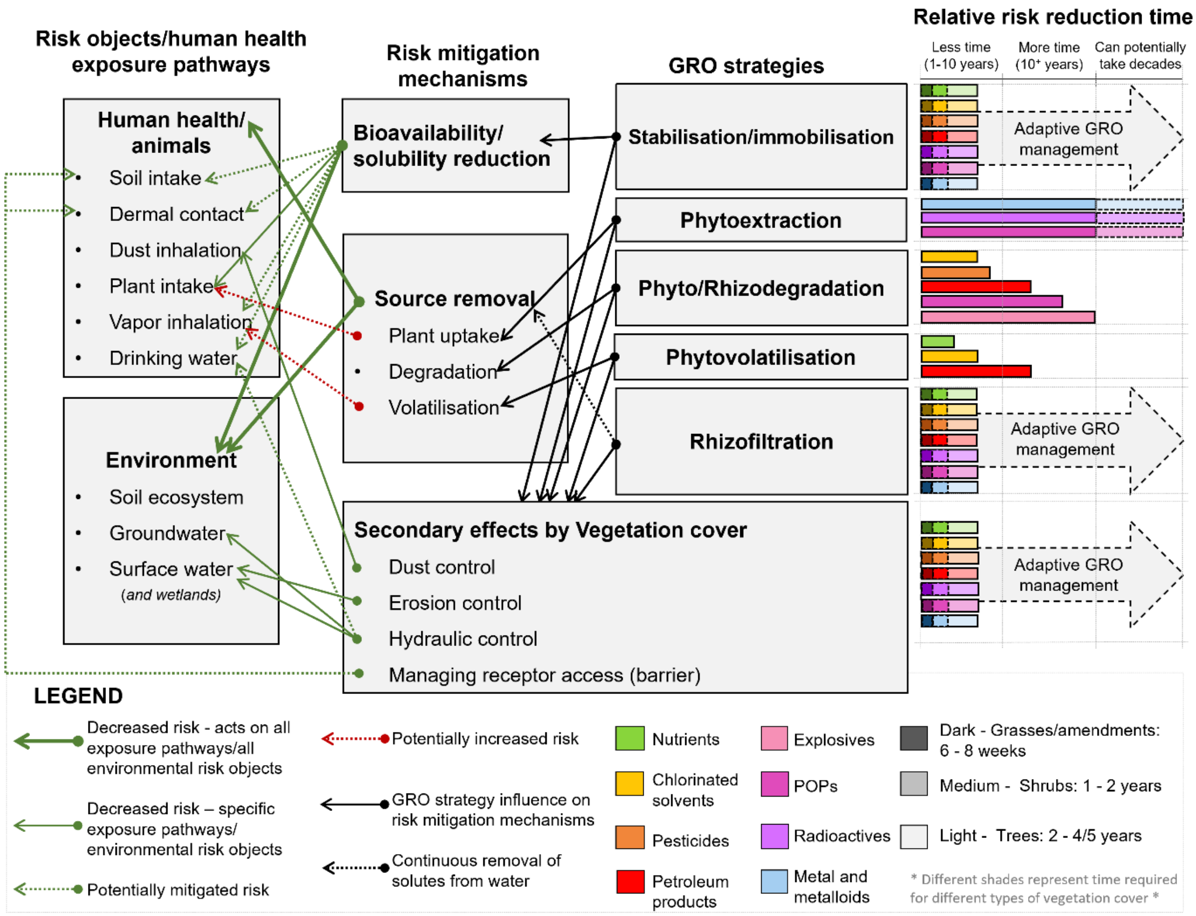

Figure 4-1. The updated generic risk management and communication framework for GRO with columns for Risk objects, Risk mitigation mechanisms, GRO strategies and a bar chart depicting relative risk reduction time for each GRO strategy. Relative risk reduction times are based on (Kennen and Kirkwood, 2015; OVAM, 2019). *Relative times for stabilisation/immobilisation, rhizofiltration and vegetation cover are based on literature. Adaptive GRO management is needed for all GRO strategies during their implementation, and includes long-term monitoring, watering, etc. for upkeep and to ensure the risk reduction is maintained over time. From* (Drenning et al., 2022).

4.2.2 Framework application for Kolleberga tree nursery

Given its current and expect future land use, the risk management framework has been applied at Kolleberga for only one green land use, a tree nursery, which is essentially equivalent to a Biofuel Park as modelled in the SEPA guideline value model (Figure 4-2). Instead of the SGVs created using generic assumptions, the site-specific guideline values generated by Tyréns (Nilsson, 2019) for DDT were incorporated into the framework to better account for site-specific risk conditions. The dominating human health exposure pathway(s), or most sensitive environmental receptor(s), per contaminant and land use are indicated and linked with the corresponding risk mitigation mechanisms and potential GRO strategies. The calculated risk quotients are shown in the figure, and a RQ > 1 indicates an elevated risk (i.e. above the site-specific SGV). Accordingly, DDT contamination in concentrations measured at the site can be deemed to pose a potential risk primarily to the environment (RQ = 7.25, primary receptor: soil ecosystem) and not human health (RQ = 0.45), which aligns with the risk assessment performed by Tyréns (Sandström et al., 2020). The GRO strategies that are identified to be able to mitigate the dominating exposure pathways are highlighted in green boxes in Figure 4-2.

The risks posed to the soil ecosystem are of primary concern and GRO strategies can be applied to mitigate this risk by 1) reducing the bioavailability and consequent exposure for soil organisms by using e.g. soil amendments, and 2) removing the source of the contamination by either phytoextraction or phyto-/-rhizodegradation. A combination of these strategies could reduce the exposure risks in the short-term (stabilisation/immobilisation) and/or achieve source removal in the longer term (extraction, degradation). The risk of DDT spreading to the groundwater could also be managed through the use of vegetation cover to limit infiltration of water through the soil profile thereby reducing leakage and providing hydraulic control. Thus, a combination of different GRO strategies, a multi-mechanism application, could be the optimal solution for Kolleberga. In its current state, the agricultural fields remain largely unused with no immediate plans for redevelopment other than re-use as a biofuel park or tree nursery (similar land use) at some point in the future. Therefore, gradual removal of the source term via extraction or degradation could be well-suited to this site since there is no time constraint and the risks are relatively low and feasibly managed using GRO. Cultivating crops with potential economic benefits could further improve the value proposition of phytomanagement at Kolleberga.

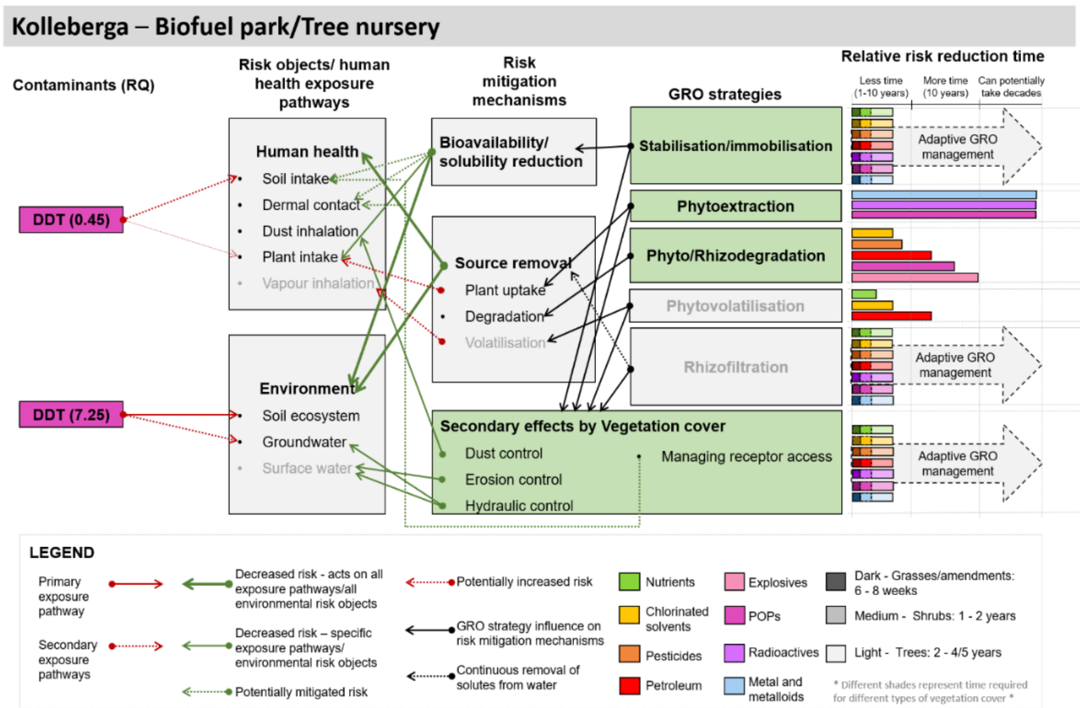

Figure 4-2. Site-specific application of the GRO risk management framework for the green land use of Biofuel park or Tree nursery. The contaminants detected at the site, Kolleberga, and risk quotients (RQ) are included in the furthest left column and are separated into exposure pathways for human health (above) or for the environment (below). Note: this is the original version of the GRO framework, from (Drenning, 2021a).

4.3 Cost-benefit analysis of PFAS remediation alternatives at Stockholm Arlanda Airport

This section will briefly present the main results from Paper III (Drenning et al., 2023) to compared PFAS soil remediation alternatives for the Stockholm Arlanda Airport case study site described previously in 3.3.1.

4.3.1 CBA results: the base scenario

The simulated mean present values of cost and benefit items for a discount rate of 3.5% and time horizon of 120 years are shown in Table 4-2. These values were used in the CBA to calculate *NPVs* for the respective PFAS remediation alternatives for both large and small spreading scenarios.

Table 4-2. Summary of cost and benefit values used in the cost-benefit analysis to calculate net present values. Mean PV: the mean present value of cost and benefit items. L: Large spreading scenario. S: Small spreading scenario. The annual avoided cost of inaction (B2-B3) in Alt 0-Alt 5 is assumed to be 7.5 MSEK. The social discount rate is 3.5%. The time horizon is 120 years. Note: the last row is corrected from the published version in (Drenning et al., 2023).

Time horizon (years): 120 / Discount rate: 3.5%		Alt 0 S/S hotspot & disposal rest		Alt 1 S/S hotspot & S/S rest		Alt 2 S/S hotspot & Stabilization AC		Alt 3 S/S hotspot & Phytoremediation		Alt 4 T/T hotspot & Phytoremediation		Alt 5 SW hotspot & Phytoremediation	
Category	**Item**	**Mean PV (MSEK)**		**Mean PV (MSEK)**		**Mean PV (MSEK)**		**Mean PV (MSEK)**		**Mean PV (MSEK)**		**Mean PV (MSEK)**	
Benefit categories and items													
	Spreading scenarios	*L*	*S*	*L*	*S*	*L*	*S*	*L*	*S*	*L*	*S*	*L*	*S*
B2-B3. Avoided cost of inaction	Improved health and increased provision of ecosystem services	197	197	197	197	197	197	177	177	177	177	177	177
Cost categories and items													
C1. Remediation costs	C1a-e.I. Short-term costs (total area)	109	60.0	80.4	52.1	67.5	48.6	129	75.9	322	268	129	75.3
	C1b.II,C1e.II. Long-term costs of management, monitoring (rest of the site)	0	0	0	0	7.39	7.39	7.39	7.39	7.39	7.39	7.39	7.39
	C1f. Project risks	32.1	17.3	23.4	14.9	19.6	13.9	16.8	13.1	74.4	70.8	16.6	12.9
	Total C1	141	77.3	104	67.0	94.5	69.8	154	96.4	404	346	153	95.6
C2. Impaired health due to remedial action	C2b. From transport activities and C2c. At a disposal site	0.709	0.201	0.0346	0.0157	0.0197	0.0116	0.00854	0.00852	0.401	0.400	0	0
C3. Decreased provision of ecosystem services due to remedial action (C3a-C3c)		82.5	10.1	79.3	9.23	6.57	3.68	2.58	2.58	32.3	32.3	4.76	4.76

The outcome of the probabilistic CBA model for the ‘base scenario’ is shown in Figure 4-3. Alt 2 (excavation and S/S of the hotspot and stabilization of PFAS at the rest of the site with activated carbon) generates the greatest mean *NPV* for both the large and small spreading scenarios, 123 MSEK and 14.1 MSEK, respectively. The results indicate that all studied remediation alternatives except for Alt 4 are associated with remediation cost savings (see Paper III SM) compared to Alt 0. This is valid for both spreading scenarios. The ranking of the other alternatives varies depending on the spreading scenario. For the small spreading scenario, Alt 1 and Alt 2 generate an almost equally positive mean *NPV*. Alt 3 and Alt 5 generate a slight negative mean *NPV* in the small spreading scenario but have the second highest mean *NPV* in the large spreading scenario. The mean *NPV* of Alt 4 is substantially negative in both spreading scenarios.

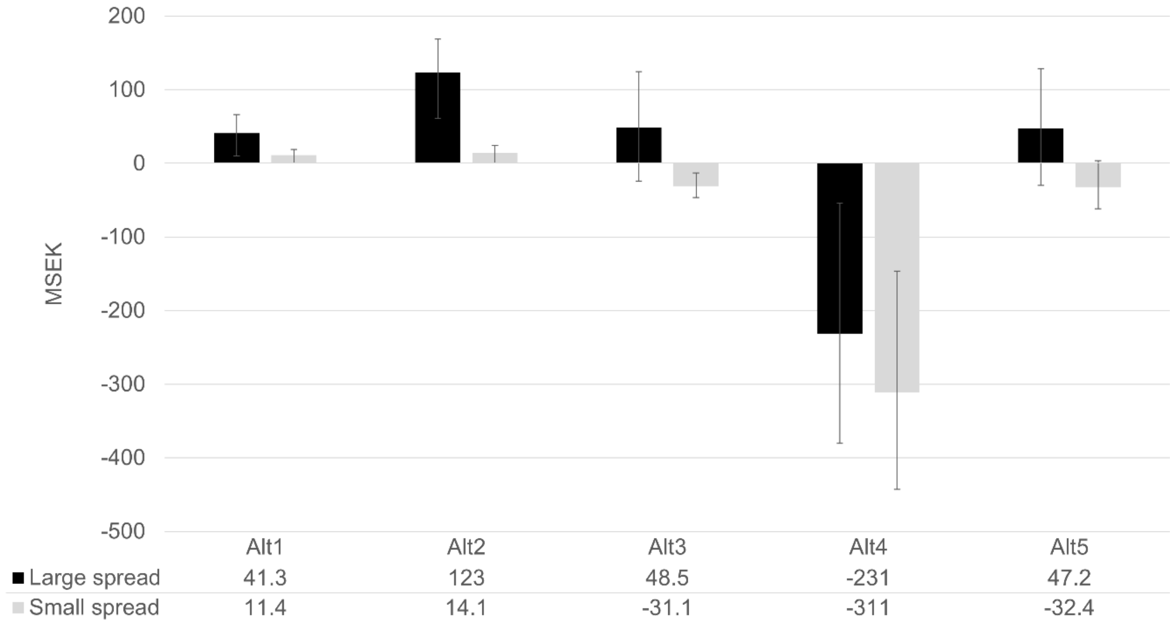

Figure 4-3. The simulated mean of the net present values (NPV) for Alt 1-5 in comparison to Alt 0 as the reference alternative; the 5^{th} and 95^{th} percentiles are shown as error bars. The values in the data table below the chart area represent the simulated mean values of the NPV for each alternative and spreading scenario in millions of SEK (MSEK).

An additional point of comparison for the remediation alternatives is the potential generation of reduced negative externalities (i.e., negative effects on health and the environment in terms of provisioning of ecosystem services, avoided carbon emissions, noise, traffic accidents etc.) as a result of the remedial action (Figure 4-4). The reference alternative (Alt 0) generates substantial negative externalities due to the remedial action, and any alternative that generates reduced negative externalities will therefore result in reduced costs (shown as a 'negative cost' in Table S4 in Paper III SM). In comparison to Alt 0, all alternatives, except for Alt 4 in the small spreading scenario, are associated with reduced negative externalities during the remedial action compared to Alt 0. Alt 1 is just slightly better than the reference Alt 0 with respect to externalities during remedial action. Alt 4 is even worse than the reference alternative in the small spread scenario because of more extensive air emissions and noise from the ex-situ thermal treatment of the hotspot. However, the externalities are associated with large uncertainties (shown as error bars in Figure 4-4) in the large spreading scenario in particular, and Alt 4 may generate even more negative externalities than Alt 0 in this spreading scenario.

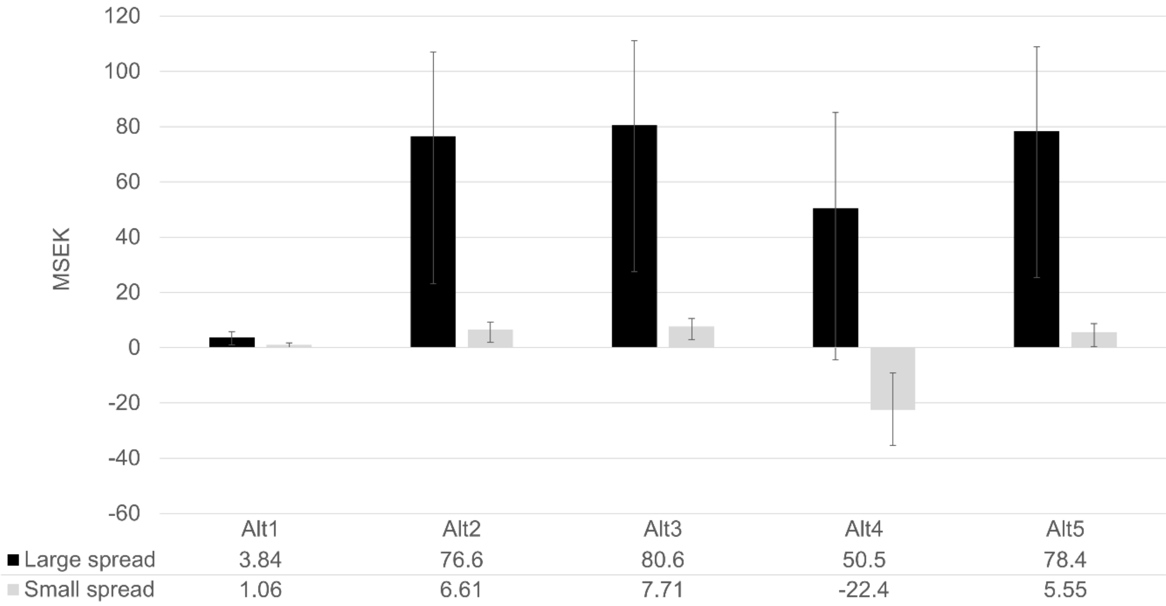

Figure 4-4. Cost reductions in terms of reduced negative externalities for remediation alternatives Alt 1-5, in comparison to Alt 0. The 5th and 95th percentiles are shown as error bars. The values in the data table below the chart area represent the simulated mean values of the reduced negative externalities for each alternative and spreading scenario in millions of SEK (MSEK).

4.3.2 Scenario analysis

When 'do nothing' is used as reference, all alternatives, including Alt 0 but excepting Alt 4, generate a positive mean *NPV* for the base scenario with an annual cost of inaction of 7.5 MSEK in the small spreading scenario, social discount rate of 3.5%, and time horizon of 120 years (Figure S13, SM). However, only Alt 2, Alt 3 and Alt 5 generate a positive mean *NPV* in the large spreading scenario, though with larger uncertainties for Alt 3 and Alt 5. The mean *NPV* for Alt 2 compared to the 'do nothing' alternative is the greatest for both spreading scenarios (95.4 MSEK and 123 MSEK for large and small spreading, respectively). All remediation alternatives generate negative externalities when compared to the 'Do nothing' reference alternative; however, the alternatives utilizing gentle remediation options (GRO) without thermal treatment (Alt 2, Alt 3, Alt 5), incurred the least negative externalities (Figure S14, in Paper III SM). Tables compiling the present values (*PV*) of each cost and benefit item as well as resulting mean *NPV* for each alternative compared to the 'do nothing' reference alternative for both large and small spreading scenarios are available in the SM (Table S6).

The sensitivity of the outcome of the CBA in relation to the 'annual avoided cost of inaction' (AACOI, i.e., the aggregated benefit of B2-B3) is investigated by identifying at which value of AACOI an alternative is socially profitable (*NPV* > 0) with at least 50% probability for either the large or small spreading scenarios (see Figure 4 in Paper III). This value is referred to as the 'breakeven point' and is an indication of when it is feasible to 'do something' with the currently available technologies given the potential benefit versus letting the problem get worse over time and wait for a better technology to arise to solve the issue.

Alt 2 has the lowest value of the breakeven point: an AACOI of approximately 7.5 and 5.75 MSEK for large and small spreading of PFAS, respectively. The difference in breakeven points between alternatives is clearly distinguishable in the large PFAS spreading scenario. Alt 2 is socially profitable with a very high probability (>90%) at an AACOI of ca. 9 MSEK, but the AACOI would have to be at least 12.5 MSEK/year to make Alt 1, Alt 3 and Alt 5 socially profitable with a probability >90% or 20 MSEK/year for Alt 0. In the small PFAS spreading scenario, all alternatives, including Alt 0 but excepting Alt 4, have similar breakeven points of avoided cost of inaction (ca. 5.5-7 MSEK) for generating an *NPV* > 0 (for details see Table S7 in Paper III SM). An avoided cost of inaction of at least 8 MSEK/year will generate a positive NPV for Alt 0, 1, 2, 3 and 5 with a probability of at least 85% in the small spreading scenario.

4.4 Time requirements for the phytoextraction of ΣDDX at Kolleberga

This section will present the main results of Paper IV (Drenning et al., 2024 – *preliminarily accepted for publication*) to estimate the time requirements for phytoextraction to clarify stakeholder expectations and phytoextraction feasibility at Kolleberga and similar sites.

4.4.1 Linear versus first-order exponential decay phytoextraction

The first-order exponential decay extraction model results in a much longer expected remediation time than the linear steady-state extraction model, which is shown in Figure 4-5 for the literature dataset with a $C_{soil,i}$ of 10 mg_{DDX}/kg_{dw}. The decrease in ΣDDX concentrations in soil is roughly similar during the first 20 years of phytoextraction but simulations with the first-order exponential model results in much less efficient removal soon thereafter as the ΣDDX pool diminishes. Also, the uncertainty intervals representing the 5th and 95th percentile of the probable percent decrease in ΣDDX concentration broaden over time indicating that there is greater uncertainty as to how the ΣDDX concentrations are expected to change in the long-term, especially for the linear analytical model.

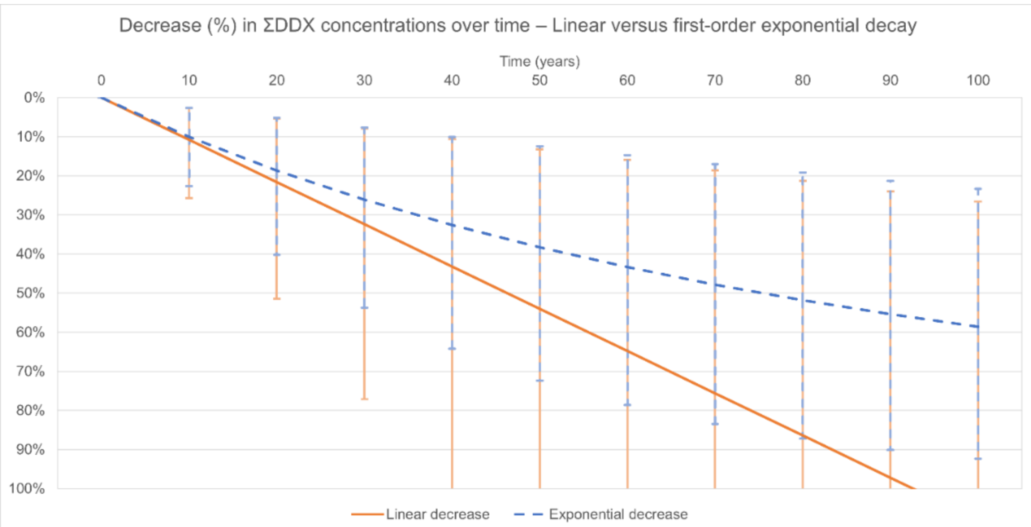

Figure 4-5. Steady-state linear extraction vs. first-order exponential decay extraction of ΣDDX from the soil by pumpkin (Cucurbita pepo ssp. pepo, cv. Howden), shown for literature data. Simulated results are mean values with the error bars representing the uncertainty interval [5th and 95th percentile], the solid orange line and error bars represents the linear model and the blue dashed line and error bars represents first-order exponential decay.

4.4.2 Time expectations and removal rates based on literature or site-specific data

The results from the probabilistic models, linear and the first-order exponential decay, and a comparative analysis between literature data and site-specific data for phytoextraction of ΣDDX with pumpkin are shown in Table 4-3 (and Figures S11, S13, S15, and S17 in Paper IV SM). Large differences between input data for *BAF* and *BMP* result in a substantially higher mostly likely (*mode*) simulated removal rate (k) of 0.606% per year when using literature data compared to the site-specific data, with a much lower most likely simulated removal rate of only 0.0127% per year. The models were applied to determine the expected time required to reduce the $C_{soil,i}$ of ΣDDX of 10 mg_{DDX}/kg_{dw} in the experimental plots to the regulatory soil guideline value in Sweden for less sensitive land uses of 1 mg_{DDX}/kg_{dw}, which requires a likely unachievable 90% reduction in soil ΣDDX concentrations. When using literature data, the most likely remediation time is 47.9 years with an uncertainty interval for the 5th and 95th percentile of 35.3 and 340 years (hereafter shown in brackets as [5th; 95th]), respectively, using the linear extraction model, or 123 years [90.3; 870] when using the first-order exponential decay model. For the experimental data from Kolleberga, predicted remediation time is much longer: approximately 3 570 years [2 280 ;16 400] for the linear steady-state extraction model or 9 120 years [5 840; 42 000] when using the first-order exponential decay model.

Table 4-3. ***Results from simulations of phytoextraction using linear or first order exponential models – comparison between literature and site-specific data and estimated time required to reduce soil ΣDDX concentrations from 10 to 1 mg/kg*** $_{dw}$ ***(≈90% reduction). Simulated results are the most likely value (mode), and the uncertainty interval [5th and 95th percentile] in brackets.***

Dataset	Removal rate, *k* (% per year)	Remediation time (years) Linear	First-order
Literature	0.606% [0.267; 2.51]	47.9 [35.3; 340]	123 [90.3; 870]
Site-specific	0.0127% [5.43 E^{-5}; 0.0392]	3 570 [2 280; 16 400]	9 120 [5 840; 42 000]

4.4.3 Phytoextraction of specific DDT metabolites

The *BAF* of *C. pepo* can differ between the different DDT metabolites, which has proven to be valid in the Kolleberga field experiment (Table 4-4), as indicated by the different *BAF* and resulting mean simulated removal rates for the metabolites. Despite the removal rate being higher for *p,p'*-DDE, it would still most likely take approximately 1 400 years to reduce the initial concentration by 90% according to the linear analytical model. Even using the maximum *BAF* for *p,p'*-DDE reported in literature of 18 (Eevers et al., 2018), a 90% reduction of the initial *p,p'*-DDE concentration would still most likely require a time of 214 years by linear extraction, all other things being equal.

Table 4-4. Varying effectiveness for different DDX – site-specific data. Simulated results are generated using the linear model and reported as most likely values (mode), expected remediation time is estimated for a 90% reduction for each metabolite.

	ΣDDX	*p,p'*-DDT	*o,p'*-DDT	p,p'-DDD	p,p'-DDE
Mean BAF_{stems}	0.890	0.718	2.67	1.63	1.08
Mean BAF_{leaves}	0.134	0.114	0.286	0.114	0.199
$C_{soil,i}$ (mg/kg $_{dw}$)	10	7.9	1.0	0.48	0.76
Extraction potential, *E* (mg/year)	0.743	0.436	0.236	0.0308	0.186
Removal rate, *k* (%/year)	0.0127%	0.0105%	0.0431%	0.0116%	0.0457%
Remediation time, t_{final} (years)	3 570	3 550	1 420	1 210	1 400

4.4.4 Scenario analysis

The average C_{soil} in the experimental plots at Kolleberga is ca. 10 mg_{DDX}/kg $_{dw}$, which corresponds with a 'high' concentration of ΣDDX according to similar studies and may also exceed a 'threshold' at which the effectiveness of pumpkin for phytoextraction diminishes significantly (Denyes et al., 2016; Lunney et al., 2004; Paul et al., 2015). By testing different $C_{soil,i}$ and adding an 'efficiency gradient' as a factor ($\%_{eff}$) to modify the extraction potential (*E*) the models can account for the likely negative correlation of phytoextraction performance with increasing soil ΣDDX concentrations. As shown in Table 4-5, the efficiency factor greatly impacts the resulting removal rates and expected remediation times (using the linear model) between 'high', 'moderate' and 'low' levels of ΣDDX contamination, with a most likely simulated removal rate of 0.22% (214-555 years), 0.66% (63-127 years) or 1.0% (24.6-34.0 years), respectively.

Table 4-5. Estimated differences in time requirements to reduce different $C_{soil,i}$ to target value of 1 mg_{DDX} kg $_{dw}$ for a 1 m^2 unit area –using literature data (input data in left box). Simulated results (right box) are most likely values (mode), and the uncertainty interval [5^{th} and 95^{th} percentile] in brackets.

Efficiency gradient		Initial soil ΣDDX			Remediation time, t_{final} (years)	
ΣDDX Level	Efficiency factor ($\%_{eff}$)	$C_{soil,i}$ (mg/kg $_{dw}$)	$m_{soil,i}$ (mg_{DDX})	Removal rate, *k* (% per year)	Linear	Exponential
High	33%	10	5250	0.219% [0.0900; 0.853]	214 [106; 1 000]	555 [276; 2 610]
	47%	9	4730	0.283% [0.126; 1.19]	150 [74.5; 704]	370 [184; 1 740]
	60%	8	4200	0.364% [0.162; 1.53]	115 [57.0; 539]	273 [136; 1 280]
	73%	7	3680	0.444% [0.198; 1.88]	92.0 [45.7; 432]	209 [104; 981]
	87%	6	3150	0.525% [0.234; 2.22]	75.7 [37.6; 356]	163 [80.8; 765]

Moderate	100%	5	2630	0.656% [0.270; 2.56]	63.0 [31.3; 296]	127 [62.9; 595]
	118%	4	2100	0.776% [0.311; 3.04]	48.2 [24.6;241]	89.1 [45.5; 445]
	136%	3	1580	0.895% [0.359; 3.51]	37.1 [19.0; 186]	61.2 [31.3; 308]
Low	155%	2	1050	1.01% [0.407; 3.98]	24.6 [12.6; 123]	34.0 [17.4; 170]

Results of simulations to determine the minimum required BAF_{stem} to reduce different $C_{soil,i}$ to the Swedish soil guideline value of 1 mg_{DDX}/kg_{dw} within 25 years (i.e., 'a reasonable timeframe' (Robinson et al., 2015)) are shown in Figure 4-6. The simulations were run using literature data for BAF_{leaves}, BMP_{stems} and BMP_{leaves} but testing different values for BAF_{stem}: 8, 10, 12, 14, 16, and 18 in the linear phytoextraction model. The simulations show that there is a <40% probability of reducing the 'high' $C_{soil,i}$ of 10 mg_{DDX}/kg_{dw} to the SGV within 25 years, even with the highest simulated BAF_{stem} value of 18. The probabilities remain consistent and increase slowly with decreasing ΣDDX concentrations until reaching the 'moderate' $C_{soil,i}$ of 5 mg_{DDX}/kg_{dw} where a BAF_{stem} of 18 and 16 have an approximately 50% and 38% probability to achieve a 90% reduction within 25 years, respectively. The probabilities increase more sharply approaching a 'low' $C_{soil,i}$, and a BAF_{stem} of 18, 16, 14, and 12 results in a >50% of achieving the 90% reduction target within 25 years for an $C_{soil,i}$ of 2 mg_{DDX}/kg_{dw}. The lowest tested BAF_{stem} of 8 and 10 have a <10% probability of achieving the 90% reduction target within 25 years for all concentrations above 4 mg_{DDX}/kg_{dw} and do not exceed a 20% or 40% probability, respectively, for even the lowest $C_{soil,i}$ of 2 mg_{DDX}/kg_{dw}.

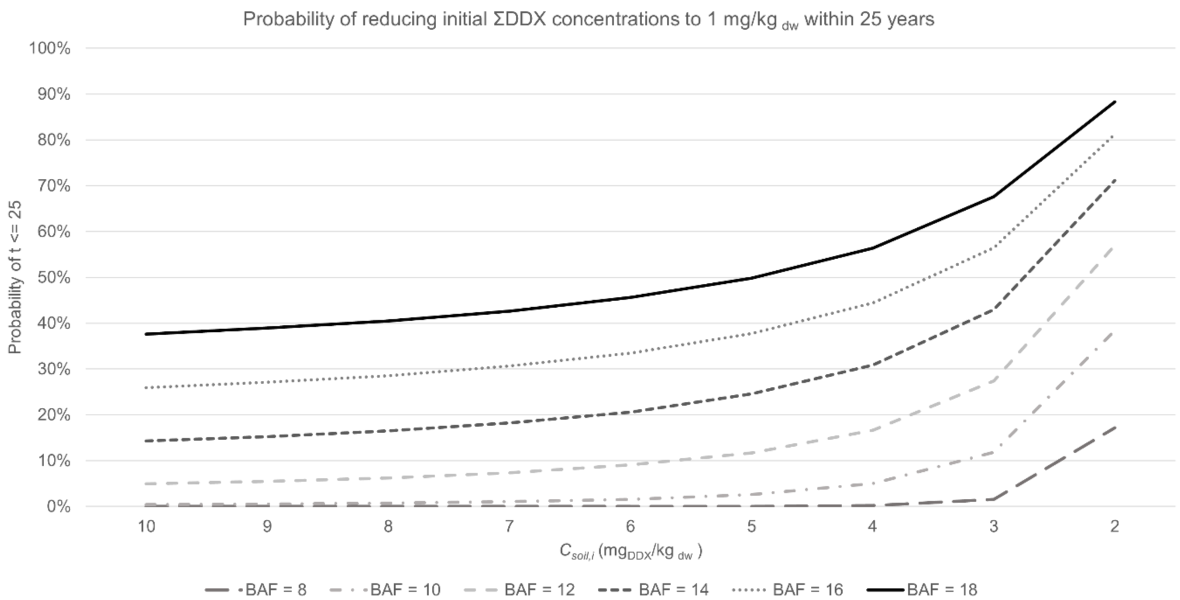

Figure 4-6. Simulations to determine what BAF is required to reduce initial C_{soil} to 1 mg kg^{-1} ΣDDX (corresponding to the soil guideline value for 'less sensitive land use' in Sweden) within 25 years for different initial C_{soil} , calculated using literature data for linear analytical model – BMP_{lit},). The simulated values are most likely (mode) values.

4.5 Effects of GRO on soil health

This section will present the main results Paper V (Drenning et al., 2024 – *submitted manuscript*) to develop and test a method to evaluate the effects of GRO on soil health using quantitative treated-soil health indices.

4.5.1 Selection of soil quality indicators and specific GRO treatment effects

The results of the simplified logical sieve to filter a shortlist of possible *SQI* are presented in Table S5 in the SM. A test battery of the highest scoring indicators was selected and used to evaluate the effects of GRO treatment on soil health. Statistical analysis of the data to determine the *SQI* where GRO treatment had statistically significant effects (according to ANOVA and Tukey's HSD test), and removing indicators with strong correlation, resulted in 6 physical and chemical indicators (Figure 4-7) and 6 biological indicators (Figure 4-8) for calculation of the treated-*SF* indices. See Paper V for more details and a longer discussion of specific GRO treatment effects.

Significant effects were observed from biochar ($p<0.01$) on WHC_{max} according to ANOVA, with a general increase by 4-30% compared to plots without biochar (Figure 4-7, Table S10), likely due to greater water retention and moisture content with the addition of organic carbon. Several studies report an increase in water retention, porosity, aeration and aggregate stability with the addition of biochar (Blanco-Canqui, 2021; Gul et al., 2015; Hou, 2021; Hou et al., 2023; Zhang et al., 2021) and effects may be particularly pronounced in sandy soils (Li et al., 2021; Razzaghi et al., 2020). However, the difference in WHC_{max} between T3 (ref. soil) and the biochar-amended soils was not statistically significant according to Tukey's HSD test, despite observed differences on group level. ANOVA showed a highly significant effect ($p<0.001$) of biochar on TOC, N-tot and available nutrients (P-AL, K-AL) compared to the unamended soils (Figure 4-7, Table S10), indicating that the fertility of the soil in terms of nutrient and carbon content is significantly improved by the addition of biochar. However, the effect from fertilization (particular for pumpkin and willow treatments) likely impacts these results.

Plant-available nitrite- and nitrate-nitrogen (NO_2+NO_3-N) in the soil was shown to be significantly ($p<0.05$) lower in the biochar-amended soils (47-61%), in line with other studies on effects of biochar produced in high temperature pyrolysis (Brtnicky et al., 2021; El-Naggar et al., 2019; Kookana et al., 2011; Lehmann et al., 2011; Zhang et al., 2021; Zhao et al., 2023) (Figure 4-7, Table S10). Nitrogen-fixing legumes (i.e., clover and alfalfa – T5 & T6) were used to counteract reduced N availability, which had a highly significant positive effect on both N-tot ($p<0.01$) and NO_2,NO_3-N ($p<0.001$) concentrations in the soil, increased by 473% (T5) and, respectively, 195% (T6) compared to T3 (ref. soil).

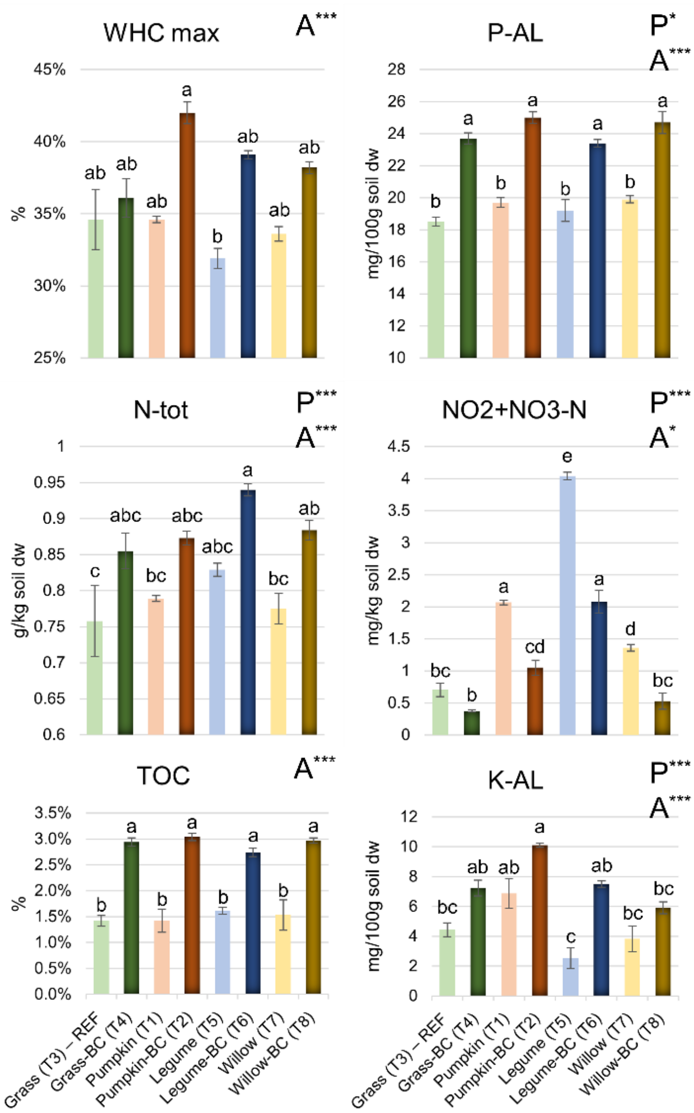

Figure 4-7. GRO treatment effects on selected physical and chemical soil quality indicators, data from the second year of the experiment (Y2). Data are mean values (n = 3), error bars are standard error. The bars with darker shading indicate treatments with biochar soil amendment. Probability values from two-way ANOVA shown in the top right corner for each SQI: p< 0.001(***), p< 0.01 (**), p<0.05 (*). Pairwise comparisons between groups were calculated using the Tukey HSD test and results shown using compact letter display: if two or more means share the same grouping letter, then they are not shown to be significantly different. TOC: total organic carbon (%$_{carbon}$); WHC$_{max}$: maximum water holding capacity (%$_{soil\ dw}$); N-tot: total organic nitrogen (g/kg$_{soil\ dw}$); NO_2+NO_3-N: sum of nitrite and nitrate-nitrogen (mg/kg$_{soil\ dw}$); P-AL: available phospohorous (mg/100 g$_{soil\ dw}$); K-AL: available potassium (mg/100 g$_{soil\ dw}$). REF: reference soil, experimental control.

Regarding biological indicators (Figure 4-8, Table S11), the results are more mixed with both significant positive and negative effects. The effect of biochar amendment (A) on the potential nitrification rate (PotNit) was significant (p<0.05) with a general increase in soils with biochar amendment, and the effect of legume mix plants (P) was highly significant (p<0.001) increasing PotNit by 111-117% compared to the T3 reference soil. Pumpkin treatments (T1, T2) also show significantly increased PotNit, but this is likely a result of fertilization rather than direct plant effects. Biochar amendment had a highly significant effect on BR (p<0.001) with an increase by 56-100% and grass (T4) and legume mix (T6) with biochar amendment had a significantly higher BR compared to T3 (ref. soil). Contrary to expectations, biochar amendment was observed to have no significant effect on microbial biomass carbon (MBC) between treatments and compared to T3 (ref. soil). The metabolic quotient (qCO_2) index (ratio BR:MBC) is significantly affected (p<0.001) by biochar amendment with a 33-133% increase (Table S13).

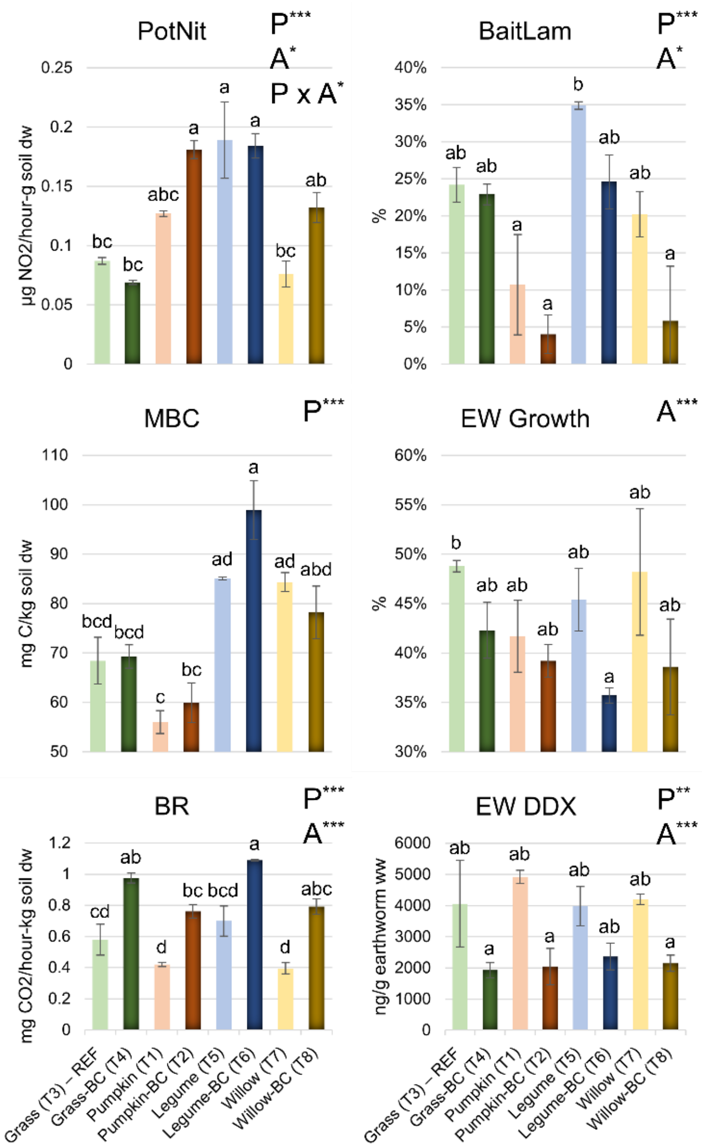

Figure 4-8. GRO treatment effects on biological soil quality indicators, data from the second year of the experiment (Y2). Data are mean values (n = 3), error bars are standard error. The bars with darker shading indicate treatments with biochar soil amendment. Probability values from two-way ANOVA shown in the top right corner for each SQI: p< 0.001(***), p< 0.01 (**), p<0.05 (*). Pairwise comparisons between groups were calculated using the Tukey HSD test and results shown using compact letter display: if two or more means share the same grouping letter, then they are not shown to be significantly different. BR: basal respiration (mg CO_2 / hour-kg$_{soil\ dw}$); MBC: microbial biomass carbon (mg C / kg$_{soil\ dw}$); PotNit: potential nitrification rate (µg NO_2/hour-g$_{soil\ dw}$); BaitLam: bait lamina (puncture count per strip); EW_{Growth}: earthworm growth; EW_{DDX}: earthworm DDX uptake into tissue (ng/g$_{earthworm\ ww}$). REF: reference soil, experimental control.

Regarding soil fauna, biochar amendment was observed to have a highly significant effect ($p<0.001$) on EW_{Growth} with a decrease of 6-27% compared to unamended soils, although Tukey's HSD test showed the group differences were only significant between the T3 reference soil and legumes with biochar (T6), for the decrease of 27% (Figure 4-8, Table S11). GRO effects on the feeding activity of soil meso- and macrofauna as measured using bait lamina (BaitLam) were highly variable but the effects of plants (P) were significant ($p<0.05$) as the legume mix was generally higher (Figure 4-8, Table S11). The effects of biochar amendment on BaitLam appeared to be broadly negative with a decrease of 5-71% compared to treatments without biochar but these results are not significant according to ANOVA, likely due to the large variability within groups. Similarly, no significant effects from GRO were observed for EW_{Rep} (Table S11), which could also be due to high variability within groups, and indicate that it may not be a sensitive indicator for assessing biochar effects on earthworms. These results seem to largely agree with the scientific consensus as many studies have shown that biochar (across different pyrolysis temperatures and application rates) can negatively impact soil fauna (e.g., earthworms, Collembola, enchytraeids, nematodes), which can include reduced feeding activity as measured with bait lamina (Marks et al., 2016; Prodana et al., 2021, 2019); demonstrating avoidance behaviour to certain types of biochar (Domene et al., 2015; Prodana et al., 2019; Tammeorg et al., 2014); reduced abundance, density, growth or reproduction of soil fauna (particularly earthworms) in biochar amended soils (Briones et al., 2020; Brtnicky et al., 2021; Conti et al., 2018; Liu et al., 2020; Zhao et al., 2023); and toxic effects to soil fauna from biochar, particularly at high pyrolysis temperature and application rates (Brtnicky et al., 2021; Gruss et al., 2019; Zhao et al., 2023). However, other studies have shown contradictory results with no clear negative effects from biochar on soil fauna (Bamminger et al., 2014; Brtnicky et al., 2021; Domene et al., 2014, 2015; Gruss et al., 2019; Honvault et al., 2023; Jeffery et al., 2022; Verheijen et al., 2009), and certain mesofauna (e.g., Collembola) may even benefit from biochar amendment (Briones et al., 2020; Gruss et al., 2019; Jeffery et al., 2022; Marks et al., 2014) especially if a biochar mixture had higher microbial biomass (Domene et al., 2015). In general, the effects from biochar vary and the type of biochar (temperature, ash content, particle size, feedstock, etc.) and application rate strongly influence its effects on soil fauna and their ability to perform their essential functions (Brtnicky et al., 2021; Lehmann et al., 2011; Marks et al., 2016; Prodana et al., 2019; Verheijen et al., 2009).

An important positive effect from biochar, particularly in the context of contaminated sites, is that it has a highly significant ($p<0.001$) effect on reducing the uptake of DDX in earthworm fatty tissue (EW_{DDX}), although Tukey's HSD test does not indicate these are significant group differences (Figure 4-8, Table S11). These data indicate that biochar could ameliorate the potential toxic pressure from DDX on soil organisms such as earthworms, in line with (Denyes et al., 2016; Wang et al., 2018).

4.5.2 Connecting soil quality indicators to soil functions and soil-based ecosystem services

Figure 4-9 presents a conceptualization of the hierarchical connections by which the selected *SQI* used to evaluate effects of GRO were grouped within the higher-level categories of soil functions (*SF*), which in turn underpin the delivery of soil-based ecosystem services (*ES*). The selected indicators were linked to five aggregated, well-established *SF* with a variety of constituent sub-functions and processes, included in this assessment: nutrient cycling and provisioning (*NCP*), water cycling and storage (*WCS*), pollutant attenuation and degradation (*PAD*), soil structure maintenance (*SSM*), and carbon cycling and storage (*CCS*) (Appendix A, Table S15). By compiling their principal sub-functions and processes, the aggregated *SF* were in turn connected to one or more relevant soil-based *ES* whose delivery they underpin (Appendix B, Table S16). Soil-based *ES* are primarily regulating services (Turbé et al., 2010), though the *SSM* function can still be considered relevant for the highly vegetation-dependent *ES-erosion control* and the provisioning *ES-biomass production* is linked to all the included *SF* as they pertain to the capacity of a soil to produce plant food, fibre and fuel for human use.

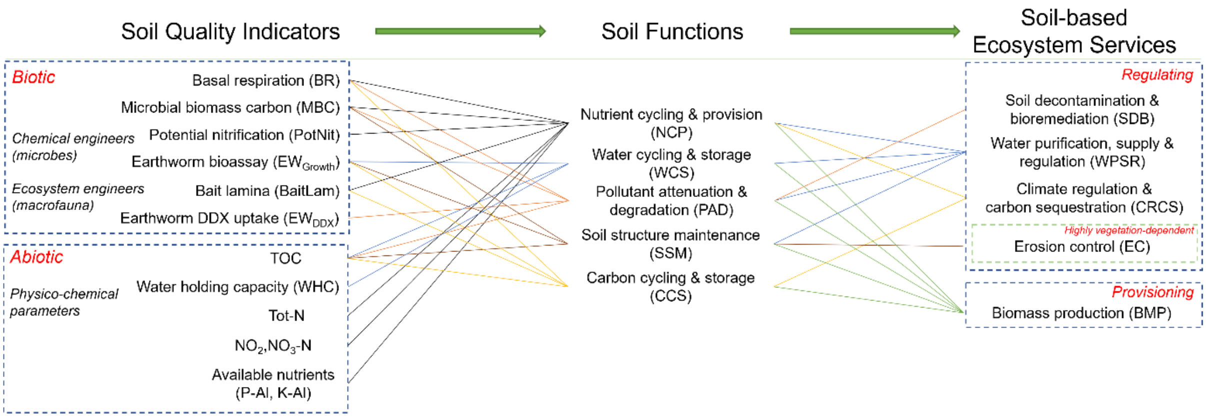

Figure 4-9. Conceptualization of the hierarchical connections between indicators, soil functions and soil-based ecosystem services.

4.5.3 The effects of GRO on soil health

The effects of GRO on soil health in Kolleberga compared to reference soil T3 (grass without biochar) were determined by calculating multiple treated-*SF* indices, presented in Table 4-6 (including results of ANOVA and Tukey's HSD test) and visualised in Figure 4-10A. The overall treated-soil health index (*SHI*) for each treatment is also shown in Table 4-6 and the treated *ES* indices are visualised in Figure 4-10B.

Broadly, biochar amendment has a significant positive effect on the resulting index values for *NCP* ($p<0.01$) and highly significant positive effects on *WCS*, *PAD*, *SSM* and the overall *SHI* ($p<0.001$). The effects of biochar on *CCS* show no clear statistical differences between groups with and without biochar. Plants are shown to have a significant effect on *SSM* ($p<0.05$) and highly significant effects on *NCP*, *CCS* and the overall *SHI* ($p<0.001$), with most of the positive differences associated with the legume mix (T5, T6) while grass (T3, T4) and willow (T7, T8) were more neutral, and pumpkin (T1, T2) was consistently the most negative. Many of the GRO treatments showed significant improvements in multiple treated-*SF* indices compared to the T3 reference soil (Table 4-6; Figure 4-10A): *NCP* – legumes both with (T6) and without (T5) biochar; *WCS* – pumpkin with biochar (T2) and grass with biochar (T4); *PAD* – all treatments with biochar (T2, T4, T6, T8); *SSM* – legumes with biochar (T6) and willow with biochar (T8); *CCS* – grass with biochar (T4) and legumes with biochar (T6). Overall, as an aggregated *SHI*, the treatments that are statistically different compared to the T3 reference soil are grass with biochar (T4) and legumes with biochar (T6). Of these, T6 has the higher comparative index value and this significant difference in soil health is due to the positive effects of both legumes and biochar on multiple *SQI*, which in turn results in an overall improvement in multiple *SF*.

Correspondingly, the treated-*ES* indices show that GRO treatment can have positive effects on multiple soil-based *ES* compared to the grass experimental control (T3) (Figure 4-10B). Importantly for the context of a contaminated site such as Kolleberga, the relative improvement is especially strong for *soil decontamination & bioremediation* (*SDB*) due to the positive effects of biochar to reduce DDX bioavailability (lowered EW_{DDX}) as well as improving overall microbial activity (BR) from biochar amendment and the legume mix. Large relative improvements can also be seen for *water purification, supply & regulation* (*WPSR*) and *erosion control* (EC) for all treatments with biochar due to significant improvements in *SQI* relating to *NCP*, *SSM* and *WCS*, e.g., large increase in organic carbon and available nutrients. However, the results are more mixed for *climate regulation & carbon sequestration* (*CRCS*), despite the large increase in TOC for increased carbon storage, and can be linked to variable effects of biochar amendment on soil fauna relating to carbon turnover. Biochar did not lead to significant improvements in the overall production function, *biomass production* (*BMP*), either. A sensitivity analysis to calculating either the arithmetic or geometric mean showed only minor differences in the treated-*SF* and *ES* indices that did not substantially change the results.

Table 4-6. Treated-soil function indices – using grass control as reference soil (mean value set to 100%). Data are mean values (n = 3) ± standard deviation; Probability values from two-way ANOVA shown below: p< 0.001(***), p< 0.01 (**), p<0.05 (*), n.s. = not significant. If significant differences were shown in ANOVA, pairwise comparisons between groups were calculated using the Tukey HSD test and results shown using compact letter display: if two or more means share the same grouping letter, then they are not shown to be significantly different. Significant differences from the reference (T3 – REF) are shown in bold. NCP: nutrient cycling and provision; WCS: water cycling and storage; PAD: pollutant attenuation and degradation; SSM: soil structure maintenance; CCS: carbon cycling and storage; SHI: overall treated-soil health index.

Treatment		NCP	WCS	PAD	SSM	CCS	SHI
Pumpkin (T1)		108% ± 10^{b}	95% ± 3^{d}	84% ± 9^{b}	89% ± 4^{c}	72% ± 9^{b}	89% ± 7^{d}
Pumpkin-BC (T2)		109% ± 4^{bc}	**128% ± 6^{a}**	**153% ± 23^{a}**	115% ± 5^{ab}	78% ± 3^{b}	116% ± 7^{bc}
Grass (T3) – REF		100% ± 3^{b}	100% ± 5^{bd}	100% ± 3^{b}	100% ± 3^{bc}	100% ± 2^{ab}	100% ± 1^{cd}
Grass-BC (T4)		105% ± 2^{bc}	**123% ± 1^{ac}**	**166% ± 7^{a}**	122% ± 2^{ab}	**128% ± 13^{a}**	**128% ± 1^{ab}**
Legume (T5)		**143% ± 1^{ac}**	99% ± 3^{bd}	115% ± 6^{b}	109% ± 4^{abc}	116% ± 2^{ab}	116% ± 1^{bc}
Legume-BC (T6)		**161% ± 2^{a}**	116% ± 15^{abcd}	**174% ± 12^{a}**	**126% ± 18^{a}**	**127% ± 10^{a}**	**141% ± 10^{a}**
Willow (T7)		98% ± 6^{b}	101% ± 3^{bcd}	97% ± 6^{b}	109% ± 1^{abc}	84% ± 13^{ab}	98% ± 3^{cd}
Willow-BC (T8)		87% ± 22^{b}	121% ± 5^{abc}	**159% ± 14^{a}**	**123% ± 5^{a}**	72% ± 32^{b}	112% ± 12^{bc}
ANOVA	Plant (P)	***	n.s.	n.s.	*	***	***
	Amendment (A)	**	***	***	***	n.s.	***
	P × A	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.

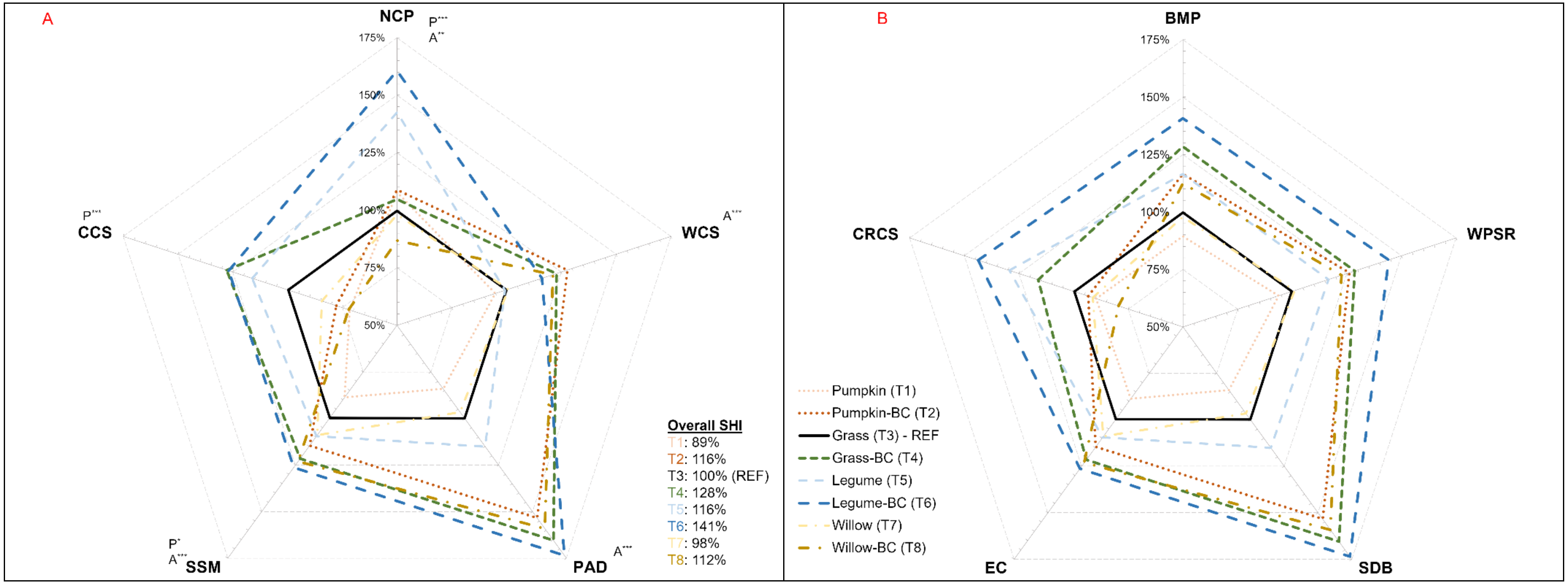

Figure 4-10. Radar diagrams showing the effects of GRO using: A) treated-soil function indices (NCP: nutrient cycling and provision; WCS: water cycling and storage; PAD: pollutant attenuation and degradation; SSM: soil structure maintenance; CCS: carbon cycling and storage) – with probability values from two-way ANOVA showing effects of plants (P) and biochar amendment (A) for each SF: $p < 0.001$(***), $p < 0.01$ (**), $p < 0.05$ (*); and B) treated-ecosystem service indices (BMP: biomass production; WPSR: water purification, supply and regulation; SDB: soil decontamination and bioremediation; EC: erosion control; CRCS: climate regulation and carbon sequestration). The legend is the same for both radar diagrams: the solid black line is the reference soil (T3) that has been set to 100% and darker, thicker lines indicate treatments where biochar was used as a soil amendment.

5 DISCUSSION

This chapter provides a summary discussion on the thesis output with a focus on GRO opportunity windows and added value, including sustainable remediation and development, risk management, economic analysis, time expectations, effects on soil health as well as possibilities and challenges connected to the generic workflow for contaminated land management in Sweden.

5.1 GRO for sustainable remediation and urban development

Brownfields are increasingly recognized as valuable land and soil resources, which have great potential, especially in urban areas, for sustainable remediation and redevelopment as part of a circular economy (Amenta and van Timmeren, 2018; Bardos et al., 2016; Breure et al., 2018; Chowdhury et al., 2020; Loures, 2015). Where commercially viable, brownfield land can be redeveloped for a 'hard' end use, e.g., industrial, commercial or residential purposes, but there is also a demand for regenerating such derelict or vacant land with low economic potential for 'soft' end uses such as green spaces (Bardos et al., 2016; Cundy et al., 2016; De Sousa, 2004; Doick et al., 2009, 2006; Masood and Russo, 2023; Olofsdotter et al., 2013). Indeed, vegetation-covered urban brownfields are important, but underappreciated, elements of urban green infrastructure that provide ecosystem services (Mathey et al., 2018, 2015). Sustainable management of our common soil and land resources to support soil functioning and delivery of ecosystem services is increasingly acknowledged as necessary to achieving the SDGs (FAO et al., 2020; Keesstra et al., 2016), for which NBS are identified as promising strategies (Keesstra et al., 2018b). In this regard, phytomanagement with GRO is a viable strategy for nature-based remediation and redevelopment of brownfields that can also provide many wider economic, environmental and social benefits such as ecosystem services (Burges et al., 2018; Cundy et al., 2016; Hou et al., 2023; Song et al., 2019). The potential of NBS for contaminated land management has been explored by many authors (Chowdhury et al., 2020; Hou et al., 2023; Masiero et al., 2022; Song et al., 2019), and NBS are shown to be viable for both remediation as well as the gradual conversion of a brownfield to a soft end use such as a greenspace.

The cost-effectiveness and economic benefits of phytomanagement are undoubtedly important for long-term sustainability; however, the wider environmental benefits generated in phytomanagement, especially at larger sites, are becoming increasingly salient in the modern context of widespread environmental degradation, biodiversity loss, rising sea levels, climate change and other challenges to meet the Sustainable Development Goals (Bardos et al., 2020a; Keesstra et al., 2018a, 2016; O'Connor et al., 2019). Also, when viewed in this broader context as a nature-based solution (NBS), phytomanagement may gain wider acceptance as a mainstream land management strategy for broader situational applicability to contribute to sustainable development (Bardos et al., 2020a; Cundy et al., 2016; Keesstra et al., 2018b; O'Connor et al., 2019; Song et al., 2019). Especially now, as we enter the UN Decade on Ecosystem Restoration, phytomanagement can play a valuable role in 'upgrading degraded land' and achieving the EU goal of 'degradation neutrality' to preserve and restore land and soil resources that provide critical ecosystem services.

5.2 Risk management with GRO

As shown in the risk management framework for GRO (Figure 4-1), there are several risk mitigation mechanisms through which GRO can be used to manage contaminants in soil. Often, the primary mechanism considered is source removal (and is the most desirable result from the perspective of stakeholders) but the feasibility depends strongly on the type of contaminant and site-specific conditions and may not be achievable within a reasonable timeframe. For organic contaminants, GRO degradation mechanisms have been shown to be effective for many contaminants (e.g., lightweight PAHs, BTEX, chlorinated solvents) and could reduce contaminant levels over a shorter time. However, effectiveness of phyto- and rhizodegradation is variable and depends on factors such as the type of organic compounds present, bioavailability, soil type, aging of contaminants, suitability and tolerance of plant species as well as broader concerns about generating recalcitrant or volatile breakdown products (Mench et al., 2010; OVAM, 2019; Vangronsveld et al., 2009). Regarding removal of inorganic contaminants, phytoextraction with the narrow focus of exclusively taking up metals as a stand-alone technology may rarely be suitable for strictly remediation purposes (Dickinson et al., 2009; Robinson et al., 2015, 2006; Van Nevel et al., 2007). However, alternative phytoextraction strategies like soil polishing (reducing marginally elevated concentrations to threshold levels) and bioavailable contaminant stripping (reducing the soluble, plant-available fraction of metals) are viable niche-solutions which could be more widely applicable at various scales and shorten remediation times from decades to just a few years (Dickinson et al., 2009; Gerhardt et al., 2017; Herzig et al., 2014; Mench et al., 2010; Robinson et al., 2015, 2009, 2006; Van Nevel et al., 2007; Vangronsveld et al., 2009). In addition, GRO strategies not intended for source removal such as phytostabilization or in-situ immobilization using soil amendments could also significantly reduce the bioavailability and solubility of (in)organic contaminants in a relatively short time. The vegetation cover itself controls erosion, dust and groundwater hydraulics to physically reduce the risk of spreading while also serving as a barrier to prevent receptors (humans) from accessing the soil (Cundy et al., 2016; GREENLAND, 2014a; Mench et al., 2010). There are, however, still uncertainties and limitations with GRO which must be considered in any potential application, such as the risk for uptake into plants that could transfer upwards into the food chain and managing contaminated biomass.

In addition, best practices to optimize and enhance phytomanagement of both inorganic and organic contaminants ought to be considered in any GRO application, including e.g., enhancing standard phytoremediation strategies with soil amendments and/or bacterial inoculates and mycorrhizal fungi, creating tree plantations based on short-rotation coppicing of woody plants such as poplar and willow, using high-biomass annual or perennial herbaceous species (e.g., rapeseed, sunflower, tobacco, bioenergy grasses, maize, etc.), and applying best practice agronomic techniques like crop rotations, intercropping with legumes, agroforestry, cover crops, etc. to improve phytoremediation effectiveness (Garbisu et al., 2019; Gómez-Sagasti et al., 2018; GREENLAND, 2014a, 2014b; Kidd et al., 2015; Mench et al., 2019; Moreira et al., 2021, 2019). There are a variety of applications to use GRO for 'situational risk management' (ITRC, 2009; Kennen and Kirkwood, 2015; OVAM, 2019). Such applications were previously synthesized in Drenning (2021a) and could be considered as templates for decision-support to identify the opportunity windows when considering how to design and apply GRO to manage site-specific risks. Facilitating discussion about contamination risks may also be possible to enable future, or concurrent, more sensitive land uses by applying GRO over time and re-evaluating the risks. Depending on the site conditions and land use, exposure risks like possible human exposure due to plant intake necessitates caution and more in-depth risk assessment before sensitive land uses are validated on contaminated sites.

While there are already several in-depth DST already developed for GRO (e.g., Andersson-Sköld et al., 2014; Cundy et al., 2015; ITRC, 2009; Onwubuya et al., 2009; OVAM, 2019), the risk management framework developed in Drenning et al. (2022) is intended to complement (not replace) these DST. The framework is intended to be used in the early stages of a brownfield redevelopment project to facilitate communication about risks with stakeholders as well as identify situations and conditions (i.e., opportunity windows) and relative timeframes where GRO would be feasible to manage the contaminant linkages at a particular site (such as at Kolleberga, Figure 4-2).

As previously discussed in (Drenning, 2021a), there are still concerns and knowledge gaps regarding GRO and the framework can help to address these. The framework was updated since its initial publication to include additional exposure pathways, protection targets and mitigation mechanisms in response to the specific requests and feedback from stakeholders and other experts from workshops (Table 3-1). Responses to the framework were generally positive – many said yes, they could use the framework – and validated that it could be a useful basis for discussion on the potential of GRO – e.g., "there are a lot of arrows in the framework, but it can be read and used anyway". There are, however, still difficulties with transferring knowledge about GRO and the information contained in the framework can be rather dense. To help with communication, a simplified 'communication version' of the GRO framework was developed to break down the necessary information into more easily accessible pieces of information to make understanding the framework easier. Indeed, this simplified version was requested by stakeholders (not all of whom were experts in CLM) along with additional guidance material for which fact sheets (Appendix C) were created to condense the necessary information to a few pages. Also, complementary databases of suitable plant species and good examples are desired. The relative risk reduction times are simplified generalizations but were considered helpful especially since time requirements are typically one of the primary concerns regarding potential GRO application. Further guidance has been requested by stakeholders regarding; for example, providing a range of RQ values within which GRO would be feasible for source removal via e.g., phytoextraction. That is, when are contaminant levels too high? This feeds into the broader issue of focusing on the effectiveness of GRO to reduce total concentrations of contaminants to generic guideline values instead of managing bioavailable concentrations and the actual, site-specific risks. One limitation is that the risk quotients (RQ) currently used for the site-specific framework application are based on total concentrations, but they could still provide a relative indication of the risk for each contaminant specific to its primary exposure pathway/protection target as a starting point to deeper discussions.

5.3 Economic analysis

The economic analysis using CBA for the soil PFAS remediation alternatives at Stockholm Arlanda Airport showed that each alternative entailed distinct advantages and disadvantages that affect its overall ranking (Figure 4-3, Table 4-2). The rankings in turn varied depending on the extent of the PFAS spreading and resulting size of the site to be managed. While the intensive hotspot remediation over a short period of time (2 years) generates much of the direct health and environmental benefits from the PFAS remediation, the alternatives which employed GRO for the 'rest of the site' generally resulted in a higher mean simulated *NPV* in comparison to the reference alternatives. The highest-ranking alternative in all scenarios (Alt 2) uses an activated carbon stabilizing agent for the rest of the site to achieve rapid risk reduction by mitigating PFAS spreading off-site but will require long-term project management and monitoring costs. Similarly, the next highest-ranking alternatives use phytoremediation for the rest of the site (Alt 3, Alt 5) to mitigate risks but entail long-term management and monitoring costs in addition to an added project risk cost to account for potential failure. Phytoextraction of the PFAS compounds in the soil is assumed to take a very long time, and may be effective only for short-chain PFAS with carbon chain length (<C6), but mitigation of spreading risks from short-chain PFAS through hydraulic control of groundwater and stabilization of long-chain PFAS in the roots via phytomanagement could be achieved in a shorter timeframe while also providing valuable ecosystem services (Evangelou and Robinson, 2022). However, spruce and birch may not be efficient PFAS phytoextractors, and recent reviews have indicated a range of species discovered as the field develops that could be more suitable (Evangelou and Robinson, 2022; Kavusi et al., 2023; Mayakaduwage et al., 2022; Shahsavari et al., 2021)

In general, GRO can provide added value to both the site owner (e.g., lower remediation costs) and to society (e.g., increased retention or delivery of ecosystem services) due to the reduced negative externalities associated with GRO compared to conventional techniques (Figure 4-4). An additional advantage with GRO is that it could potentially generate a greater mean *NPV* if accounting for long-term additional benefits such as provision of ecosystem services and the production of valuable biomass, which are currently not included. The longer timeframe for GRO may not even be such a disadvantage in the Stockholm Arlanda Airport case if there are no plans for rapidly redeveloping the site for immediate profit, especially in the large PFAS spreading scenario which supports the view that GRO are well-suited for large areas where there are no time restrictions (Cundy et al., 2016). Also, using GRO to manage the subareas of a larger site with lower contaminant risks (e.g., 'rest of site') in combination with intensive remediation of the hotspot is a promising opportunity window. GRO may even be a profitable option for both the problem owner and society in the long-term if the present value of remediation cost savings (benefit) exceeds the present value of postponed increased property value (cost) resulting from capital costs and long-term monitoring and management activities necessary to carry out the remediation alternative (Bell, 1996). However, if there are plans for immediate development then the long timeframe of remediation alternatives that include gentle remediation via stabilization with active carbon or phytomanagement (Alt 2, 3, 5) may be a disadvantage when compared to Alt 0.

Two different reference alternatives were included in Drenning et al. (2023) that serve different purposes: i) Alt 0 represents a modified 'business-as-usual' case entailing ‘total excavation’, which is the conventional remediation approach in Sweden and therefore useful as a comparison case; and ii) ‘Do nothing’, which is a helpful reference alternative in a CBA for obtaining indications on whether it is economically reasonable for society to spend scarce resources on remediating a particular site or rather use its resources for other purposes. The choice of discount rate is also important as it reflects the emphasis placed on future values – the higher the discount rate the lower the present value of the future benefits and costs, other things being equal (Johansson and Kriström, 2018) – which is important when valuing, for example, the expected, long-term positive externalities (or avoided damage). Furthermore, the choice of discount rate can become an issue of inter-generational equity, particularly in the case of PFAS with its large current and expected future impacts, and where the expected value of some remediation projects is long into the future and can only be accurately reflected in a CBA with a suitably low, long-term discount rate, or even a declining discount rate over a long time horizon (Johansson and Kriström, 2018; OMB, 2023). Indeed, part of the aim of economic analysis is to support decision-makers to allocate limited societal resources to projects where the benefits outweigh the costs to society and to answer the question of where it is better to do something (i.e., remediation) rather than do nothing. This is particularly salient in the case of widespread PFAS contamination, since the expected costs of remediation are massive and the available remediation technologies are still limited, and must be considered weighed against the even greater ‘costs of inaction’ of PFAS in the environment: at least €2.1-2.4 billion annually in the Nordics alone (Goldenman et al., 2019) Since the annual avoided cost of inaction (AACOI) at a particular site like Stockholm Arlanda Airport is highly uncertain, the simulations to determine the breakeven point (i.e., the magnitude of AACOI where a remediation alternative becomes socially profitable in comparison to doing nothing) can provide valuable information as an indication of when remediation may be justified from an economic standpoint and which alternative has the highest probability of $NPV > 0$.

There is an acknowledged need to clarify the expected costs and benefits of NBS, but, despite awareness of their multifunctionality, quantification of social and economic aspects and the distribution across stakeholders is still limited (EEA, 2023). Connecting economic valuation to soil health multifunctionality is also a recognized need and a clear framework would be useful for future valuation of soil-based *ES* (Löbmann et al., 2022). In the case of GRO, effectively capturing the added value of GRO to improve/retain soil functionality in monetary terms would be a substantial contribution to the value proposition offered by GRO. For instance, the ‘damage cost’ to the soil ecosystem avoided by using GRO, compared to conventional remediation techniques, could be a significant sum particularly where soil functionality is desired for soft end uses to provide ecosystem services such as green spaces. CBA is a tool that could be used to compare GRO to conventional soil remediation alternatives and show for which situations and conditions (i.e., opportunity windows) the use of GRO could result in a socially profitable project.

5.4 Phytoextraction time expectations

The exponential model, which accounts for a decreasing contaminant pool over time, resulted in most likely time estimates of 123 years using literature data or 9 120 years using site-specific data to reduce C_{soil} by 90% and is far beyond a 'reasonable timeframe' for phytoextraction of less than 10 or 25 years (Robinson et al., 2015; Vangronsveld et al., 2009). Thus, phytoextraction with pumpkin is not likely to be feasible at Kolleberga, or indeed similar sites, where the ΣDDX concentrations are above the threshold range of 5 – 10 $mg_{DDX}/kg_{soil\ dw}$ (Denyes et al., 2016; Paul et al., 2015) and the target value is based on a total concentration of 1 $mg_{DDX}/kg_{soil\ dw}$. As shown in the scenario "A" analysis, where different $C_{soil,i}$ and efficiency factors were used to increase or decrease the extraction potential, E (Table 4-5), the predicted remediation time was reduced from 555 to 127 years at a $C_{soil,i}$ of 10 or 5 mg_{DDX}/kg_{dw}, respectively, according to the first-order exponential decay model. The predicted timeframe is much shorter approaching 'low' DDX concentrations: as short as 34.0 years at 2 $mg_{DDX}/kg_{soil\ dw}$. The applied efficiency gradient is however a simplification that likely overestimates the time predictions since the efficiency would be expected to change over time and possibly improve at lower concentrations although the available DDX pool would also diminish as C_{soil} decreases over time.

Much research on phytoextraction of ΣDDX has focused on the uptake on *p,p'*-DDE in particular due to its tendency to bioaccumulate in human fatty tissue (Antignac et al., 2023; Beard, 2006) and usually being the most abundant and persistent degradation product of DDT at many sites (e.g., Eevers et al., 2018; Kelsey and White, 2005b; Wang et al., 2004; White, 2002, 2001; White et al., 2006a, 2005b). However, at Kolleberga, *p,p'*-DDT (77%) and *o,p'*-DDT (9%) are present in greater concentrations with a smaller proportion of *p,p'*-DDE (8%) and *p,p'*-DDD (4%). Further, the site-specific data showed a difference in *BAF* for different metabolites (BAF_{stem} lowest for *p,p'*-DDT, but >1 for *o,p'*-DDT, *p,p'*-DDE and *p,p'*-DDD, Table 4-4), indicating a potential for phytoextraction for certain metabolites but not the one which makes up the greatest proportion of the ΣDDX at Kolleberga. An aggregated *BAF* for ΣDDX may indeed not be truly representative of the total uptake and can differ substantially between sites with different ΣDDX compositions.

The simulations with different values for *BAF* and *BMP* aimed to determine the necessary effectiveness for phytoextraction at Kolleberga with pumpkin to be feasible (Figure 4-6). To improve the prospects of phytoextraction of ΣDDX at Kolleberga, the removal rate would need to be greatly increased. This could be done through enhancing pumpkin's *BAF*, which has been done successfully by using biosurfactants such as *Pseudomonas* spp. (Wang et al., 2017; White et al., 2006b), mycorrhizal fungi (White et al., 2006c, 2006b; Whitfield Åslund et al., 2010), bioaugmentation with endophytic bacteria (Eevers et al., 2018), earthworms (Kelsey and White, 2005b), and chemical surfactants or organic acids (White et al., 2007, 2003; Whitfield Åslund et al., 2010). The maximum tested BAF_{stem} in the models was 18 for ΣDDX and is likely unattainable; however, Eevers et al., (2018) achieved a BAF_{stem} of 18 for the metabolite *p,p'*-DDE (C_{soil} of ca. 0.15 mg/kg_{dw}) in their study by inoculating zucchini (*C. pepo* ssp. *pepo* cv. Raven) with a consortium of DDE-degrading endophytes derived from zucchini, which the authors suggest improves phytoextraction's feasibility by improving plant growth and overall *p,p'*-DDE removal by promoting biological degradation. Similarly, various agronomic practices have been tested to improve the *BMP* of *C. pepo* for phytoextraction of DDT (Denyes et al., 2016; Lunney et al., 2010; White et al., 2005a; Whitfield Åslund et al., 2010). Using high biomass producing species and further improving the amount of produced biomass through the use of organic soil amendments, microbial amendments such as mycorrhizal fungi, and other agronomic practices is a widely accepted strategy to improve *BMP* and thus the effectiveness of phytoextraction (Kidd et al., 2015; Mench et al., 2010; Vangronsveld et al., 2009).

DDX bioavailability must be taken into considering since the main risks from contaminants to humans and ecological receptors is dependent on the bioavailable fraction of contaminants and not on the total content of contaminants in the soil, much of which may be inaccessible to humans and other living organisms due to soil conditions and aging processes (Herzig et al., 2014; Kumpiene et al., 2014, 2009; Vangronsveld et al., 2009). For example, the bioaccessibility of aged DDT may even be less than 4% of the total concentration (Smith et al., 2012). Also, studies have successfully demonstrated that the bioavailable ΣDDX concentrations and uptake into earthworms could be significantly reduced by using either biochar or activated carbon amendments, which is an alternative GRO strategy to manage environmental risks at DDX-contaminated sites (Denyes et al., 2016; Wang et al., 2018). In addition, it is likely that the pool of readily available ΣDDX at Kolleberga and similar sites, may decrease over time due to aging effects that will immobilize the contaminant in the soil matrix and make it "permanently" inaccessible to soil organisms or plant roots. Thus, it may be impossible for plants to remove the entire total concentration of ΣDDX from the soil and achieve the 90% reduction target. However, from a risk perspective, this may be beneficial, especially if the main risks are related to the soil ecosystem and to bioaccumulation in the food chain.

Given phytoextraction's inherent limitations and inefficiencies, it is reasonable to ask what to expect with phytoextraction, what is possible to achieve, and for which situations would it be feasible? Indeed, the first question asked of phytoextraction is often "how long time does it take?" In general, due to the excessive time requirements to achieve reduction targets, many authors consider phytoextraction to be infeasible in most cases, especially if national regulation is based on total soil contaminant concentrations instead of bioavailable concentrations (Dickinson et al., 2009; Mertens et al., 2005; Neaman et al., 2020; Robinson et al., 2015; Santa-Cruz et al., 2022; Van Nevel et al., 2007). The opportunity windows for phytoextraction are most likely for low contaminant concentrations (only slightly exceeding soil guideline values) that are readily bioavailable for effective extraction and bioaccumulation by plants. There are, however, still obstacles and uncertainties regarding replenishment of bioavailable pools and acceptance by regulatory agencies (Neaman et al., 2020; Santa-Cruz et al., 2022; Thijs et al., 2018). Estimating the time required for phytoextraction, which can potentially take up to a few decades, is thus a critical aspect of determining the feasibility of phytoextraction. Although the analytical probabilistic models are simplified, they can be useful as a practical tool to provide an initial estimation of the remediation time required for phytoextraction at a particular site, including uncertainties, and could complement more generalized approximations such as those proposed in Drenning et al., (2022).

5.5 Improvement of soil health with GRO

Overall, the results of the treated-*ES* indices indicate where there could be potential synergies in GRO treatment to improve the multifunctionality of the soil to provide multiple *ES*. Indeed, a main, frequently-cited advantage of GRO is the potential for multifunctionality: to potentially both manage risks and improve (or at least reduce) soil functionality to provide *ES* (Burges et al., 2018; Cundy et al., 2016; Drenning et al., 2022). While there can be site-specific differences and uncertainty in GRO's effectiveness to reduce total contaminant concentrations, many studies have corroborated these results that GRO can indeed have positive effects on soil health as measured using a variety of *SQI*, e.g., (Anza et al., 2019; Burges et al., 2016, 2017, 2018, 2020, 2021; Epelde et al., 2008; Foucault et al., 2013; Gajić et al., 2018; Garaiyurrebaso et al., 2017; Gómez-sagasti et al., 2012; Gómez-Sagasti et al., 2021; Kumpiene et al., 2009; Lacalle et al., 2018; Mench et al., 2018; Quintela-Sabarís et al., 2017; Touceda-González et al., 2017). The results of the treated-*SF* and *ES* indices and overall *SHI* presented in this study also align with several studies that have utilized a soil health/quality index to demonstrate the positive effects of biochar (Bera et al., 2016; Carnier et al., 2023), phytoremediation (Barrutia et al., 2011; Burges et al., 2017, 2016; Mench et al., 2022), and a combination of these methods (Yadav et al., 2023).

Regarding biochar specifically, there is a large body of literature suggesting that biochar can have highly positive impacts on soil health and improve SF to provide multiple ES such as climate regulation (including reducing N2O emission, N leaching and runoff) and carbon sequestration through carbon storage with a long residence time (potentially hundreds to thousands of years (Gul et al., 2015; Kuzyakov et al., 2009)), improving biodiversity and habitat, soil fertility, biomass production, and others (Blanco-Canqui, 2021; Bolan et al., 2021; He et al., 2021; Hou, 2021). Less fertile, degraded and contaminated sandy soils of marginal quality may especially benefit from biochar amendment (Bekchanova et al., 2021; Tang et al., 2013). This seems to be true for the loamy sandy soil at Kolleberga, where the biochar amendment is shown to stimulate the soil microbes and increase overall activity, which may be due to additional nutrient availability and retention, a small pool of labile C as well as improved soil pH, porosity, aeration and water retention that provides a favorable soil environment for microbes (El-Naggar et al., 2019; Lehmann et al., 2011).

Biochar has also been shown to be an environmentally sustainable alternative remediation technique to conventional methods (Papageorgiou et al., 2021). There are, however, contradictory or mixed results regarding biochar's improvement on *ES* in some studies and results may depend on the type and application of biochar as well as soil conditions. For example, while biochar is shown here to generally improve *erosion control* (*EC*), erosion rates can potentially increase from sandy soils especially when biochar is applied to the soil surface (Brtnicky et al., 2021). Similarly, the effects of biochar on *biomass production (BMP)* also varies between studies, which can be a result of the immobilization of plant-available forms of nitrogen, but is generally considered to improve overall crop production in most cases (Brtnicky et al., 2021). It is also important to account for other potential drawbacks to biochar amendment such as nutrient immobilization, reduced efficacy of agrochemicals and ecological risks from biochar amendment due to potential toxic effects on different groups of soil organisms such as earthworms (Brtnicky et al., 2021; Zhao et al., 2023). Trade-offs between *ES* are also possible (Blanco-Canqui, 2021), which may not be evident in these results. For example, *SDB* may be favored by higher application rates of high temperature biochar due to increased sorption capacity to manage contaminants but possibly does not improve *BMP* as much as lower temperature biochar at a lower application rates. Also, there are uncertainties in the results as suggested by the standard deviations for both individual *SQI* (Table S10 and S11) and resulting *SF* indices (Table 4-6), which can be large and indicate high variability in the data resulting from differential treatment effects, soil heterogeneity, or other sampling effects.

The soil health assessment method followed here is not fully comprehensive but is an attempt to develop a systematic method with which to account for the potential positive/negative impacts of remediation alternatives on soil health. As argued by Smith et al. (2015), while there are still important knowledge gaps and more fundamental research is needed, there is enough knowledge to start moving in the right direction and implement best practices to both improve the delivery of and raise awareness about the valuable ecosystem services underpinned by soils and the natural capital they provide.

The multifunctionality of soils and their contribution to providing multiple *ES* is still not fully accounted for in many *ES* assessments or ontologies such as the Common International Classification of Ecosystem Services, CICES (Faber et al., 2022; Haines-Young and Potschin, 2018; Paul et al., 2021), see Appendix B for potential matches with CICES v5.1 classes. This work aims to address this gap and provide decision-makers with an indication of the relative change in soil health and the added value of GRO treatment for restoring contaminated soils to provide *ES* for human benefit. For, the ultimate objective of a risk-based and sustainable remediation process must be not only to remove the contaminants from the soils (or instead break contaminant linkages) but also to restore soil health (Epelde et al., 2008; FAO et al., 2020; Gómez-sagasti et al., 2012). Grouping individual, correlated *SQI* into higher-level categories such as *SF* and *ES* can facilitate interpretation of laboratory data for soil health assessments, improve communication with stakeholders as well as provide long-term monitoring programs with the ability to adapt through time against changes in techniques, methods, interests, etc. (Burges et al., 2018; Epelde et al., 2014a, 2014b; Faber et al., 2013; Garbisu et al., 2011; Gómez-sagasti et al., 2012). While the overall *SHI* may not provide specific information regarding the improvement/diminishment of individual *SF* or *ES*, it can provide a simple indication of the aggregated effects of the GRO treatment that could useful when communicating with stakeholders. Indeed, developing practical assessment methods, interpreting data, and facilitating communication about soil health with stakeholders are important objectives in this work. For instance, in Sweden, practitioners have reported that many aspects of soil health assessment, including measuring biological indicators, still belonged primarily to the scientific realm and were not practically applicable (Faber et al., 2022). There are, however, important limitations and assumptions made in developing and applying the method which must be taken into consideration and improved in further iterations of soil health assessment at contaminated sites, which are discussed in more detail in Paper V.

The connections between *SQI*, *SF* and *ES* were based on the prevailing scientific literature, e.g., (El Mujtar et al., 2019; Faber et al., 2022; Kibblewhite et al., 2008), but in some cases the linkages made here may not be clear or sufficiently well-supported. An additional challenge was whether to separate the ‘biotic’ from the ‘abiotic’ component of soil with regards to both *SQI* and *ES*, as has been done in CICES v.5.1 (Haines-Young and Potschin, 2018). Some experts propose that *ES* are outputs of the biological component of the soil ecosystem, i.e., soil biota as ‘service providing units’ providing *ES* through quantitative ‘ecological production functions’ (Faber et al., 2022, 2021; Munns et al., 2015), but other authors maintain that the dichotomy between biotic and abiotic soil services is confusing and that abiotic flows should be an inherent part of *ES* (Fox et al., 2020; Meulen et al., 2016; Meulen and Maring, 2018). Further, the *SF* and *ES* included in this soil health assessment are not exhaustive, and additional or different categories could be included. For example, an important sixth soil function *biodiversity and habitat* was identified in the literature review (Appendix B) but not included in this assessment since the *SQI* used here were not considered to be relevant to this function. Similarly, there are numerous ontologies for *ES*, and specifically soil-based *ES*, which have different names or additional *ES* that could be included in an *ES* assessment (e.g., local climate regulation, noise abatement, recreational and aesthetic cultural services) but were either not easily accommodated into this soil health assessment or outside the scope of this study The demand or prioritization of *SF* and *ES* may differ depending on the type of soil and land use as well as stakeholder preferences. Considering Kolleberga, all *SF* and *ES* are currently weighted equally but given that it is agricultural land, and the planned future land use is as a tree nursery, the site owner’s primary interest is likely to ensure that the soil is fit for *biomass production* (*BMP*) while also managing the DDX contamination. *BMP* is here linked to all *SF* and an overall improvement, shown particularly for *NCP, WCS* and *SSM*, is beneficial and the *PAD* function is also significantly improved indicating that the toxic pressure from DDX is also mitigated.

5.6 Reflection on GRO opportunity windows and added value

Despite the great potential of GRO to manage risks and improve soil functionality on contaminated land, which is acknowledged by many practitioners in Sweden and elsewhere (Berghel et al., 2021; Drenning, 2021a; White arkitekter AB, 2021), they are still seldom used in practice. In general, widespread adoption is still lacking due to perceived (and actual) limitations, uncertainties and challenges (Cundy et al., 2016; Gerhardt et al., 2017; Mench et al., 2010; Vangronsveld et al., 2009). Broadly, these include a 'status quo bias' and preference for conventional methods like dig-and-dump by practitioners (Montpetit and Lachapelle, 2017); 'nonknowledge' by practitioners regarding their functionality, methods and dealing with uncertainties, limitations or inefficiencies in GRO application (Bleicher, 2016); ecological risks from secondary poisoning due to wildlife grazing on metal-enriched plants or the improper handling of harvested biomass that may have higher concentrations of contaminants (Dickinson et al., 2009; Wang et al., 2019); and other practical challenges and limitations such as uncertainties relating to the required timeframes for GRO and their effectiveness as risk management strategies, applicability for different types of sites and contaminants, insufficient knowledge and experience, need for long-term monitoring, and lack of convincing proof-of-concept, amongst other concerns (Cundy et al., 2016; Gerhardt et al., 2017). Further, many studies have reported the lack of knowledge amongst stakeholders of GRO generally and of currently available decision-support tools (DST) for brownfield redevelopment and GRO application, (Bert et al., 2017; Cundy et al., 2016, 2015; Gerhardt et al., 2017; GREENLAND, 2014b; Onwubuya et al., 2009). In the Swedish context, awareness and knowledge about GRO, relevant techniques and their effectiveness are low and are major obstacles to their implementation (Berghel et al., 2021; Drenning, 2021a; White arkitekter AB, 2021). Several other challenges have also been noted in the Swedish context: conservative regulatory guidelines based on total concentrations and full source removal coupled with more stringent evidence requirements make the risk reduction via GRO prohibitively difficult to demonstrate, general skepticism regarding time aspects, uncertainties and limitations, requirement for a 'one-time solution' and risk aversion to long-term management and residual contaminants, lack of good examples and practical guidance, conflict between managing risks and total removal of contaminants; desire to get rid of a problem to avoid long-term liability and difficultly transferring liability, fear to make a mistake, lack of knowledge and experience transfer from research to practice, and not sufficiently considering alternative methods in options appraisal (White arkitekter AB, 2021).

To overcome the abovementioned challenges and obstacles, a shift in mindset and practice in managing contaminated land is likely required. Not least of these is accounting for bioavailability in risk assessment as a standard practice that is accepted by regulatory agencies, and also reformulating the remediation objectives in terms of 'upgrading degraded land' to provide wider benefits and 'risk reduction and management' instead of 'full source removal and decontamination'. Furthermore, the added value of alternative remediation options such as GRO may not be adequately considered in the current paradigm. Effectively valuing the benefits of GRO, accounting for them during options appraisal and raising GRO as viable remediation techniques are key aspects to their broader integration as viable land management strategies. Indeed, it is crucial to identify the situations and conditions – i.e., 'opportunity windows' – where GRO application would be feasible and have the greatest likelihood of success. For example, GRO may not be well-suited to highly contaminated sites, 'hotspots' or point source terms such as buried tanks or oil spills, but are particularly suitable for contaminated sites that pose low to medium risks to human health and the environment (Andersson-Sköld et al., 2014; Cundy et al., 2016; Enell et al., 2016; GREENLAND, 2014a). As a general starting point, the detailed operating windows identified in the Greenland project could be used to preliminarily screen brownfields to identify where GRO may be feasible for a particular site (GREENLAND, 2014b, 2014a), which include where i) there are budgetary and deployment constraints (e.g., large areas with diffuse contamination not causing immediate concern such as abandoned rail tracks); ii) biological functioning is desired post remediation for soft reuse (e.g., greenspaces); iii) ecosystem services are highly valued (e.g., riverbank greens, urban wilderness); iv) there is a need to restore land and a potential to produce non-food crops (e.g., marginal land for biofuel production)

Conversely, it is just as important to identify where GRO has limited potential such as where there is time pressure for short-term redevelopment of a site (i.e., within 1-2 years), the majority of the site is or will be under hard cover or has buildings under active use, and other site-specific factors constraining deployment due to e.g., poor soil quality, water availability, depth of contamination, climate, site topography and other local factors (GREENLAND, 2014a, 2014b).

An additional consideration for expanding/scaling the opportunity windows for GRO application is accounting for brownfield remediation and redevelopment (and opportunities for GRO) in long-term spatial planning. Strategic planning of land use over time is compatible with GRO to facilitate brownfield regeneration for soft reuses for circular land use management and preventing 'land take' of undeveloped greenfield land, as highlighted in the Brownfield Opportunity Matrix developed in the HOMBRE project (Bardos et al., 2016; Menger et al., 2013). Land and soil are finite resources facing growing pressures and conflicts over their use as conventional land use planning and soil management struggle to balance the supply of ecosystem services with society's demands (Breure et al., 2018; Maring et al., 2019). Indeed, traditional spatial planning does not consider soil quality or soil functions sufficiently (Lehmann and Stahr, 2010; Maring et al., 2019). Ideally, soil would be used according to its capability and best condition (Volchko et al., 2019); where, the designated land use is optimized to match the potential of the soil to sustainably provide ecosystem services (Beumer et al., 2014; Blanchart et al., 2018; Lehmann and Stahr, 2010; Maring et al., 2019), particularly for future green areas. For example, the soils with greatest capability to provide ecosystem services at a site could be protected with e.g., soil protection zones (Soils in Planning and Construction Task Force, 2022). In general, accounting for the added value of GRO for improving soil health should also improve their value proposition and expand their opportunity windows for where improvement or retention of soil functionality is an important project goal, especially in light of the newly proposed *Soil Directive* (EC, 2023).

A prerequisite for sustainable soil management in urban spatial planning is to plan according to a longer time-horizon to allow for more proactive remediation (Norrman et al., 2016), which would enable alternative land management and remediation approaches. If planners adopt a long-term perspective, phytomanagement could be used proactively as a land management strategy to both mitigate risks from contaminants and provide wider benefits at (potentially) contaminated land intended for redevelopment in long-term (5+ years) plans. Given the long time horizons and uncertainties, long-term monitoring and maintenance to evaluate the effectiveness of GRO will entail non-negligible costs and effort that must be considered early in collaboration with stakeholders and regulators when planning a GRO project (Cundy et al., 2020). Adaptive management (i.e., maintenance and monitoring programs that evolve iteratively to reduce uncertainty as management proceeds) can be tailored for phytomanagement projects to evaluate project goals and reduce uncertainty regarding remediation effectiveness and effects on soil health using key performance indicators, which can then be linked to important soil functions and ecosystem services (Birgé et al., 2016; Chapman, 2012; Epelde et al., 2014a; Gómez-sagasti et al., 2012). By including iterative decision points (e.g., every 5 years), it is also possible to re-examine the risk situation at the site after a period of phytomanagement to determine whether the site is fit for a different, more sensitive type of land use. For example, many green land uses with various degrees of permanency may be made possible over time with GRO interventions (Chowdhury et al., 2020).

Phytomanagement strategies could either applied on a long-term basis as a self-funding land management regime using crop-based systems for SRBLM (Andersson-Sköld et al., 2013; Bardos et al., 2011a) or as an interim 'holding strategy' allowing alternate, green land uses at vacant sites (Chowdhury et al., 2020; Cundy et al., 2016; Todd et al., 2016). Nature-based remediation to revegetate brownfields could also be used for ‘temporary conservation’ (Kattwinkel et al., 2011), ‘rewilding’ (Kowarik, 2018; Masood and Russo, 2023), and ‘renaturalization’ (URBiNAT, 2020) to promote biodiversity in urban areas and facilitate other NBS such as urban forests, constructed wetlands, industrial heritage parks, bioenergy production, and others (Chowdhury et al., 2020; Hou et al., 2023; Masiero et al., 2022; Nissim and Labrecque, 2021; Song et al., 2019). There is also great potential to integrate GRO within other, closely associated fields such as landscape architecture and sustainable urban water management (Cundy et al., 2016; Song et al., 2019). For instance, different site designs incorporating GRO applications as ‘phytotypologies’ (Kennen and Kirkwood, 2015) as well as ‘design guidelines’ (Todd et al., 2016) have been extensively covered from the perspective of landscape architecture that could be used as templates to integrate GRO into an aesthetically pleasing landscape design. Indeed, the aesthetic dimension of GRO application, particularly in urban context, ought not be neglected and certain families of plants such as Asteraceae and Brassicaceae are useful for both phytoremediation and aesthetic landscape design (Nikolić and Stevović, 2015). Importantly, small green interventions can have large positive ecological effects in urban areas (Mata et al., 2023), and applying GRO for brownfield revegetation could be vital elements of ‘urban environmental acupuncture’ (Starzewská-Sikorská et al., 2022) or other forms of small-scale NBS for marginal or neglected areas (Petrova et al., 2022a, 2022b).

5.7 Integrating GRO opportunity windows in contaminated land management

To facilitate inclusion of GRO in the decision-making process, the various methods and results presented in this Ph.D.-thesis are connected to the generic workflow for contaminated land management (CLM) in Sweden (SEPA, 2021b) and shown as a modified flowchart in Figure 5-1. Lines connect a specific output to the relevant step where the paper provides practical methods and demonstration and there is no connecting line if it is discussed in the paper but not concretely addressed.

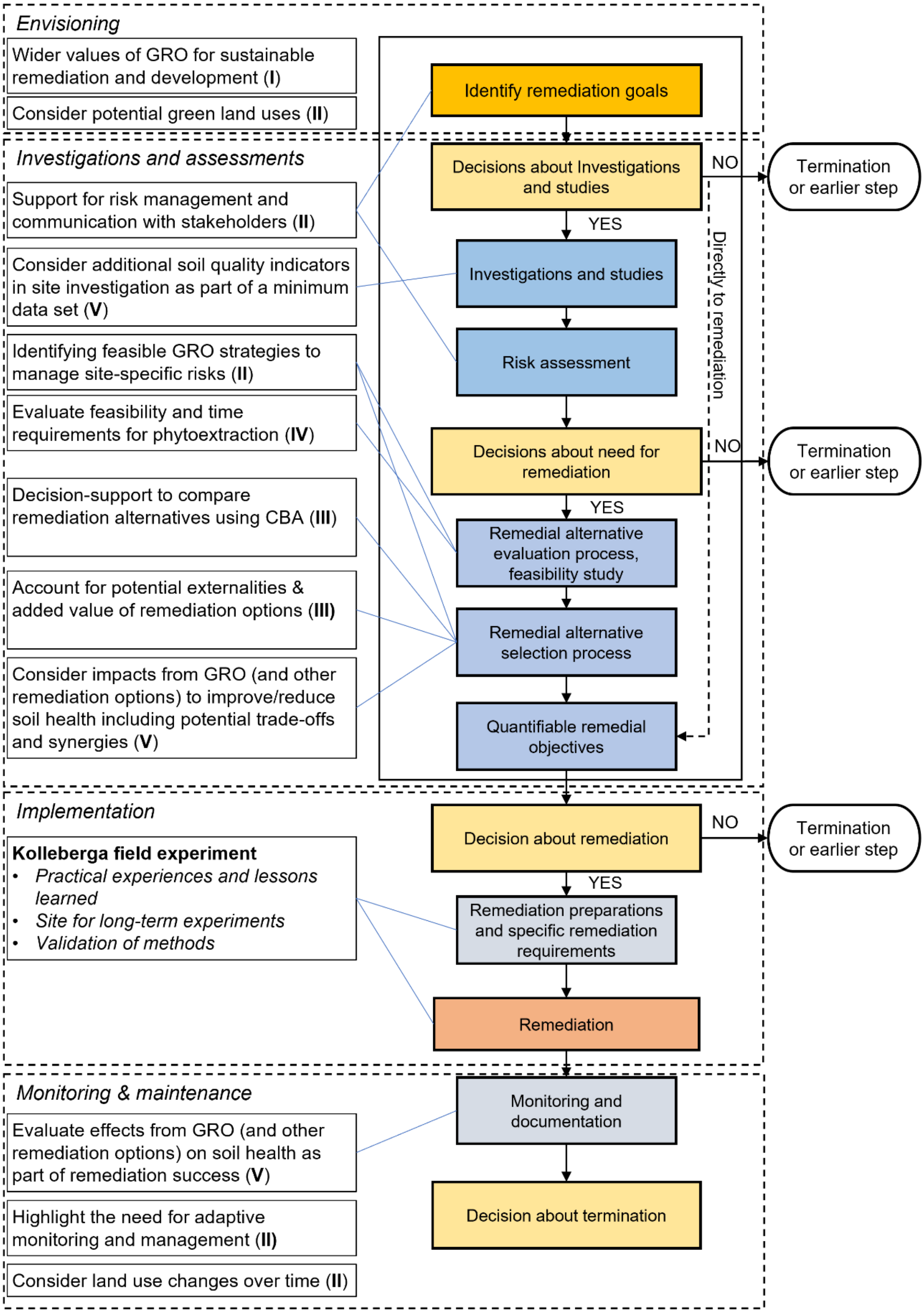

Figure 5-1. Contributions from this Ph.D.-thesis that could support integration of GRO into the generic CLM workflow and identify opportunity windows for practical application, in boxes on the right with lines connecting to the relevant step (paper number in parentheses). The flowchart is modified from (SEPA, 2021b), *and separated into four different project phases: envisioning, investigations and assessments, implementation, and monitoring and maintenance.*

6 CONCLUSIONS

This chapter presents a summary of the main conclusions from the thesis.

The main conclusions of this Ph.D.-thesis are summarized below:

- Brownfields present a significant opportunity for advancing sustainable remediation and development in urban areas to achieve the SDGs. Phytomanagement of brownfields with GRO is a nature-based solution to manage risks and provide wider values such as rehabilitating soil functionality and enhancing ecosystem services.

- There is scientific evidence to support the majority of the risk mitigation mechanisms identified to manage exposure and spreading pathways with GRO; however, evidence is still lacking for certain mechanisms and others such as reducing contaminant oral bioaccessibility are controversial and would require further examination to be considered as viable risk management strategies.

- The GRO risk management framework can facilitate communication and spreading knowledge of the risk mitigation mechanisms and required timeframes of various GRO to support remediation contractors, decision-makers, regulatory bodies and other stakeholders related to contaminated sites.

- A risk-based perspective is important for the success of GRO and management of the bioavailable fraction of contaminants at a site should be the main objective with GRO instead of reducing to a target value based on total concentrations.

- The case study and workshop applications demonstrated that an envisioned land use and site-specific contaminant linkages can be integrated into the generic framework to support the identification of relevant, site-specific GRO strategies and provide preliminary timeframes for risk reduction.

- Time requirements are an important factor in decision-making regarding the feasibility of GRO as well as the potentially social profitability of a GRO project. Probabilistic modelling could be used to estimate most likely time requirements and uncertainties of phytoextraction to further strengthen the decision basis for GRO implementation.

- For Kolleberga, the model results indicate that phytoextraction with pumpkin is impractical under current conditions to achieve the risk reduction target within a reasonable time frame unless the *BAF* and *BMP* can be significantly improved. DDX phytoextraction is more effective at lower initial soil ΣDDX concentrations and a strategy of 'soil polishing', 'bioavailable contaminant stripping', or as part of a 'treatment chain' could be feasible to manage areas at Kolleberga with ΣDDX concentrations lower than 5 mg/kg dw.

- Economic analysis provides valuable decision-support by evaluating the costs and benefits in society of GRO compared to conventional soil remediation alternatives and for which situations and conditions (i.e., opportunity windows) the use of GRO could result in a socially profitable project, e.g., for a given annual avoided cost of inaction from PFAS. As shown in the CBA, the added value of GRO can be shown in monetary terms through economic valuation of certain ES; however, not all ES are monetizable and certain benefits such as improved soil functionality may not be possible to include in a CBA.

- For the case study at Stockholm Arlanda Airport, the PFAS soil remediation alternatives that included GRO for the rest of the site (Alt 2, Alt 3 and Alt 5) generally ranked higher in both spreading scenarios than those that did not, which indicates that the added value of GRO in terms of 'reduced negative externalities' is an important consideration.

- The soil health assessment approach followed in this study provides an accessible, scientific method to evaluate the relative effects of GRO on soil health (the 'current capacity' to provide ES) that can be useful for practitioners in contaminated land management. The treated-SF and ES indices and the overall SHI provide simplified yet valuable information to decision-makers regarding the effectiveness of GRO and can highlight potential trade-offs and synergies in ES delivery.

- GRO have been shown to generally improve soil health and multifunctionality in the field experiment at the DDT-contaminated soil of the former tree nursery site Kolleberga in Sweden. This was demonstrated through the positive effects on the treated-SF indices and consequently the soil's capacity to provide ES, which is largely due to the positive effects of both legumes and biochar on multiple SQI that in turn result in an overall improvement in multiple SF.

7 IMPLICATIONS FOR PRACTICE AND FUTURE RESEARCH

This final chapter provides reflections on aspects that require further investigation to expand the opportunity windows for practical GRO implementation and future research.

The field experiment as Kolleberga provide valuable experience through 'learning by doing' and there have been many lessons learned with practical implications, including:

- Take care of the practical aspects such as irrigation, fencing, weed and pest management, etc. Investing effort in the beginning to establish a fence around the area to keep out rabbits and other herbivores as well as setting up an automated irrigation system were worth the effort and expense. Weed and pest management is also important since they can either outcompete or destroy the plant species that one wants to cultivate.
- Select indicators carefully and assess their sensitivity and usefulness. For example, the bait lamina test was difficult to time in the season and is highly impacted by seasonal climatic conditions and the first method to assess potentially mineralizable nitrogen did not perform as expected (non-detects). Certain analyses can also be expensive or difficult to source commercially.
- Best agronomic practices are important for the success of GRO. In the field experiment, biochar was shown to improve soil health but other organic amendments such as compost and bioaugmentation with endophytic bacteria and/or mycorrhizal fungi could have further improved the biomass production and overall effectiveness of GRO.

Finally, to focus research efforts moving forward, important aspects that may still be missing are connected to the CLM process in Figure 7-1 and some reflections on implications of this Ph.D.-thesis for current practice and future research are provided below:

- Guidance for working with GRO in Sweden and elsewhere is still highly demanded. Further work could go towards compiling knowledge gained and practical considerations for working with GRO to create guidance for a practical working process directed towards practitioners. The risk management framework can be integrated as a part of the working process, ideally as part of a broader 'GRO toolkit'.
- GRO should be better integrated into and harmonized with the broader field of NBS. There are many compendiums, handbooks and guidelines for NBS generated in ongoing projects (Voskamp et al., 2021); however, GRO for CLM and brownfield revegetation is seldom included. Phytoremediation is occasionally mentioned but is often limited in scope for water management to limit runoff to water bodies or mitigating air pollution.
- Soil functionality and desired ecosystem services for a future land use should be considered in the early stages of a project, as noted by the green box in Figure 7-1 similar to (Faber et al., 2013). Instead of only reducing total concentrations, GRO project goals should be formulated in terms of improving soil functionality and managing risks.
- The planned site design should also be optimized for the soil's capability and to preserve soil of good quality for soft uses. In this regard, accounting for soil quality indicators in addition to contamination levels will facilitate sustainable soil and land management.
- Long-term monitoring and management plans for phytomanagement need to be further developed. The test battery of SQI used in the soil health assessment is a first proposal for evaluating the effects of GRO and could also be used for a long-term monitoring program.

As shown in Figure 7-1, there are many aspects that connect to the ‘Envisioning’ phase to indicate that there is still a need to think more broadly about the potential role GRO can play in contaminated land management including aspects of spatial planning, integrating into adjacent fields, etc. The arrow including land stewardship, sustainable remediation and stakeholder engagement indicates that these must permeate the whole workflow.

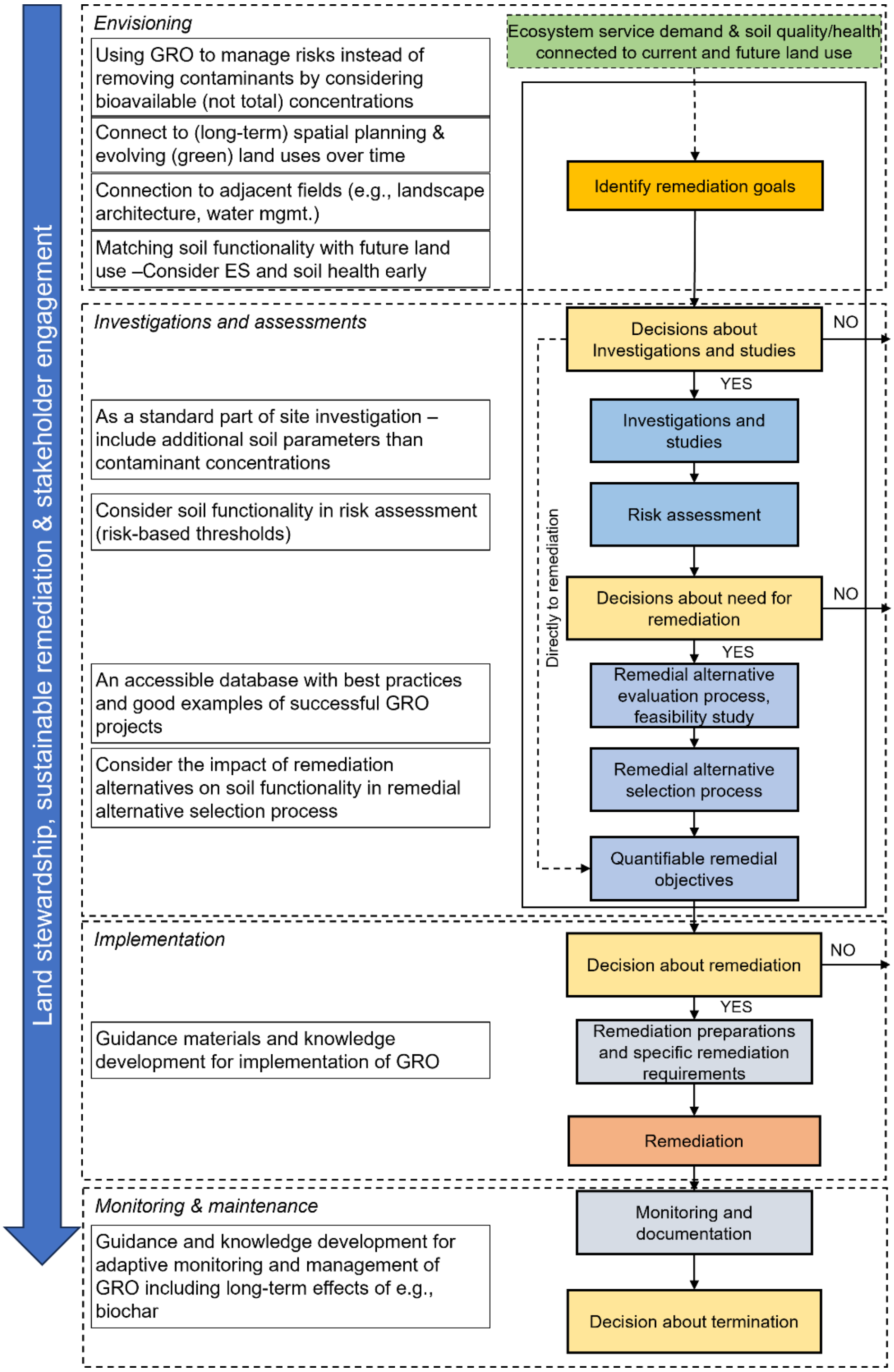

Figure 7-1. Reflections on implications for future work regarding what may still be lacking to better include GRO in CLM and expand the opportunity windows. The flowchart is modified from (SEPA, 2021b), and separated into four different project phases: envisioning, investigations and assessments, implementation, and monitoring and maintenance.

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9 APPENDIX A: SOIL FUNCTIONS

Table 9-1. Synthesis of soil functions identified in literature review.

Soil functions	Definition	Constituent processes/subfunctions	Main functional groups (soil biota)	References
Nutrient cycling and provision (*NCP*)	Generally refers to the capacity of a soil to receive nutrients in the form of by-products, to provide nutrients from intrinsic resources or to support the acquisition of nutrients from air or water, and to effectively carry over these nutrients into harvested crops (BIOSIS, 2023). Nutrient cycling and the production function (*soil fertility*) are two sides of the same coin and hardly separable (Vogel et al., 2019).	Includes a few main processes: i) the capacity to receive nutrients; ii) the capacity to store mobile nutrients within the root zone to avoid losses by leaching and gaseous emissions (*nutrient buffering capacity*); iii) the capacity of the soil to provide nutrients from the mineral and organic soil resources in available form (i.e., *nutrient mobilisation capacity* or *soil fertility*); iv) the capacity to support the uptake of nutrients by crops; and v) the capacity to support their successful removal in harvested crop (Schröder et al., 2016; Vogel et al., 2019). Biological soil processes include nutrient transformations by decomposers, element transformers, N-fixers, mycorrhizae, consisting of the main subfunctions: 1) Nutrient transformation - Fragmentation, Mineralisation, Nitrogen transformations, Sulphur transformations; 2) Nutrient reallocation - Aggregation, Bioturbation; 3) Nutrient assimilation - Food web assimilation, Mycorrhizal acquisition, Nitrogen fixation, Root foraging (BIOSIS, 2023)	Chemical engineers (bacteria, fungi); Ecosystem engineers (earthworms, microfauna); Plant roots	(BENCHMARKS, 2023; BIOSIS, 2023; Creamer et al., 2022; Kibblewhite et al., 2008; Schröder et al., 2016; Smith et al., 2015; Vogel et al., 2019, 2018)
Water cycling and storage (*WCS*)	Regulate and partition water and solute flow, including storage and infiltration of water, holding capacity and retention in organisms such as plants and soil biota	1) Infiltration and percolation - Macropore formation, bioturbation, aggregation; 2) Water storage - Bioturbation, Aggregation, fragmentation; 3) Biological retention - food web assimilation, root foraging, bioaccumulation (BIOSIS, 2023)	Ecosystem engineers (earthworms, macroarthropods); Plant roots	(BENCHMARKS, 2023; BIOSIS, 2023; Turbé et al., 2010; USDA, 2015)

Pollutant attenuation and degradation (*PAD*)	Filter, immobilize/sequester, modify, buffer, degrade, retain, and detoxify organic and inorganic pollutants by biological and physico-chemical processes (usually included within water purification)	Biological retention (biosorption, bioassimilation and transformation (biodegradation, biotransformation) processes and physico-chemical processes (sorption, complexation, precipitation, redox reactions, etc.)	Plants, Chemical Engineers, Ecosystem engineers	(BENCHMARKS, 2023; BIOSIS, 2023; Turbé et al., 2010; USDA, 2015)
Soil structure maintenance (*SSM*)	Physical stability and support for plant growth, habitat for soil organisms, and human structures	Dependent on inherent properties (e.g., texture, coarse material, OM content, etc.) but also more dynamic (soil formation) processes such as macro- and micropore formation, aggregation, bioturbation, root development, weathering of rock and accumulation of organic material, etc.	Plants (roots), Ecosystem Engineers (earthworms, macroarthropods); Chemical engineers (microbes)	(Kibblewhite et al., 2008; Turbé et al., 2010; USDA, 2015)
Carbon cycling and storage (*CCS*)	Soil decomposition, mineralisation and transformation processes that contribute to the cycling and retention of carbon in the soil	Consists of a few main processes: 1) Decomposition - fragmentation, microbial grazing, microbial respiration (mineralisation), methanogenesis; 2) Biochemical transformation - Methanotrophy, Nitrification, Denitrification; 3) Resource allocation - aggregation, bioturbation, exudation, food web assimilation (BIOSIS, 2023)	Ecosystem engineers (earthworms, other micro- and mesofauna); Chemical engineers (bacteria, fungi)	(BENCHMARKS, 2023; BIOSIS, 2023; Kibblewhite et al., 2008; Turbé et al., 2010)
Biodiversity and habitat *(not included in assessment)*	Soil's capability to contribute to ecosystem services by serving as a habitat for organisms, including as a species (gene) pool that can buffer ecosystem functions against species extinction, and sustain biological diversity, activity and productivity	Biological population regulation (Soil biota belowground interactions) - e.g., herbivory, predation, grazing, competition by 'biological regulators' or 'controllers'; Plant health promotion - resistance and defence, plant metabolism enhancement; Habitat quality - inherent conditions for life in soils (e.g., soil texture, water content, organic matter, pH, soil depth, etc.); Biodiversity - functional and structural	All (Inherent physical-chemical properties) & Biological Regulators (Acari, Collembola, nematodes); Chemical engineers (bacteria, fungi)	(BENCHMARKS, 2023; BIOSIS, 2023; Turbé et al., 2010; USDA, 2015)

10 APPENDIX B: SOIL-BASED ECOSYSTEM SERVICES

Table 10-1. Synthesis of soil-based ecosystem services identified in literature review (Adhikari and Hartemink, 2016; BENCHMARKS, 2023; BIOSIS, 2023; Bünemann et al., 2018; Dominati et al., 2010; El Mujtar et al., 2019; Faber et al., 2022; Faber and Van Wensem, 2012; FAO et al., 2020; Haygarth and Ritz, 2009; Jónsson and Davídsdóttir, 2016; Meulen and Maring, 2018; Orgiazzi et al., 2016; Paul et al., 2021; Potschin and Haines-Young, 2016; Rutgers et al., 2012, 2009; TEEB, 2010; Thomsen et al., 2012; Turbé et al., 2010; Wall et al., 2004). *(D) = direct; (I) = indirect (according to* (Meulen and Maring, 2018)*). ES with abbreviations were included in the soil health assessment.*

Category	(Soil-based) ecosystem services	Definition	Functional processes	CICES v.5.1 class
Provisioning	**Biomass production (*BMP*) – D**	By enabling plants to grow, soils enable humans to produce plant biomass for diverse purposes, including food, feed, fiber and fuel, within natural or managed ecosystem boundaries, soils physically support plants and supply them with nutrients and water	Most, if not all, soil functions contribute (to some degree) to biomass production	1.1.1.1:3 (Cultivated terrestrial plants, including fungi and algae, for nutrition, fibers and other materials and energy); 1.1.5.1:3 Wild plants (terrestrial and aquatic) for nutrition, materials and energy
	Physical structure & carrying capacity (D)	Supporting production functions for provisioning of biomass and human structures, physical base on which human infrastructures and animals (e.g., livestock) stand	Soil structure maintenance	*No CICES v5.1 equivalent*
	Raw (abiotic) materials (D)	Supplying raw materials and genetic material for human use; soils can be source of raw materials (peat, clay) but renewability of these stocks is questionable	-	4.3.1.1-2 (Mineral substances for nutrition and materials or as energy source)
	Fresh water (D)	Supplying raw water for human use (e.g., groundwater as resource for irrigation, drinking water or non-drinking water purposes)	-	4.2.1.1-2 (Surface water for drinking and non-drinking purposes); 4.2.2.1-2 (Groundwater for drinking and non-drinking purposes)
Provisioning/ Supporting	**Biodiversity conservation (incl. Gene pool) (D/I)**	The multitude of soil organisms and processes, interacting in an ecosystem, making up a significant part of the soil's natural capital, providing society with a wide range of provisioning and cultural services and other unknown services. This can include biochemicals and pharmaceuticals for human health benefits (e.g., penicillin is a soil bacteria)	Human benefit of Biodiversity & Habitat: biodiverse soil biota could be a source of new pharmaceuticals; complex soil biota and plant-soil biota belowground interactions give rise to many ecosystem services (and potential disservices)	1.2.1.1-2 (Genetic material from plants to maintain populations and for breeding); 2.2.2.3 (Nursery populations and habitats)

	Soil decontamination and bioremediation (*SDP*) – D	Soils provide pollutant management through bioremediation by plants and microorganisms: soils which can extract, absorb, transform and destroy or contain harmful compounds in soil and water to minimize release into the environment. Atmospheric deposits, applied fertilizers, pesticides or other contaminants are adsorbed into/onto soil aggregates by clay particles and organic matter, and degraded (chemically altered) by soil biota which metabolize contaminants through oxidative or reductive processes.	(Functional) Biodiversity of soil biota (primarily microbes but also earthworms) in combination with plants to perform Pollutant attenuation and degradation function to regulate soil and water quality	2.1.1.1 (Bioremediation); 2.1.1.2 (Biotic filtration, sequestration and storage of waste); 2.2.5.1-2 (Chemical condition of freshwaters/saltwaters); 5.1.1.3 (Abiotic filtration, sequestration and storage of waste)
	Water purification, supply and regulation (*WPSR*) – D	The capacity of a soil to receive, store and conduct water for subsequent use and the prevention of both prolonged droughts and flooding and erosion. Soils have the capacity to store and retain water, thereby mitigating flooding, recharging groundwater, and filtering nutrients and contaminants from water/runoff to avoid release in water bodies.	Nutrient cycling and provision (retention), Water storage and cycling (infiltration, retention, storage), Pollutant attenuation and degradation (water purification), and Soil structure maintenance (infiltration, percolation, retention)	2.2.1.3 (Hydrological cycle and flood control); 5.2.1.2 (Control of liquid flows)
Regulating and maintenance	**Climate regulation and carbon sequestration (CRCS) – D**	The capacity of a soil to reduce the negative impact of greenhouse gas fluxes (i.e., CO_2, CH_4, and N_2O) emissions on climate, as well as regulation of global temperature, precipitation, and other biologically mediated climatic processes. Soils have the ability to store C in short-lived or stable (long residence time) forms of soil organic matter (SOM).	Carbon cycling and storage (C decomposition, sequestration), Nutrient cycling and provision (nutrient transformations and release) and soil vegetation cover (primary production)	2.2.6.1 (Chemical composition of atmosphere and oceans); 2.2.4.1 (Soil quality by weathering processes); 2.2.4.2 (Soil quality by decomposition and fixing processes)
	Biological control of pests and diseases (I)	Control of potential pests and pathogens, Regulation of animal and plant (including algae, macrophytes) populations; the nature of the habitat provided by soils controls the proliferation of pests (to crops, animals or humans) and harmful disease vectors (viruses, bacteria) and regulates beneficial species populations	Biodiversity & Habitat (Soil biota and Plant-soil biota belowground interactions)	2.2.3.1 (Pest control, including invasive species); 2.2.3.2 (Disease control)
	Erosion control (EC) – I*	Soil and colloid retention within an ecosystem based on soil structure – provides a growing substrate for vegetation to bind the soil matrix with its roots to prevent erosion and landslides.	Soil structure maintenance (soil formation, aggregation, organic carbon)	2.2.1.1 (Erosion control)
	Regulation of local climate/temperature (I)*	Soils provide a habitat to vegetation that provides shading and cooling through evapotranspiration. Perceived thermal comfort may also be higher in a green environment	Soil water availability is often of importance for transpiration but most dependent on vegetation cover	2.2.6.2 (Local regulation of air temperature and humidity)

	Noise abatement (I)*	Adsorption of noise by open soil, and soil also provide a substrate for vegetation that can reduce noise	-	2.1.2.2 (Noise attenuation)
	Air quality regulation (I)*	Soil provides a habitat to vegetation that influences air quality	-	2.1.1.2 (Biotic filtration, sequestration and storage of waste); 5.1.1.3 (Abiotic filtration, sequestration and storage of waste);
	Pollination and seed dispersal (I)*	Pollination affected by soil management	Soil biota (e.g., earthworms and other ecosystem engineers) can have a significant impact on seed dispersal in soil	2.2.2.1 (Pollination); 2.2.2.2 (Seed dispersal)
Cultural	**Ethical, cultural, recreational and educational services (I)**	Direct, in-situ and outdoor interactions with living systems that depend (or do not) on presence in the environmental setting. All unsealed soils are suitable for research. Preservation potential is influenced by soil type and hydrological conditions. All soil properties as they relate to human health and comfort for use (e.g., ponding due to unfavorable hydrological conditions reduces usability for recreation): soil chemical quality texture, moisture, organic carbon, bulk density, structure, and aggregation		3.1.1.1-2 (Recreation); 3.1.2.3-4 (Culture, heritage, aesthetics); 3.1.2.1-2 (Scientific, educational, training interactions); 3.2.1.1-2 (Symbolic and spiritual meaning); 3.2.1.2 (Entertainment); 3.2.2.1-2 (Existence, option or bequest value)

**Highly vegetation-dependent – Soil quality is important for producing vegetation cover that provides the services but the soil itself does not have strong influence*

11 APPENDIX C: GRO RISK MANAGEMENT FRAMEWORK FACT SHEET

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ECO-GRO Project
Fact Sheets: #1a

Risk management framework for gentle remediation options (GRO)

The generic GRO risk management framework can be used to support the communication of the potential of GRO to manage the risks, through a variety of risk mitigation mechanisms, due to soil contamination. Also, it can be useful to provide stakeholders realistic expectations regarding GRO effectiveness and time requirements.

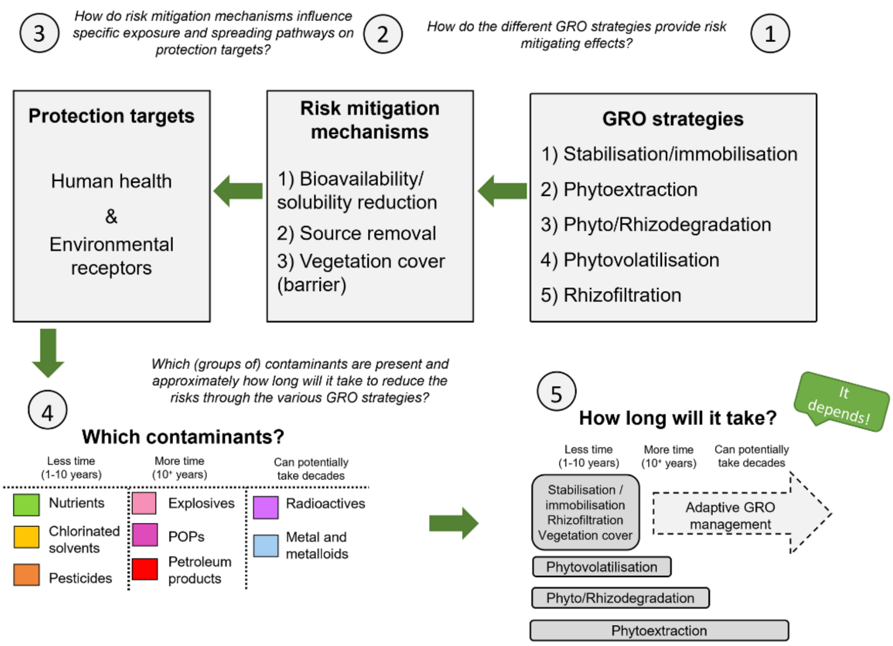
Different versions and their intended use

The risk management framework for GRO is proposed as a tool for communication and management of risks at contaminated sites and to support the application of GRO for phytomanagement. It is intended to be used for two primary purposes:

1) Risk communication: **the generic GRO framework** is developed for educating stakeholders generally about GRO and as a basis for communicating about potential risks associated with contaminated sites including how they can be managed through GRO (Fact Sheet #1)
2) Decision-support: **the site-specific application of the framework** is a tool to identify potential GRO strategies suitable for a particular site in collaboration with site owners, consultants, regulators, contractors, etc., which can in turn be useful to clarify expectations when applying GRO (Fact Sheet #2)

Understanding the generic GRO framework

As shown below, the GRO framework is best understood as a conceptualization of how 5 plant-based GRO strategies can manage specific contaminant linkages (i.e. source-pathway receptor connections) to reduce the risk (i.e., the probability of occurrence) of unacceptable harm arising at protection targets such as human health and sensitive environmental receptors. Risk mitigation could entail removing or modifying the source of contamination, interrupting the pathway or managing the receptor to reduce the risk of exposure to contaminants in soil.



Understanding the generic risk management framework for GRO

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ECO-GRO Project Fact Sheets: #1b

Risk management framework for gentle remediation options (GRO)

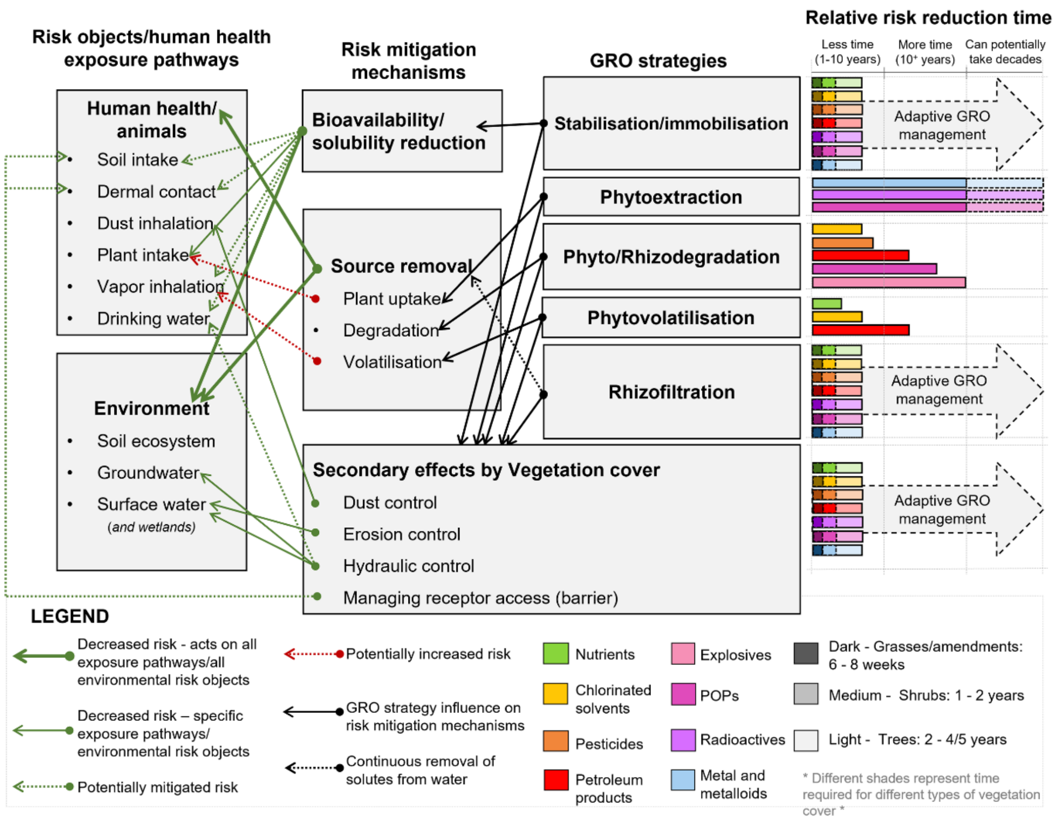
How long will it take?

The time required for risk reduction via GRO depends on a variety of site-specific conditions, including:

- Type and bioavailability of the contaminants
- Remediation objectives: bioavailable vs total concentrations
- Type of vegetation & time for establishment
- Soil physico-chemical parameters

The generic GRO risk management framework – full version

Mapping of the connections between GRO and their risk mitigating effects as well as the expected timeframes for effective risk reduction have been derived from literature. 'Adaptive GRO management' has also been included to note that adaptive (i.e. iterative and responding to changing needs/conditions) management and monitoring for upkeep of GRO is required to maintain risk mitigating effects over time, in particular for the GRO strategies for which source removal is not the primary mechanism. More information on the development of the framework as well as comprehensive tables providing the literature support and evidence for each of the stated connections is available in Drenning et al. (2022).



The generic risk management framework for GRO (Drenning et al. 2022)

Do you want to know more?

Drenning, P., Chowdhury, S., Volchko, Y., Rosén, L., Andersson-Sköld, Y., & Norrman, J. (2022). A risk management framework for Gentle Remediation Options (GRO). Science of The Total Environment, 802.

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ECO-GRO Project
Fact Sheets: #2a

Risk management framework for gentle remediation options (GRO)

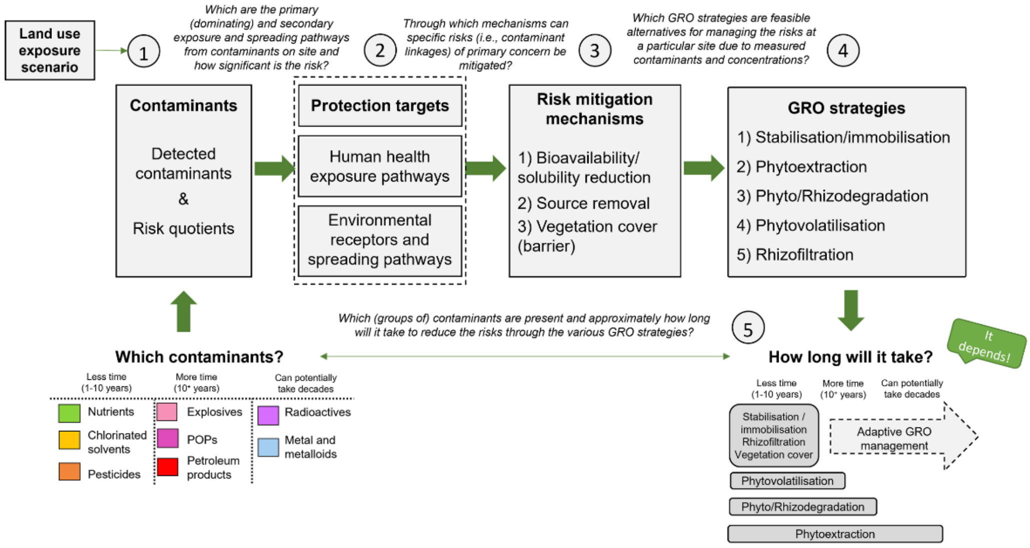
GRO and green land uses
GRO can be used to achieve a suitable 'fitness-for-use' for an envisioned end use at a particular site. 'Green' land uses are primarily considered here, as GRO are well-suited to sites where a 'soft' end use with biological functionality is desired, though it is not limited to only these.

Method for site-specific application

A series of steps can be followed in order to adapt the generic framework according to site-specific conditions:

I. Modifying the Swedish EPA's (SEPA) soil guideline value (SGV) model based on envisioned land use to create 'exposure scenarios' per potential end use
II. Based on site assessment, include contaminants of concern and identify the dominating risk pathways according to the SEPA model and extract site- and land-use specific SGVs
III. Calculate risk quotients (RQ) for contaminants by dividing concentrations present in the soil by the relevant SGV
IV. Add the contaminants of concern (and RQ's) to the framework figure, connect to exposure pathways/receptors and indicate which of these are the dominant risk per contaminant and land use scenario
V. Follow the connections in the framework to identify risk mitigation mechanisms and potential GRO strategies to manage the specific risks – use the relative risk reduction times to provide an estimate of the time requirements

Understanding the site-specific version of the GRO framework



Understanding the site-specific risk management framework for GRO

The risk management framework for GRO should be considered in collaboration with a group of stakeholders and ideally *early* while planning a brownfield redevelopment project when there is sufficient time and flexibility for GRO. In addition, there are several important questions to address before initiating a GRO project, including:

- *What is the remediation objective? Is it based on risk or total contaminant concentrations?*
- *Are ecosystem services and soil biological functionality for a 'soft' end use highly valued?*
- *Are there critical time/financial constraints?*

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Fact Sheets: #2b

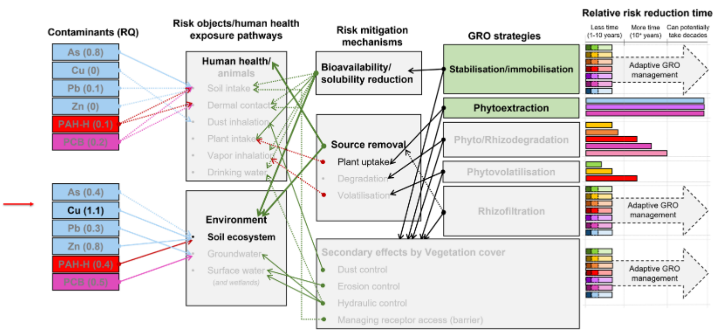
Risk management framework for gentle remediation options (GRO)

Green land uses
Three example green land uses were considered in the development of the GRO risk management framework to create exposure scenarios corresponding to differing levels of estimated risk per type of land use: biofuel park (low risk), recreational park (medium risk) and allotment gardens with onsite residence (high risk).

Site-specific application – example

The framework is demonstrated below to support the identification of GRO options for a case study site, Polstjärnegatan in Gothenburg, Sweden, given two envisioned green land uses: biofuel park and allotment garden. More detailed information on the site-specific application and case study is available in Drenning et al. (2022).

Polstjärnegatan – Biofuel park



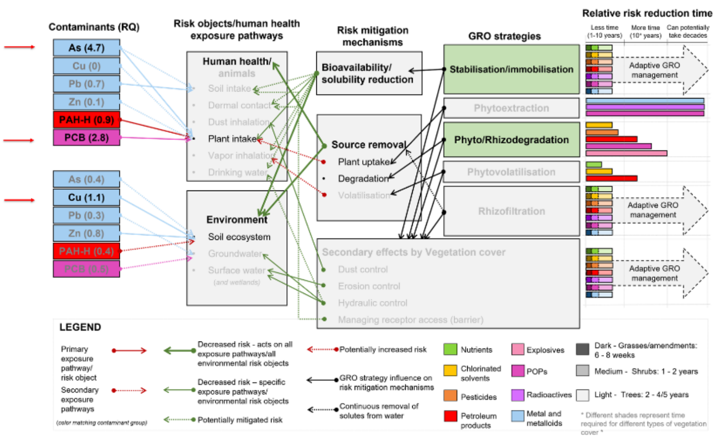
Interpretation

1. Biofuel Park

Land use immediately possible!
Primary risk: soil ecosystem (Cu)

→ A combination of GRO strategies for multi-mechanism effects to reduce risks in short-term (stabilisation/immobilisation) and long-term (extraction)

Polstjärnegatan – Allotment gardens



2. Allotment Gardens

Unrestricted land use not immediately possible (>10 years) – other intermediary green land uses should instead be considered
Primary risk: human health (plant intake) (As, PCB) & soil ecosystem (Cu)

→ GRO strategies to reduce contaminant bioavailability, selectively design vegetation cover to limit plant uptake and function as a barrier preventing human exposure
Phytoextraction may be unsuitable due to potentially increased exposure risk via plant intake

Site-specific risk management framework applied for two gren land uses (Drenning et al. 2022) — *Primary exposure/spreading pathway*

Land use sensitivity & risk assessment over time

Each land use can be modelled as a specific exposure scenario and, depending on the type/amount of contaminants that are present, different receptors and human health exposure pathways will dominate the risk assessment. Different types of 'sensitive' land uses will thus be possible immediately or over time after a period of risk reduction. This approach also considers GRO as a viable temporary ('holding strategy') or permanent land use to generate benefits like ecosystem services on otherwise derelict land.
This strategy often referred to as **phytomanagement.**

Do you want to know more?

Drenning, P., Chowdhury, S., Volchko, Y., Rosén, L., Andersson-Sköld, Y., & Norrman, J. (2022). A risk management framework for Gentle Remediation Options (GRO). Science of The Total Environment, 802.

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