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Integrated cost and environmental impact assessment of management options for dredged sediment

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ABSTRACT

Large quantities of sediment must be dredged regularly to enable marine transport and trade. The sediments are often polluted, with e.g. metals, which limits the management options. The aim of this study has been to assess costs and environmental impacts (impact on climate, marine organisms, etc.) of different management options for polluted dredged sediment, by combining life-cycle assessment (LCA) of the climate impact, scoring of other environmental aspects and a cost evaluation. This approach has been used to study both traditional and new management alternatives for a real port case. The studied options include landfilling, deep-sea disposal, construction of a port area using a stabilization and solidification (S/S) method, and a combination of the aforementioned methods with the innovative option of metal recovery through sediment electrolysis.

The LCA showed that deep-sea disposal had the lowest climate impact. The assessment of the other environmental impacts showed that the result varied depending on the pollution level and the time perspective used (short or long-term). Using sediment for construction had the highest climate impact, although other environmental impacts were comparably low. Electrolysis was found to be suitable for highly polluted sediments, as it left the sediment cleaner and enabled recovery of precious metals, however the costs were high.

The results highlight the complexity of comparing different environmental impacts and the benefits of using integrated assessments to provide clarity, and to evaluate both the synergetic and counteracting effects associated with the investigated scenarios and may aid early-stage decision making.

1. Introduction

Today's society is dependent on trading, an area where maritime transport plays an important role (Naletina and Perkov, 2017). Regular dredging of ports and other waterways is necessary to maintain sufficient water depth for maritime transport (OSPAR, 2017). As a result, large quantities of often contaminated sediment must be managed (Harrington et al., 2016). In Europe, approximately 200 million m^3 of sediment is dredged each year (MEDINGEGNERIA, 2009). In Rotterdam and Hamburg, two of the largest ports in Europe, up to 20 million m^3 of sediment is dredged every year, both within the ports and along the rivers connecting them to the sea. (Port of Rotterdam, 2005; TIDE, 2012).

Sediment in the vicinity of anthropogenic areas typically contain elevated levels of metals (Qian et al., 2015). Many metals are essential for normal biological function (e.g. copper and zinc), however, when present in very high concentrations, all metals have negative effects on biota (Besser et al., 2018; Jakimska et al., 2011). Due to their persistence, it can take a long time for a contaminated site to recover. Additionally, the metals may have long-term effects on biota once they enter the food chain (Donázar-Aramendía et al., 2020). The amount of contaminated sediment is hard to estimate, but data collected and summarized by the OSPAR monitoring program show that high concentrations of heavy metals (Cd, Pb, Hg) and other pollutants (PCB, PAH) are common in sediments from a large portion of the North Sea and Atlantic coastline. The available management options for dredged materials vary, depending on the level of contamination and local regulations (Dede et al., 2018)). The most common management methods

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are disposal in landfills or at sea (Akcil et al., 2015), although stricter environmental regulations are increasingly forcing alternative methods to be used.

One potential way to utilize dredged sediments is by applying stabilization and solidification (S/S). The S/S method is a technique used to fixate and encapsulate the contaminants inside sediment or soil by adding binders, thereby lowering the permeability and reducing leachability (John et al., 2011). Additionally, when cementitious binders are used, the masses become rigid and strong enough to be used in construction. Different types of binders can be used for stabilization of dredged sediment such as cement, lime, fly ash, slag, clays etc. (Rađenović et al. 2019, Wang et al. 2015, Zhang et al. 2021). Many studies have focused on evaluating the strength and the leaching behaviour using different mixtures of binders (de Gisi et al., 2020). The S/S method is relatively common in infrastructure projects (Stabcon, 2011), however less commonly used in water-related activities, possibly because landfill costs have previously been relatively low. The method has been used in a few full scale construction projects around the world (Bendz et al., 2011, O'Shea et al., 2019), e.g. for expanding port areas or in road construction (Achour et al., 2014).

Techniques for recovering metals from polluted sediments is another possible treatment method. Human society depends on metals. Base metals such as copper and zinc are used in large quantities in numerous applications. In contrast, special metals, such as cobalt, are used in smaller quantities, but are crucial in batteries and various appliances. Cobalt has been listed as a critical metal with scarcity concerns by the EU (Reuter, 2013). The mining of metals is dominated by a small number of countries, and consequently, the supply of many essential elements is highly sensitive to the political and socio-economic situation in these countries, and in the world overall (Bloodworth, 2014). Furthermore, climate change and water scarcity may threaten the supply of certain metals in the future (Northey et al., 2017). Increased recycling of metals and recovery from dilute and unconventional sources will be essential to secure future supply, and can be achieved e.g. through electrolysis of the polluted sediments. In electrolytic remediation a current is applied between a pair of electrodes, which produces migration of ionic species and dissolved contaminants towards the electrodes and thus decontamination of the sediment (Han et al., 2021). Electrokinetic remediation has been successfully used to remove metals and organic pollutants from polluted sediment in lab-scale (Rozas et al. 2012, Pedersen et al. 2017) and in few pilot-scale projects (Masi et al., 2017, Mao et al., 2019).

As illustrated by (Norén et al., 2020), all management strategies have different costs and environmental impacts. In addition, the short-term environmental impacts of different management approaches differ from the potential long-term impacts. The selection of management strategies for contaminated sediment is often based on the costs and regulations/concentration thresholds, thus neglecting the potential environmental impacts (Sparrevik et al., 2011). One way to assess the relative environmental impact of different management strategies is life cycle assessment (LCA), where the potential environmental impacts are compiled and evaluated from "cradle to grave" (ISO, 2006). As an example, life cycle assessments of soil/sediment remediation projects have shown that the risks originating from the remediation process can exceed the environmental impacts of a contaminated site (Barjoveanu et al., 2018; Lemming et al., 2010; Suer et al., 2009). The purpose of an LCA is to identify the step in the production chain or treatment methodology where the environmental impact is greatest. The product and/or method developer can then direct their efforts towards this step and focus on improving the weakest link in the production or treatment methodology. There are, however, only a few studies where LCA has been applied to evaluate various management strategies of marine sediments (Barjoveanu et al., 2018). To the best of our knowledge, there is no study comparing several of those management strategies to each other, and no study of sediment treated with electrolysis and its potential for S/S application.

Many LCA studies focus on the climate impact, often because this is where data is most readily available. Many other environmental aspects are either not clearly described in LCA databases, or not included for all processes. Furthermore, the LCA methodology treats all effects as global effects, e.g. air pollution, despite being applied at the local scale (Morais and Delerue-Matos, 2010). Local impacts, such as the impacts of deep-sea disposal on the local morphology, fauna, and flora, are generally not considered in LCA databases. To perform an environmental assessment it is important to include all relevant aspects, even when quantitative data is not available. By rating the importance of different aspects on different scales a combined impact of all types of aspects can be given. This approach can be used when many different types of aspects need to be evaluated, when there is limited data on certain aspects, and/or when there are varying time and geographical scales involved. Integrated assessments combining several different aspects can give valuable information for decision-making (Sparrevik et al. 2011; Todaro et al. 2021).

In this study we have performed an integrated environmental and cost assessment of management options for contaminated sediment. The management options include deep-sea disposal, landfilling, S/S, and electrolysis. A comparative LCA has been applied to study the greenhouse gas (GHG) emissions arising from the different alternatives. The LCA was complemented with a semi-quantitative scoring matrix to study other important environmental aspects and a cost evaluation. The study offers insights into the feasibility of different options and the environmental effects as well as possible ways to overcome these effects. The method aims to illustrate and compare different alternatives in a structured way, combining both quantitative and semi-quantitative data and is important as a communicative tool in an early stage of a decision process.

2. Method

2.1. Life cycle assessment

The climate impacts of the treatment strategies for contaminated dredged sediment have been analyzed by application of a comparative LCA. In a comparative LCA, only the impacts that differ between the alternatives are considered, in contrast to a standalone LCA, where the impacts of all processes in the life cycle are accounted for. The *life cycle inventory* (LCI), which is the basis of the LCA, was performed using the SimaPro v. 9.1.0 software (PRé, 2021) and the database ecoinvent, v. 3.6 (Wernet et al., 2016) with allocation at the point of substitution applied on, and complemented with, real site data.

The real site data was acquired from previously performed sediment sampling in the port of Gothenburg (COWI, 2017) and the port of Oskarshamn (VBB Viak, 1996), as well as data collected from a construction project using S/S on dredged sediment (COWI, 2017). The study has also applied data from laboratory investigations on potential sediment remediation techniques applying electrochemical oxidation (Anna Norén et al., 2021a).

The *environmental impact assessment* focused on GHG, based on the CML method. The impact category Global Warming has been used. The life cycle was limited to the management after dredging (as this step is included in all approaches, i.e. dredging will be performed regardless of the choice of subsequent management option).

2.2. The port of Gothenburg

The Port of Gothenburg is located on the Swedish west coast, at the outlet of the river of Göta älv (Fig. 1). It is a combined river and coastal port, and the largest port in Scandinavia, with a trading volume of 40 million tons, dominated by container goods and with significant passenger traffic (Port of Gothenburg, 2019).

Every three to five years, around 200 000 m^3 of sediment is dredged from the port to allow the current shipping activities (Göteborgs hamn,

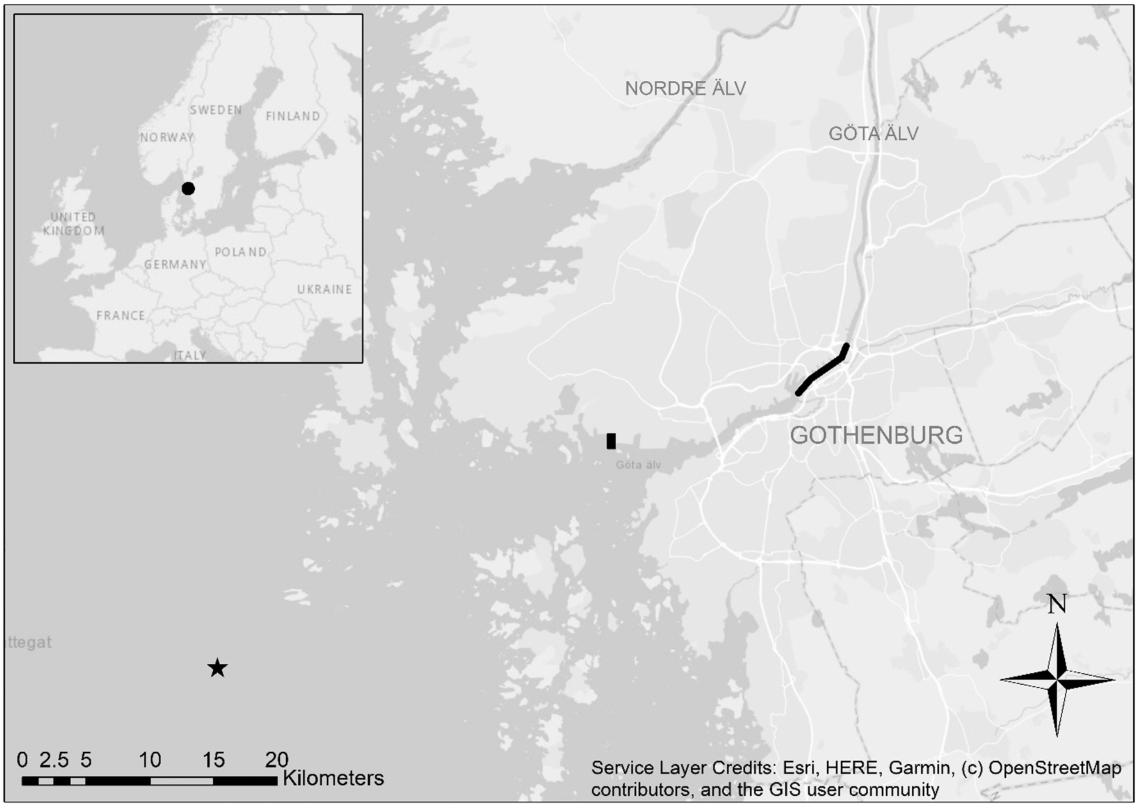

Fig. 1. The Port of Gothenburg, part of the river Göta älv and the river Nordre älv. The line in the river Göta älv marks the dredging area, the rectangle marks the construction area at Lilla Aspholmen and the star marks the location of the Vinga deep-sea disposal site. In the upper left corner, the location of the city of Gothenburg (Sweden) is marked with a dot.

2016). Additionally, the Port of Gothenburg needs to expand its berths and terminal areas to reach its target of doubling its activities within the coming 10-year period. To achieve this, the Port of Gothenburg plans an expansion of the area by constructing a new terminal at Lilla Aspholmen. A potential solution would be to, instead of using rock or other juvenile materials, make use of dredged sediment masses for the construction using the S/S method.

The sediment in the Port of Gothenburg is considered to have a relatively low metal pollution level, with copper, lead, and zinc exceeding the levels at which biota may be affected according to the Canadian guidelines for marine sediment (CCME, 2021; Norén et al., 2020). To investigate how the level of pollution affects the results, data from highly polluted sediment from the Port of Oskarshamn was also used, and applied to the Gothenburg settings. The sediment from Oskarshamn is classified as heavily polluted with cadmium, copper, lead, and zinc present at levels likely to cause acute toxic effects, according to the Norwegian sediment classification (Direktoratsgruppen vanndirektivet, 2018; Norén et al., 2020). The severe pollution led to a dredging operation being performed between 2016 and 2018, however the data used here is from before the dredging was performed.

2.3. Goal and functional unit of the LCA

The *goal* of the LCA was to evaluate differences in GHG emissions between the investigated treatment approaches for contaminated dredged sediment. Nine management scenarios were investigated, for which the functional unit was set to 100 m^3 dredged sediment. The scenarios were based on metal pollutant concentrations measured in either the less polluted sediment from the Port of Gothenburg (lowpoll) or the more polluted sediment from the Port of Oskarshamn (highpoll). The sediments from the two sites were classified into different categories based on the Swedish guidelines for hazardous waste, the Swedish soil quality guidelines, and the legal decision regarding deep-sea disposal at Vinga, as described in (Norén et al., 2020), since there is no standardized methodology for sediment classification The categories were deep-sea disposal, landfill for non-hazardous waste and hazardous waste. Supplementary Table S1, shows the metal concentrations of the sediment in the different categories. The amount of sediment in each category in Gothenburg (lowpoll) and in Oskarshamn (highpoll) was used to create

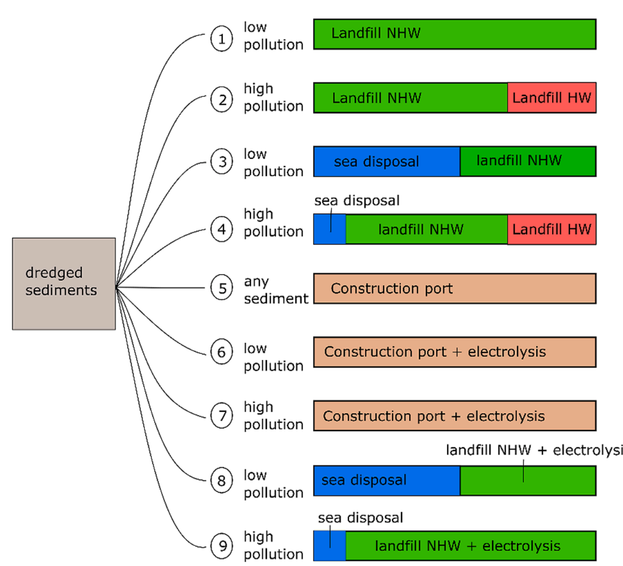

Fig. 2. Schematic view of the different management scenarios for dredged sediments. The length of each bar denotes the percentage of the sediment volume that is handled in this way.

the following nine scenarios. The scenarios are also shown schematically in Fig. 2.

1. **Landfill NHW lowpoll**: All dredged sediment masses are disposed in a landfill for non-hazardous waste (100%).
2. **Landfill NHW + HW highpoll**: Parts of the sediment are disposed in a landfill for hazardous waste (31.5%) and the rest in a landfill for non-hazardous waste (68.5%).
3. **Sea + landfill NHW lowpoll:** Deep-sea disposal (52%) and landfill (48%) for non-hazardous waste. The remaining sediment, which cannot be disposed at sea is deposited at a landfill for non-hazardous waste.
4. **Sea + landfill NHW + HW highpoll:** Deep-sea disposal (11.5%) and landfill for non-hazardous (57%) and hazardous waste (31.5%).
5. **Construction port**: Use of all the dredged sediment as filling material for port construction, regardless of pollution level (100%).
6. **Construction port + electrolysis lowpoll:** Electrolysis of low pollution sediment to extract valuable metals. Use of the remaining sediment as filling material for port construction (100%).
7. **Construction port + electrolysis highpoll:** Electrolysis of high pollution sediment to extract valuable metals. Use of the rest of the sediment as filling material for port construction (100%).
8. **Electrolysis + sea + landfill lowpoll:** Deep-sea disposal (52%) and electrolysis of the remaining sediment (48%). After electrolysis the sediment is disposed in a landfill for non-hazardous waste.
9. **Electrolysis + sea + landfill highpoll:** Deep-sea disposal (11.5%) and electrolysis of the remaining sediment (88.5%). After electrolysis the sediment is disposed in a landfill for non-hazardous waste.

2.4. Scenario descriptions and assumptions

Supplementary Table S2 and S3 contain a description of each scenario. Here, a general overview is given.

In cases where the transport distance from the factory was unknown, ecoinvent's market processes with default transport distances (Weidemaa et al., 2013) were used. Otherwise, transformation processes were used in combination with the known transport distance. For road transport, a lorry with a loading capacity > 32 tons was always used, as large quantities must be transported. A vehicle of euro class 6 was always chosen for the scenarios, as this is currently the best available technology. This may lead to road transport emissions being underestimated, but as the vehicle type was not known, this approach gives an estimate that is comparable between the scenarios. Electricity usage was based on the Swedish electricity mix, which contains a large portion of renewable and nuclear energy.

2.4.1. Deep-sea disposal and landfill

For transport of sediment to the deep-sea disposal location at Vinga and to the construction site at Lilla Aspholmen, a barge designed for inland waterways in European conditions was used.

Sediments were transported untreated and without dewatering to both landfill and deep-sea disposal. Dug dredged sediments may not need dewatering before being transported to a landfill, whereas sediments that are dredged by suction have a higher water content and therefore always need dewatering prior to landfill (Naturvårdsverket, 2003). Dewatering can be performed passively in a basin, which requires an available surface, the building of an enclosure, and the capacity to remove metals and other pollutants from the leachate. Dewatering can also be actively performed using a number of different methods, although these may be costly and energy-intensive. For simplicity, it has here been assumed that sediment is transported to landfill without dewatering, as all sediments meet the water content guidelines for landfilling, having a dry content of 50%. This approach likely overestimates the transport contribution, but underestimates other emissions that may arise due to the dewatering process.

Landfill for non-hazardous waste has been approximated by an inert material landfill for Swiss conditions, and landfill for hazardous waste by a sanitary landfill for Swiss conditions, as data for Swedish conditions were not available in ecoinvent.

2.4.2. Stabilization and solidification

The recipe for the S/S process was based on field tests in the Port of Gothenburg (COWI, 2017), and consisted of a total of 150 kg/m^3 binders, made up of 50% cement and 50% ground-granulated blast-furnace slag (GGBS). Other ingredients and chemicals were assumed to have little environmental impact.

The main ingredient in cement production is limestone, why the market process for limestone has been added. No other input materials for cement have been added, which will cause emissions to be underestimated. However, the release of carbon dioxide during the cement production dwarfs all other emissions, which implies that adding these materials would not change the conclusions. To account for the cement production, a process describing the production of Portland cement at a factory in Slite has been added to SimaPro. The factory in Slite is the only Swedish factory producing Portland cement (Hammarstrand and Millander, 2015). The carbon dioxide emissions from the cement production process were 780 kg CO_2/ton cement. The cement was transported from the factory in Slite to the construction site by freight ferry and lorry.

The GGBS is created from slag, which is a by-product of iron and steel making. The GGBS used in this study was produced in Bremen, Germany, and transported by lorry and container ship to Uddevalla, Sweden, then from Uddevalla to Gothenburg by lorry. The production emissions were 30 kg CO_2/ton GGBS (ThomasCement, 2018). A process for the production of GGBS has been added to SimaPro. As the slag is a by-product, no input materials were added.

The S/S process was based on construction in the port of Gävle (Sweden), where a process stabilization machine was built on site for the mixing of binders with sediment. The electricity consumption of the machine, 27 kWh/100 m^3, has been taken into account. The construction of the machine has not been included, which may result in a slight underestimation of the emissions.

The use of S/S sediment to construct a port has been compared to the option of filling the same area with crushed stone, as this is a commonly used material for such structures (private communication, Epifanio, Project Manager – Infrastructure, Gothenburg Port Authority, 201112). For this reason, the production and transport of 100 m^3 crushed stone has been deducted from all scenarios relating to port construction. We have assumed that the material is the only difference between the scenarios, i.e. that the building of the structure, machinery for distributing the material, asphalting, construction of embankments and quays, etc. are the same in both cases.

2.4.3. Metal extraction through electrolysis

The removal efficiency of metals that can be achieved with electrochemical methods vary widely in the scientific literature (Han et al., 2021, Pedersen et al., 2017). In a recent laboratory study with sediment from the port of Gothenburg, the removal efficiency for tributyltin (TBT) and metals by electrochemical oxidation was investigated. Although the process was optimized for TBT oxidation, metals could also be recovered, including valuable metals such as silver (40%), copper (15%), and zinc (9%), and also metals with a high toxicity, such as cadmium (50%) and lead (15%) (Norén et al., 2021a). Other studies of electrolytic metal recovery from aquatic solutions have reached efficiencies close to 100 %. (Jin and Zhang, 2020; Modin et al., 2012) In this study, it has been assumed that 60% of the content of all metals can be extracted using electrolysis. Since the value is uncertain and likely varies for different metals, a range of 30 to 90% was also studied.

In the scenarios for the Port of Oskarshamn (highpoll), it has been assumed that metals are extracted from masses classified as hazardous waste. As there are no masses classified as hazardous waste in the Port of Gothenburg, it has been assumed that metals are extracted from all sediments that cannot be disposed of at sea. The metals chosen for extraction were cobalt, copper, nickel, tin, and zinc, which are common metals, widely used in society. All five metals can be recovered from solutions using electrochemical reduction (Bai and Hu, 2002; Modin et al., 2017, 2012; Sharma et al., 2015). The amounts of metals extracted per 100 m^3 in Oskarshamn and Gothenburg are shown in Table 1.

For the electrolysis, materials are needed for the anode, cathode, and electrolyte. In the study by Norén et al., 2021a, the material used for the anode was boron-doped diamond on a niobium surface (8.57 kg per m^2 anode surface), and the cathode consisted of stainless steel (7.85 kg per m^2 cathode surface). The production rate of oxidants at the anode and the reduction rate of metals at the cathode are dependent on the electrical current applied between the two electrodes. For a given current density, the sediment mass that could be treated increases with the size of the electrode surface.

In the present electrolysis scenarios, electrolysis has been assumed to treat 1 m^3 sediment per day, implying a size of 6 m^2 for the anode and cathode materials, respectively. There are no relevant processes in SimaPro that can be used to mimic the laboratory set up, and the boron-doped diamond anode would need further development to be suitable for large-scale application (He et al., 2019). For this reason and for simplicity, both the anode and cathode materials have been assumed to be aluminum, the production of which is very carbon-dioxide intensive, therefore representing a worst case scenario. Aluminum is transported to Sweden from Guinea by container ship. It has been assumed that the electrodes can be used for electrolysis of at least 1000 m^3 sediment before they have to be replaced. In the future, better electrodes may be available, decreasing the required energy consumption.

The electrolysis has two aims; to remediate the sediments through oxidative degradation of organic pollutants, and to extract metals that can be used for commercial (or other) purposes. This process has been compared to the conventional methods for extracting and transporting the same metals. For the conventional methods, available SimaPro transformation processes were applied for each studied metal, and combined with transportation to Sweden from a region with large production of the respective metals. If there were several countries to choose from, the country closest country to Sweden in shipping distance was chosen. Because copper and zinc are mined from Swedish ore and produced in Sweden no transport distances were added for these metals. Cobalt was transported from Congo, nickel from Finland, and tin from Peru. Container ships were used to transport metals from ports all over the world. The transport distances have been estimated using the SeaRates calculator. The GHG emissions from the conventional metal extraction process have been subtracted from the scenarios involving electrolysis.

2.5. Scoring of additional environmental impacts

To complement the LCA with other environmental impacts than the climate impact, an integrated assessment model based on the previous work by Norén et al. (2020) has been used to perform a systematic evaluation. Based on the literature, the short and long-term impact on marine organisms, land use, air quality (GHG emissions excluded), terrestrial biota/health, and other potential risks were estimated for each scenario. The short- and long-term environmental impacts ($F_{i,\ k}$) were given a value between −3 (negative impact) and 3 (positive impact) and multiplied with the volume percentage of each management alternative (W_k) in each scenario to calculate the environmental impact of the scenario in question (Equation (1)). A negative result implies effects that are bad for the environment and positive results are good.

$$E_j = \sum F_{i,k} * W_k \tag{1}$$

E_j is the estimated total impact for each management scenario, j.

W_k is the relative amount of materials used for each management option, k.

$F_{i,k}$ is the impact factor for each impact, i, and management alternative, k.

Additionally, we have performed a workshop were the methodology and criteria of the environmental assessment have been discussed with experts and planners from the port of Gothenburg, national and regional planners and soil, water and sediment experts.

Table 1
Calculated mass of metals extracted using electrolysis of sediment from Oskarshamn (highpoll) and Gothenburg (lowpoll) with the assumption of 60% recovery efficiency.

Metal	Amount of metal extracted (kg/100 m^3 dredged sediment)	
	Oskarshamn	Gothenburg
Co	3.0	0.38
Cu	49	2.6
Ni	2.1	0.95
Sn	0.006*	0.006
Zn	95	9.0
Total amount:	149	13

* Sn was not measured in Oskarshamn; the same content has therefore been used for both sites.

2.6. Cost evaluation

For a management option to be feasible it should also be cost-effective. To include this aspect we perform a cost evaluation for the management options based on literature data. Landfill costs were set to approximately 130 USD/m^3 for non-hazardous waste and 150 USD/m^3 for hazardous waste based on Norén et al. (2020). Costs for transporting sediment to landfill were assumed to be 120 USD/h. Sea disposal costs were set to 2.3 USD/m^3 based on Norén et al., 2020. There are only few estimates of S/S costs in the literature, and with varying figures. According to O'Shea et a., (2019), S/S costs including dredging range from 30 to 100 €/m^3. Cost for S/S are in the range 40–350 USD/m^3 according to Mulligan et al. (2001) and in the range 10–40 €/m^3 according to Sednet (2004). Rađenović et al. (2019) estimates the costs of a S/S plant to 5–6 €/m^3. Wang et al. (2015) estimated the cost of creating sediment blocks with the S/S method to 23 €/m^3, mainly considering the cost of cement and water. The cost of the S/S machine used in the port of Gothenburg including binders was estimated to 36 USD/m^3 (Thulin, 2018), which is the value used here. Many studies rate S/S as cost-effective compared to landfilling (Couvidat et al., 2016). There is also a potential revenue when the S/S sediment is used as filling material. In this case, the cost of excavation and transport of crushed stone is avoided for the port of Gothenburg. The cost of crushed stone is approximately 22 USD/m^3. Lastly, the cost for electrolysis is very uncertain, because there are very few full-scale applications and no standard setup. Falciglia et al. (2020) estimated the cost of energy for electrokinetic decontamination of sediments to 1–3.5 €/m^3. The energy consumption for the present study is approximately 1.7 USD/m^3. However, an estimate should also include all the costs for setting up a facility. Mulligan et al. (2001) reports costs of 140–350 USD/m^3, but without any information on the project. Masi et al. (2017) evaluated the costs for a small electrokinetic remediation plant to approximately 350 €/m^3, taking into account costs for electrodes, pipes, acid and energy. Mao et al. (2019) estimated the costs of energy, material and labor for a small scale remediation project to 127 €/m^3. For simplicity, we assume a cost of 200 €/m^3, but this number is very speculative. However, there is a potential revenue in electrolysis projects because of the retrieved metals. This revenue has been calculated as the average 5-year metal price at the London Metal Exchange (lme.com) multiplied by the metal content for each of the retrieved metals.

3. Results and discussion

3.1. Life cycle assessment

The LCA results are summarized in Fig. 3, which shows the global warming potential for the nine scenarios and the different types of processes contributing to global warming. The proportions in Fig. 3b are based on the user input processes as illustrated in Fig. 4 (with two additional examples in Supplementary Figure S1).

As illustrated by Fig. 3, the lowest contribution to global warming among the different scenarios relates to sea disposal in combination with landfill for non-hazardous waste (scenario 3). The reason is that this scenario does not involve any pre-treatment, and that no energy is used for the sea disposal, which means that emissions are mostly related to transportation. The contribution to global warming increases when hazardous waste is included (scenario 4), due to the increased transport distance and the energy needed to control and minimize toxic emissions from the hazardous waste landfill. In scenario 1–4 transportation is the main emission source. The road transport distance therefore plays an important role for the total impact, and this can vary for different projects. Generally landfill for non-hazardous waste are more common than for hazardous waste, which gives shorter transport distances if the masses are relatively clean. The emissions from sea transport are much less sensitive to the distance.

The port construction (scenario 5) results in a substantially higher global warming contribution than any of the landfill and sea disposal scenarios. This is due to the use of cement, which contributes 93% of the GHG emissions in this scenario. The emissions may even be slightly underestimated because all the ingredient for cement production were not added. However, the possible uptake of CO_2 by cement during its lifetime is also not included. Cement production is energy-demanding, and the production process releases high amounts of carbon dioxide. At present, cement production constitutes 14% of Sweden's industrial contribution to GHG emissions (IVA, 2019); globally it represents approximately 4% of the carbon dioxide emissions from fossil fuels (Andrew, 2018). However, there is a move towards more climate neutral methods in cement industry, e.g. by using the Carbon Capture and Storage (CCS) technique (GCCA, 2021; HEIDELBERGCEMENTGROUP, 2021). With CCS, the carbon dioxide from large point sources, such as coal or gas power plants, steelworks and cement industries, are separated from the exhaust fumes and stored in the bedrock or under the seabed. Another (future) option is storage via algae, which can be used in the production of biofuels (Moreira and Pires, 2016).

The scenarios including port construction in combination with electrolysis (scenarios 6–7) have slightly higher GHG emissions than they would without the electrolysis, however, the scenarios with electrolysis involve a number of assumptions.The energy consumption has been estimated based on upscaling from trial laboratory experiments,

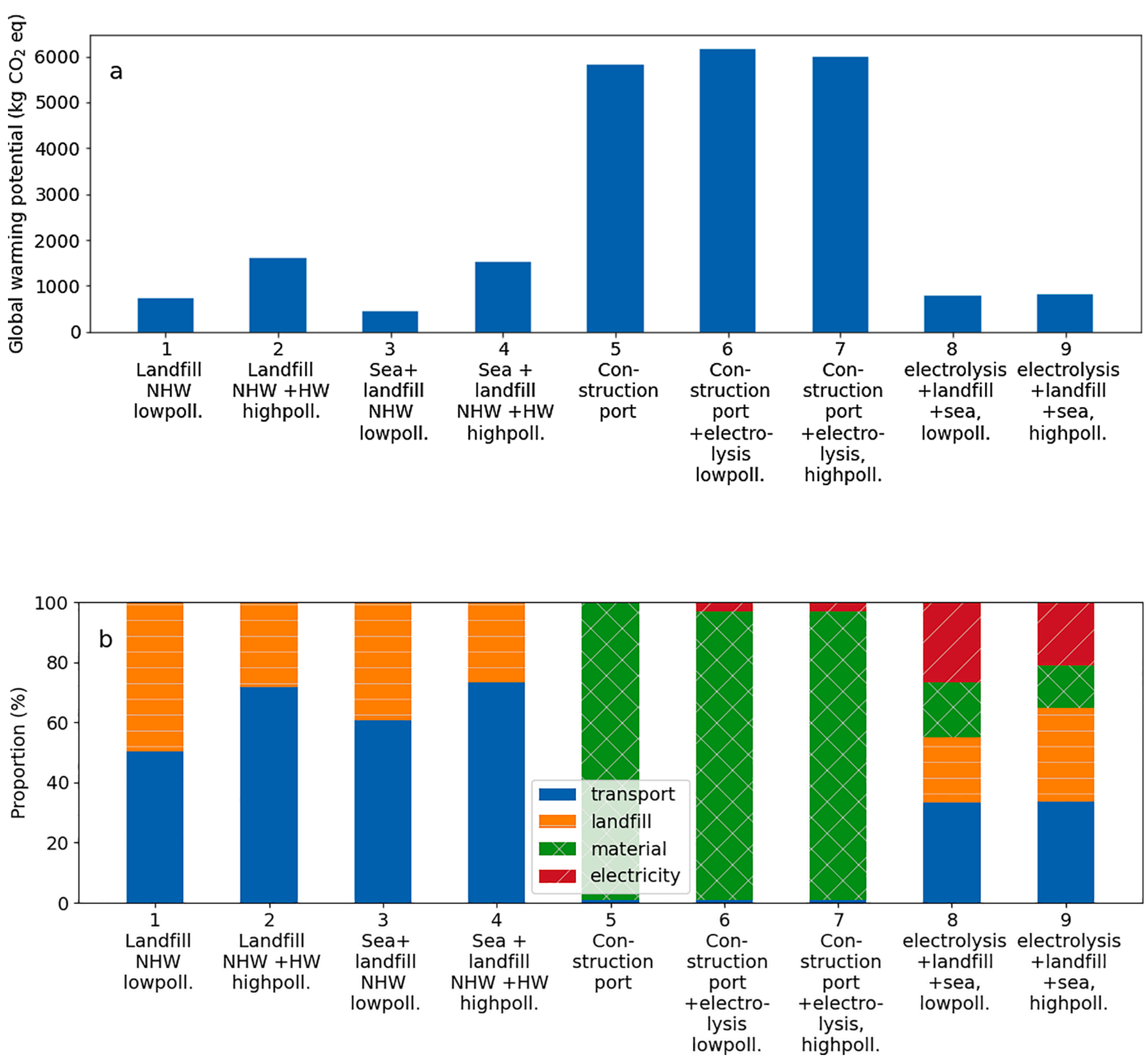

Fig. 3. a) Global warming potential per functional unit (100 m^3) for the different scenarios b) The percentage contribution of different processes to the global warming potential.

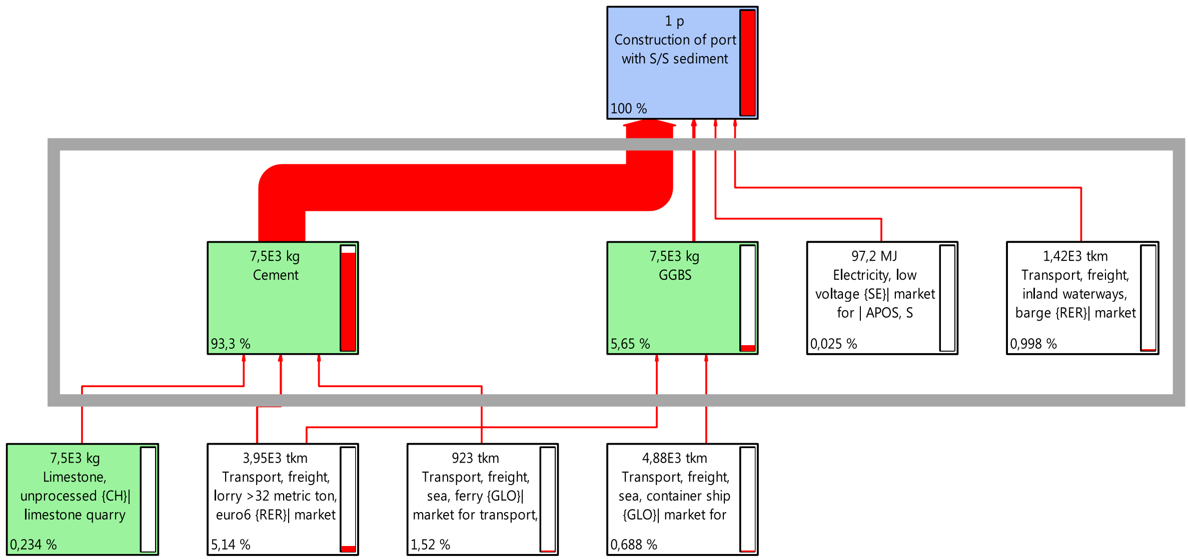

Fig. 4. Example of a process tree, for scenario 5: port construction. The thickness of the arrows shows the relative GHG emissions. The grey box shows the top-level processes, which the classification in Fig. 3b is based on.

and a worst-case estimate has been used for the production of electrodes. In Fig. 5, the bar denoted *Electrolysis, Al* shows the GHG emissions for the default setting, using aluminum electrodes with a speed of 1 m^3/day which must be replaced after electrolysis of 1000 m^3 sediment. If the speed is doubled, or the electrodes have to be changed after half this time, the emissions are doubled. However, using new electrode types that may become available, here exemplified by steel, the emissions decrease by 36% and become approximately the same magnitude as emissions from conventional metal extraction. Another possibility is to use metals from recycled waste for the electrode material, which could decrease the carbon dioxide footprint considerably. There are also possibilities for reconditioning or reuse of the electrode material. The value of the electrolysis is also dependent on the removal efficiency of metals. We have assumed an efficiency of 60 %, but the error bars in Fig. 5 also show the effect of efficiencies from 30 to 90 %.

Based on the assumptions used in this study, the electrolysis would also take a prohibitively long time. The method must be developed and upscaled before it can be used on the amounts of sediment considered here. For the electrolysis to be feasible in terms of GHG emissions, the selection of electrodes is crucial, but there is also a need to improve the effectiveness of the electrolysis. Further, it would only be relevant to extract metals from highly contaminated sediment. This is clearly not the case for Gothenburg but may be meaningful in sediment with higher metal content such as in Oskarshamn (Fathollahzadeh et al., 2012). Whether there is a positive effect on the GHG emissions from electrolysis depends on several factors; the sediment characteristics and metal content, the extraction efficiency, the electrode material etc. Therefore it is hard to set a threshold above which the electrolysis is meaningful.

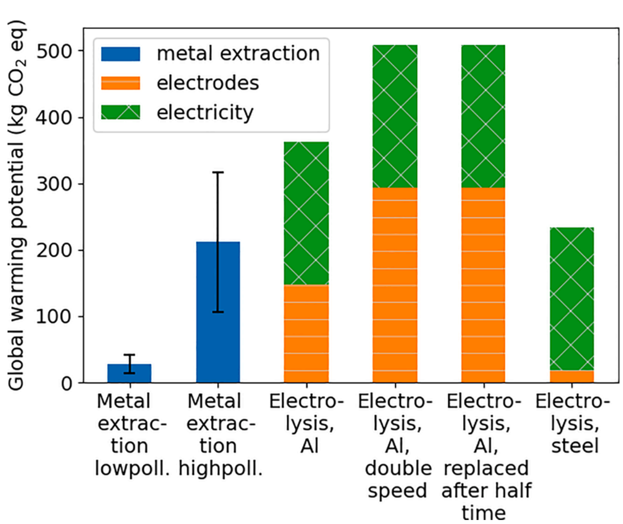

Fig. 5. Global warming potential per functional unit (100 m^3) for conventional metal extraction and electrolysis with different electrodes.

One implication of using electrolysis for sediment remediation is that it enables reclassification of the sediment from hazardous to non-hazardous waste (scenario 9). As a result of the decreased transport distance, less energy-intensive landfill and the re-use of metals, the GHG emissions decrease by 47% compared to not performing the electrolysis (scenario 4). Apart from the assumptions regarding the electrolysis, the improvement in global warming potential is dependent on the transport distance. Assuming e.g. that the landfill for hazardous and non-hazardous waste were located at the same distance, the global warming potential of scenario 9 would be approximately equal to scenario 4. On the other hand, with electrolysis (and possibly other pretreatment methods) the sediment could become clean enough for sea disposal, which would decrease the GHG emissions by 68 % compared to scenario 4. If the sediment does not contain any hazardous waste, no positive climate effect will be achieved by extracting metals using electrolysis.

3.2. Additional environmental impacts

The combined results of the LCA,the scoring of other environmental effects and the cost evaluation are summarized in Table 2. The motivation for the scoring is discussed below, and a more detailed account of the scoring, including references, is available in Supplementary Table S4.

Deep-sea disposal combined with landfill produces the lowest GHG emissions (Table 2, scenario 3), however this method may have undesired impacts on the local terrestrial and marine environment. The effects of landfilling depend on the contaminant content of the sediment, as the masses may pose a risk to terrestrial biota and human health (Camerini and Groppali, 2014; Tribot et al., 2018). Contaminants may spread to the surrounding area through leachate. Other risks include gas formation and management activities at the landfill (Suer and Andersson-Sköld, 2011). The landfill does not only affect the environment during the time it is active, but continues to do so when it is no longer in use. Land use at the site may be restricted long after its active time and

Table 2
The CO_2 equivalent (CO_2 eq), the short and long-term environmental impacts of different management alternatives and the cost estimates for each scenario. Higher negative (positive) numbers imply environmental burden (credits).

	CO_2 eq	Impact on marine organisms	Landuse impact	Air quality	Impact on terrestrial biota/ health	Other potential risks	SUM	Costs (USD/100 m^3)	Revenue (USD/100 m^3)
Scenario	(kg/100 m^3 sediment)	(Short-/long-term)	(Short-/ long-term)	(Short-/ long-term)	(Short-/long-term)	(Short-/ long-term)	(Short-/ long-term)		
1 Landfill NHW (lowpoll)	731	0/0	−2/-1	−3/0	−1/−2	−2/−1	−8/−4	13,600	–
2 Landfill NHW + Landfill HW (highpoll)	1600	0/0	−2/−1	−3/0	−1/−2	−2/−1	−8/−4	14,400	–
3 Sea disposal + Landfill NHW (lowpoll)	442	−1.6/−0.5	−1/−0.5	−3/0	−0.5/−1	−2/−0.5	−8/−2.4	6600	–
4 Sea disposal + Landfill NHW + Landfill HW (highpoll)	1530	−0.3/−0.1	−1.8/−0.9	−3/0	−0.9/−1.8	−2/−0.9	−8/−3.7	12,900	–
5 Construction port (lowpoll)	5832	−1/−3	3/3	1/0	0/0	−3/0	0/0	3600	2300
6 Construction port + electrolysis (lowpoll)	6167	−1/−3	5/3	0/0	0/0	−4/0	0/0	23,600	2400
7 Construction port + electrolysis (highpoll)	5877	−1/−3	5/3	0/0	0/0	−4/0	0/0	23,600	3000
8 Sea disposal + Landfill NHW + electrolysis (lowpoll)	777	−1.6/−0.5	0/−0.5	−3.5/0	0/0	−2/0	−7/−1	26,600	2200
9 Sea disposal + Landfill NHW + electrolysis (highpoll)	714	−0.3/−0.1	0/−0.9	−3.9/0	0/0	−2/0	−6.2/−1	32,000	2900

the long-term release of contaminants have not been fully investigated (Kjeldsen et al., 2002). Clean masses pose lower risk than more polluted sediments (Suer, Andersson-Sköld, and Andersson 2009).

For the deep-sea disposal of masses, the deposition criteria for the Vinga site have been set to be equal to the content already present here. For this reason, it is not expected that sediment disposal will lead to increased contamination at the site. However, it is likely that contaminants will spread during the disposal, although this could be minimized by using techniques such as silt curtains. Disposal of sediment will have a significant impact on the biota at the site, in particular for certain benthic organisms (OSPAR Commission, 2009; Witt et al., 2004). Studies at deep-sea disposal sites have indicated that, provided dumping is only carried out occasionally, both fauna and flora recover after a few years (Donázar-Aramendía et al., 2020; Wilber et al., 2007). With time, the site is expected to recover, although it may never return to the same state as before being used for disposal (Bolam and Rees, 2003; Donázar-Aramendía et al., 2020). Recurrent disposal, and a less than optimal design, may however result in long-term effects (Lee et al., 2010; Stronkhorst et al., 2003; van der Wal et al., 2011). The scoring (Table 2 and Table S4) is based on the assumption that deep-sea disposal causes severe impacts at a site, however the site investigated in this study is an old deep-sea disposal site, why no significant change in the marine morphology was identified and is therefore omitted from the evaluation. Changes in morphology must, however, be investigated for generic purposes, as mass disposal will change the morphology permanently and may affect currents and sedimentation processes. The vessel used to transport the masses does not only impact the release of GHG (which is captured in the LCA), but also the release of particulate emissions, NO_x, sulfur, and leaching from antifouling paints on the barge (Turner et al., 2017).

The options with the highest climate impact were all associated with the port construction, despite the fact that the short and long-term environmental impacts were all found to be neutral (0) for these options (Table 2, scenario 5–7). Stabilization and solidification enable the use of masses for construction purposes. It also prevents excavation of virgin resources, such as gravel, thereby removing the environmental impact of the excavation that would otherwise occur (Table S4). By using wet sediment instead of dry masses in the construction, the risk of releasing particulate matter is decreased (Azarov et al., 2017). In the longer term, the construction enables land use, but marine organisms at the site are greatly affected, both during the construction and because the area will be unable to recover. There is also a risk that contaminants, such as TBT and metals, are leached from the construction site (Norén, 2021b; Zhang et al., 2020). Highly polluted sediment may also leach more pollutants into the surrounding water. Leaching will reach a peak during the construction phase and decrease over time, although some studies have claimed that the leaching decreases considerably even in a short-term perspective (Couvidat et al., 2016). The severity of the leaching could be managed by using e.g. silt curtains to trap suspended particles and pH adjustment of any alkaline water being released during the construction phase. The leaching of metals, such as copper and zinc, could also be reduced by electrochemical pretreatment of the sediment before stabilization (Norén, 2021b). The use of electrolyzed sediment in construction may have additional negative impacts on the working environment. Despite the negative aspects, these options also contribute positively to the impact assessment. For instance, virgin resources may be saved if precious metals are extracted from the sediment. The need to produce more construction materials would also be reduced, and the use of landfills may decrease. Additionally, it may be possible to use the structure for other purposes than as a terminal surface in the future.

However, if the port area needs to be reconstructed in future, regulations relating to the management of the material may apply. Pretreatment of the masses may be a solution to reduce the risk of leaching during construction and to minimize the potential end-of-life restrictions. A risk that relates to the electrochemical treatment and must be managed, is the possible formation of chlorine gas, due to the presence of chlorides in marine sediment. Metal extraction through electrolysis would reduce the toxicity of the sediment, however even after treatment the sediment may still be toxic to some species, e.g. copepods (Benamar et al., 2019). In the investigated scenarios, the treated masses were landfilled or used for construction, and the effects on biota are therefore deemed to be low. So far, metal recovery has not been implemented in any large-scale dredging operations (Akcil et al., 2015), but experimental studies on other materials, such as soil and ashes, have shown that reducing the metal content prior to landfilling has benefits compared to landfilling without this treatment (Schlumberger et al., 2007; Volchko et al., 2017). If electrolysis was performed on masses before landfilling, metals occurring in the sediments could be recovered (Anna Norén et al., 2021a), thereby saving virgin resources and reducing the need for mining, which is often associated with high environmental impacts. The environmental impact of electrolysis depends partially on the origin of the electricity. If renewable energy sources, such as wind power, are used, the impact is considered to be low (Vocciante et al., 2021). The more polluted the sediment, the more can be gained from this approach. In ports with large dredging operations much could be gained if heavily polluted sediment could be cleaned and disposed of in landfill for non-hazardous waste or disposed at sea.

To sum up, the results from the environmental impact assessment (LCA + scoring) highlight the complexity of assessing different impacts and the benefits of showing how different scenarios and parameters interact. As an example; deep-sea disposal can have a negative effect on marine biota, but is beneficial for terrestrial organisms and has a low climate impact. This way of combining quantitative and semi-quantitative data for environmental assessment is a useful approach to be able to include all important aspects, even where detailed data on certain aspects are missing. The climate impact of various products, transportation, and waste management options are fairly well described in available databases. Other important environmental aspects can be harder to quantify from these databases, either because they relate specifically to the project or site in question, or because data is scarce. For example, landfilling and sea disposal may affect the environment differently in different geographical regions, air pollution will have a greater negative impact in densely populated areas, certain natural habitats may be very sensitive to disturbances, etc. The scoring matrix describes these aspects in a structured way, taking local impacts into account, thereby complementing the LCA in the early-stage decision-making process. The combination of quantitative and semi-quantitative data also highlights possible conflicting aspects and can be used to identify where new, innovative solutions are needed, e.g. the use of CCS or using algae for storing carbon dioxide.

3.3. Cost evaluation

The results from the cost evaluation (Table 2) shows that scenario 5 (port construction) is the most cost-effective option. However, there is a large range in cost estimates of S/S in literature, many of which are higher than the chosen 36 USD/m^3. The second least costly option is scenario 3 (sea disposal + landfill NHW). The scenarios 6–9, involving electrolysis, are the costliest ones, even when taking the potential revenue of retrieved metals into account. However, electrolysis is a novel method with very limited experiences of upscaling. Optimization of the setup (electrode material and configuration, chemicals used etc.) for the sediment in question will have to be done to find the most cost-effective and efficient method for metal retrieval. The metal revenues are also likely to keep increasing within the near-future due the electrification of the transport system, creating a high demand for various metals.

4. Conclusions

An integrated cost and environmental impact assessment has been performed on different management options for sediments with high and low levels of contamination, respectively, in the port of Gothenburg, Sweden. The climate impact was investigated using a comparative LCA approach, and showed that if sediments are sufficiently clean, deep-sea disposal has the smallest climate impact. If the sediments are more polluted, metal extraction can be used to clean them and recover any metals they contain, thereby decreasing the climate impact. Using the sediments as construction material for new port areas resulted in the highest GHG emissions by far, due to the carbon dioxide intensive cement production.

The scoring of other environmental aspects showed that the impacts vary, depending on whether a short or long-term perspective is adopted. Using sediment for construction had the lowest environmental impacts, both from a short and long-term perspective, however these scenarios were associated with the highest GHG emissions. This highlights the complexity of comparing environmental impacts, and the benefits of performing an integrated assessment, which gives clarity and highlights both the synergetic and counteracting effects associated with the investigated scenarios. Electrolysis was deemed to be a suitable option if the sediments were highly polluted.

The cost evaluation showed that electrolysis, being the newest technology, also was the costliest and will have to be optimized in order to become a feasible option, while port construction with S/S sediment already is a cost-effective solution.

Therefore, suggestions for future studies include investigating how the metal removal efficiency of electrolysis can be increased, and how the environmental impact of this technique can be reduced in terms of the choice of material for electrodes.

This type of integrated assessment, taking into account several, contrasting aspects is important for early-stage decision-making, to illustrate the combined effect of complex environmental and cost impacts.

Declaration of Competing Interest

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

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Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.wasman.2021.11.031.

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