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An overview of operations and processes for circular management of dredged sediments

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Keywords

Circular economy; dredged sediment; heavy metals; organic pollutants; sediment treatment.

Abstract

Dredging is an essential technique to maintain proper water depths in ports and bays. Many dredged sediments are considered as toxic waste due to their significant amounts of metals and other pollutants. In consequence, they need to be treated to reduce this toxicity and avoid pollutant resuspensions. Physical operations and chemical, thermal and biological processes have been conventionally used to this aim, but the traditional linear sediment approach is often unsustainable and economically and environmentally demanding. Considering the increasing people's awareness in environmental issues, more efficient dredged sediment management schemes are required. Some authors are making significant efforts to improve circularity in sediment management processes by taking advantage of the mineral composition of sediments to obtain products for the building and road construction sectors, therefore decreasing the need of raw materials while reducing the amounts of sediments wasted to landfills. However, information related to the characteristics of these products, their mechanical behaviour and their functionality is still scarce, being sediment-based by-products developed mainly at low Technological Readiness Level (TRL), showing low global impact in the market. To implement circular economy in the dredged sediment sector, some technical and socio-political barriers must be still overcome. To this aim, further research and technological applications must be developed, with the support of decision makers and stakeholders. This review aims at giving an overview of the circular trends applied to toxic dredged sediment management, pointing at current opportunities, barriers and constraints that hinder its wide development.

1. Introduction

Sediments are solid particles of sand, silt, clay and other substances that result from the erosion of rocks, soils and other solids. They are one of the main elements of aquatic ecosystems and serve as habitat and food resources to water life (Bortone and Palumbo, 2007). Sediments are transported by natural and human activities and settle at the bottom of water bodies, which may hinder boat navigation and the physicochemical balance of water masses (Ferrans et al., 2019; Mehdizadeh et al., 2021; Pal and Hogland, 2022). Moreover, sediment accumulation in dams can significantly decrease their water storage capacity (de Vincenzo et al., 2018). To maintain adequate depths in water ways and dams, regular dredging is needed (Chen et al., 2019; Norén et al., 2020). Apart from maintenance, there are other dredging activities such as the construction of new waterways or remediation dredging (Bortone and Palumbo, 2007). All these practices imply the production of large amounts of dredged sediments, accounting for around 300 million of m^3 per year only in Europe (Snellings et al., 2016). Dredged sediments have been globally considered as a waste to be got rid of (Beddaa et al., 2020), being many of them considered as toxic due to their high pollutant concentrations (Section 3). Sediments categorised as toxic by current regulations (Section 4) are only allowed to be either directly disposed in landfills for toxic wastes or to be treated to reduce their toxicity to admissible levels.

Considering the high amounts of dredged sediments produced worldwide and the increasing legal requirements in terms of sediment management, the “no action” scenario as well as conventional practices such as direct disposal in confined facilities are no longer feasible economically, environmentally and socially (Barjoveanu et al., 2018; Mehdizadeh et al., 2021; Pal and Hogland, 2022; Pellenz et al., 2020). Sediment dredging therefore represent a challenge for port authorities (Loudini et al., 2020b), which are usually open to a new approach of sediment management that help them to cope with this environmental problem. In this respect, increasing number of studies are trying to apply Circular Economy (CE) principles to the dredged sediment management sector, i.e., to transform management processes to consider sediments a source of sustainable resources instead of a toxic waste (Section 6).

This review aims at giving a new insight to conventional dredged sediment management, highlighting the novel trends that aim to manage dredged sediments according to circular economy principles and pointing at current opportunities, barriers and constraints that hinder its wide development. Previously, an overview of sediment characteristics, current legislation and the conventional treatment processes to reduce the toxicity of sediments will be given.

2. Methodology

The study is divided in the following sections: 1) an Introduction section where current generic issues related to sediment management are addressed; 2) the description of the methodology followed in this manuscript; 3) the analysis of the main characteristics of dredged sediments, focusing on those that make them be considered as toxic wastes; 4) as this consideration is also dependent on specific regulations, these are also reviewed; 5) the evaluation of operations and processes to treat toxic dredged sediments, which has been conventionally carried out in a linear approach; 6) the novel approach of sediment management based on circularity, which not only focuses on reducing pollutants but also on decreasing impacts of treatment processes while recovering by-products. Some pilot cases developed in research projects are described. Socio-political barriers for the development of this approach are also evaluated in this section.

To develop this study, a deep search on the Elsevier Scopus and Google Scholar database was carried out using the following keywords as search queries: “dredged sediment”, “sediment treatment”, “dredged sediment circular economy” and “dredged sediment treatment process”. Generally, results were limited to the last five years (2017-2022) to give an updated approach, especially for the literature related to circular economy in sediment management (Section 6). However, since the conventional dredged sediment treatment has been widely investigated during last two decades, for the analysis of the linear treatment approach (Section 5), the searching time period was lengthened to the last twenty years. Other criteria for the selection of the literature were related to: the quality of the results, assessed by the impact factor of the journals and the number of citations; articles related to sediment treatment and management on site (without dredging) were excluded; the studies analysed tried to cover as many regions as possible to make an overall overview of current situation; articles based on process engineering were prioritised while others which focus on aspects such as pollutant transport between sediments and water or toxic effects of sediment pollutants in wildlife were discarded.

3. Characteristics of dredged sediments

Marine dredged sediments contain quite variable characteristics, even in zones that are close to each other (Table 1). These differences depend on many factors such as the intrinsic characteristics of the raw materials that form the sediments, the degree of erosion, the economic activities in the surroundings, and even the dredging time at which sediments are produced (Safhi et al., 2020). Hence, adequate sediment characterisation is essential to assess the most appropriate sediment management process for each case.

Generally, dredged sediments contain high amounts of salts (Doni et al., 2018; Wang et al., 2018a) and water, which is present in high variable range, i.e., 24-90% (Zhao et al., 2022). A significant part of this water is difficult to be removed as it remains bound to sediments by electrical charges (Chi et al., 2018). These characteristics tend to hinder sediment treatment operation and processes (Section 5). In addition, sediments are commonly dominated by silt and clay fractions, which normally account for 60-90% of the solid content of sediments (Ferrans et al., 2021; Wang et al., 2018b, 2018a). These fractions, due to their higher specific surface, tend to accumulate more pollutants than coarser particles like sand and gravel (Anand et al., 2021; Mymrin et al., 2017; Todaro et al., 2018; Yoobanpot et al., 2020). Furthermore, sediment pH tends to be slightly basic, i.e. around 8.0-8.8 (Doni et al., 2018; Wang et al., 2018a; Zhou et al., 2022). Consequently, many sediments present significant concentrations of precipitated metals (Debnath et al., 2021; Pal and Hogland, 2022; Todaro et al., 2016; Wang et al., 2018b). Metals such as copper, zinc, cadmium, chromium, nickel, plomb, and arsenic are often found in dredged sediments in the order of tens and/or hundreds of mg per Kg (Table 1). In the case of iron, the concentration in sediments can reach the order of $g \cdot Kg^{-1}$ (Fonti et al., 2013; Lopez et al., 2022). Many authors have also found hydrocarbons and other organic pollutants in sediments also in the range of $g \cdot Kg^{-1}$, including TBT, TPH, PAHs, and PCBs (Table 1), as well as significant amounts of nutrients (Doni et al., 2018; Maletic et al., 2019; Ferrans et al., 2021). These pollutants can be resuspended and re-enter to water bodies, often entailing serious environmental, health and ecological issues due to the persistence of these pollutants and their negative effects on living organisms and water quality (Barjoveanu et al., 2018; Debnath et al., 2021; Liu et al., 2022; Maletic et al., 2019; Norén et al., 2020; Zhang et al., 2021). Appropriate management of toxic sediments is therefore essential to maintain adequate environmental quality and economic activities in ports, harbours and bays, underlying the need for clear legislation that regulates their possible uses and applications.

[TABLE 1 NEAR HERE]

4. Regulations regarding dredged sediments

At European level there are directives, laws and regulations where sediments are mentioned directly or indirectly, also considering sediment management (Bortone and

Palumbo, 2007). They indicate, according to their maximum pollutant levels and their mobility, which sediments are considered as toxic waste and the specific applications to which sediments are allowed (Buceta et al., 2015; Ferrans et al., 2021). These European guidelines and conventions have been adopted in numerous countries (in Europe and beyond) but there is a lack of uniformity and coherence in them in terms of characteristics to be analysed, limits of pollutant concentrations, categories of classification of sediments, etc. (Jersak et al., 2016; Sapota et al., 2012), as can be seen in Table 2. By means of example, in Italy, the legislative framework about sediment management (DM 173, 2016) regulates the sediment ocean disposal, their use for nourishment and backflow activities for limited environments. The choice of management options is dictated by the sediment quality class (sediment type A, B, C, D and E), which is determined through criteria of integration weighted by both ecotoxicological and chemical hazard class. In particular, the ecotoxicological classification is based on a judgment of ecotoxicological risk elaborated by the weighted integration of the results of all the components of the entire battery of bioassay tests. The chemical classification is based on the development of a chemical Hazard Quotient (HQ) index which considers the type and number of non-compliant parameters, as well as the extent of such exceedances (DM 173, 2016). When the sediments are directly or indirectly (i.e., after treatment) disposed or used to obtain by-products, these final products have to be also submitted to release tests to assess the pollutant mobility (Bortone and Palumbo, 2007; Zhao et al., 2022). The leachates that come from a release test in water of the matrices for a period of 24 hours cannot overpass the thresholds reported in DM 5th February 1998.

In the case of France, the decision on sediment suitability for sea disposal is primarily based on chemical findings, being the ecotoxicological analyses only advised to be done on guidelines. According to the concentration limits of the required chemical parameters (such as particle size, metals, trace elements and organic micro-pollutants), N1 (high quality) and N2 (medium quality) levels are legally defined. Only in the cases where any of the compounds exceeds the N2 threshold, a biological characterisation of sediments is needed (Mugnai et al., 2018; Tessier et al., 2011).

On the other hand, in Spain, sediment characterisation is carried out in two steps: one preliminary step to analyse their potential toxicity and a secondary step based on chemical and biological characterisations. If the dredged material presents silt-clay fraction lower than 10%, total organic carbon (TOC) concentration lower than 2%, and EC50 of the bacterium *Vibrio fischeri* higher than 2000 $mg \cdot L^{-1}$, they are not required to be submitted to the second step where some target metals, hydrocarbons and phenols are analysed (Buceta et al., 2015). If sediments overpass the defined limits in one of the target pollutants, they will be defined as toxic waste and have to be managed according to Law 22/2011 *on Contaminated Wastes and Soils*. Non-toxic sediments are classified in three different categories according to their quality in terms of chemical concentrations: A (highest quality), B, and C (lowest quality). Only type-C sediments are submitted to biological characterisation (Buceta et al., 2015).

[TABLE 2 NEAR HERE]

The degree of restriction of the legislation will determine (amongst other factors) the level of treatment to which sediments will be subjected and, consequently, the first step to develop the appropriate management of dredged sediments is to analyse their chemical and physical characteristics (Ferrans et al., 2019; Zhou et al., 2022).

5. Dredged sediment treatment operations and processes

Non-toxic sediments can be directly used in land for soil filling, construction purposes, coastal nourishment and as an amendment in agriculture, horticulture, and forestry as long as they comply with the pollutant-specific regulations or are classified in the appropriate category (as explained in Section 4). In these cases, they could be also disposed directly in landfills or oceans, which are common and simple techniques but present considerably high environmental impacts (Akcil et al., 2015; Barjoveanu et al., 2018; Bhairappanavar et al., 2018). In case of landfilling of toxic sediments, apart from the huge land requirements, dredged sediments can be only disposed in facilities that present appropriate soil characteristics and rigorous dumpsite management to assure their safe deposit, avoiding toxic emissions (Mehdizadeh et al., 2021; Pal and Hogland, 2022; Pellenz et al., 2020; Wang et al., 2019). There are a reduced number of these landfills for toxic compounds in comparison to facilities for inert materials, which not only increases the difficulty of managing the dredged sediments, but also raises the treatment and transport costs drastically (Bhairappanavar et al., 2018; Carpenter et al., 2018; Kim et al., 2016). In fact, while landfill costs for inert sediments is around 27 $\$\cdot t^{-1}$, they increase up to 67-80 $\$\cdot t^{-1}$ when wastes are toxic (Norén et al., 2020). Considering these aspects together with the increasingly restrictive legislation in terms of dredged sediment management and general public's awareness in environmental issues, direct disposal and confinement of sediments is not recommended (Todaro et al., 2016; Zentar et al., 2009).

To reduce the toxicity of dredged sediments in order to dispose them afterwards (minimising toxic risks) or to obtain by-products from them (Section 6.1), several physical operations (usually) combined with chemical, thermal or biological processes have been developed along last decades (Maletic et al., 2019).

This review will mainly deepen in treatment operations and processes which are commonly used to treat toxic sediments ex-situ. In-situ sediment remediation processes such as capping or electrokinetic remediation (Benamar et al., 2019; Debnath et al., 2021; Maletic et al., 2019) will not be taken into account since no dredging is needed.

5.1. Physical operations

Physical operations aim to separate sediment fractions according to their size and to reduce the content of water and pollutants in sediments through mechanical separation, dewatering and washing (Debnath et al., 2021).

5.1.1. Mechanical separation

Mechanical separation allows dividing dredged sediments into different fractions according to their size (e.g., sand, silt and clay) or to their mineral composition based on differences in the particles' specific weights or in the conditions of the particles' surfaces (Hakstege, 2007). It can be carried out by centrifugation, screening, flocculation, sedimentation or by using hydrocyclones, spirals or combinations of these techniques and technologies (Kim et al., 2016; Mulligan et al., 2001; Pal and Hogland, 2022). Since the smaller sediment particles (silt and clay) retain higher amounts of metals and organics due to their larger surface-volume ratio as aforementioned, mechanical separation also enables to concentrate pollutants into these fractions (Pal and Hogland, 2022). This can help to reduce the amounts of toxic wastes to be managed as long as the coarser fractions accomplish with the authorities' requirements. The treatment costs for sediment separation are in the range of 3 - 11 €·m^{-3} (ref. year 2007), depending on the operation, scale and local conditions (Hakstege, 2007).

5.1.2. Dewatering

Dewatering consists of the removal of water from sediments to reduce their volume and, in turn, to decrease the costs of downstream treatments. Dewatering can be carried out by natural drainage and evaporation (Bortone and Palumbo, 2007) but these techniques require vast space and long retention times, while the degree of dewatering is limited. To increase dewatering performance and reduce land demand, mechanical units such as filter presses, draining screens, vacuum filters, or membranes could be used. By way of example, dry matter content of sediments can increase from 40-45% to 65-80% by using filter presses. The costs of this operation vary in the range of 10-35 €·m^{-3}(ref. year 2007) (Hakstege, 2007). However, when the organic content of sediments is high (especially in terms of extracellular polymeric substances (EPS)), more water is bound to sediments, hindering dewatering. To release the interstitial water and facilitate its removal, chemical conditioners such as iron and aluminium salts, or organic polymers like polyacrylamide (PAM) and poly dimethyldiallylammonium chloride (PDMDAAC), can be used (Chi et al., 2018; Zhao et al., 2022).

It must be highlighted that mechanical separation and dewatering are not able to reduce the toxicity of the sediments significantly. They mainly concentrate pollutants in certain fractions to reduce the volume of wastes to be treated. For this reason, these operations often need to be combined with other processes to assure the safe use of the products obtained from sediment or their safe disposal (Hakstege, 2007).

5.1.3. Washing

One possibility to reduce the pollutants contained in sediments is washing. This operation can simply consist of mechanical washing with water or can be improved by adding chemicals such as acids, bases, chelants, surfactants, and others (Debnath et al., 2021; Ferrans et al., 2021; Pal and Hogland, 2022; Polettini et al., 2009). Acid washing is usually effective to remove heavy metals with their subsequent release to the washing solution (usually water) (Mulligan et al., 2001). Washing also extracts salts from marine sediments (Doni et al., 2018; Todaro et al., 2020), which is usually beneficial for downstream processes or for possible applications of the sediments treated. Strong acids such as nitric, hydrochloric and sulfuric acids can be used to reach the acid conditions in the washing solution (Löser et al., 2007). Ferrans et al. (2021) also studied metal extraction with weaker acids (i.e., chelating agents which are less abrasive to sediments) such as ethylenediaminetetraacetic acid (EDTA) and ethylenediamine-disuccinic acid (EDDS), obtaining extraction efficiencies in the range 29.7 – 74.1% for lead, zinc, copper arsenic and nickel. However, chromium was scarcely extracted (2.1-3.1%).

The performance of each acid is variable and should be thus a matter of study as it depends on several variables such as chemical dosage, contact time, liquid-solid ratio, pH, temperature and the number of extraction steps (Beiyuan et al., 2018; Ferrans et al., 2021). Washing is usually more effective when applied in coarser particles, being smaller fractions more difficult to clean (Hakstege, 2007; Jeon et al., 2015). For this, the washing process can be more effective if it is preceded by a mechanical separation step. Anyhow, acid washing operations are quite expensive, being in the range of 40–73 $\$\cdot\text{m}^{-}$

^{3} (ref. year 2009) (Pal and Hogland, 2022; Tsai et al., 2009), which is something to take into account when the sediment treatment scheme is designed.

5.2. Chemical processes

Chemical treatment includes all the sediment treatment processes where chemical reactions occur, for instance, dechlorination, oxidation, chemical stabilisation and coagulation (Debnath et al., 2021; Kim et al., 2016). They are applied to sediments with the purpose of improving the stabilisation of the final by-products obtained from sediments and/or to reduce the toxicity and bioavailability of pollutants (Hakstege, 2007; Todaro et al., 2018; Wang et al., 2018a).

5.2.1. Chemical oxidation

Chemical oxidants such as ozone, potassium permanganate, hydrogen peroxide, Fenton's reagent, and activated sodium persulphate can be effective to transform the organic pollutants contained in sediments into non-hazardous compounds. In fact, Ferrarese et al. (2008) reported PAH degradations over 95% when Fenton's reagent, hydrogen peroxide and potassium permanganate were used at dosages of 100 mmols per 30 g of sediments. However, removal efficiencies are highly variable depending on the reagent used, while the optimum dose should be determined for each specific case (Ferrarese et al., 2008; Maletic et al., 2019). It must be noted that chemical oxidation is not commonly efficient to degrade metals (Bortone and Palumbo, 2007; Mulligan et al., 2001).

5.2.2. Solidification/Stabilisation (S/S)

Chemical reagents such as lime, silicates, cement and others can be added to polluted sediments with the goal of immobilising and stabilising the metals contained in the dredged sediments by increasing the compressive strength of the matrix and decreasing their leachability, thus becoming less mobile (Kou et al., 2021; Maletic et al., 2019; Radenovic et al., 2020). This way, even if metals are not removed, the risk of being released to air, water and soil is significantly reduced. This process is commonly known as solidification/stabilisation (S/S) (Barjoveanu et al., 2018; De Gisi et al., 2020: Hossain et al., 2020; Radenovic et al., 2020; Tang et al., 2020).

As an advantage, S/S is less expensive than other processes to reduce metal toxicity such as those based on thermal processes (Amar et al., 2021), but it has to be considered that the additives used in the process, together with the type of pollutant, water content and characteristics of the dredged material will significantly influence the process performance, and in consequence, the costs and impacts (Wang et al., 2015; Zhou et al., 2022). For instance, arsenic, lead, chromium (VI), mercury, cadmium, copper and zinc are normally stabilised successfully by S/S. However, the stabilisation of other metals could be less effective (Mulligan et al., 2001). In addition, sediments with high saline contents, high amounts of low-size fractions or high organic pollutant concentrations normally present low performance in this type of process (Barjoveanu et al., 2018; De Gisi et al., 2020; Norén et al., 2020; Todaro et al., 2020). Anyhow, continuous monitoring is required as solidification can be reversible (Pal and Hogland, 2022; Tang et al., 2020). Consequently, this process cannot be widely applied to treat toxic dredged sediments and is often combined with other techniques. It must be also considered that S/S processes increase the reagent requirements exponentially when water content in sediments is higher than 20% or when chlorinated hydrocarbons concentrations surpass 5% (Mulligan et al., 2001).

5.3. Thermal processes

Thermal treatments entail the use of high temperature, i.e., in the range of 800 - 1200ºC (Safhi et al., 2020; Zhao et al., 2022), to separate the pollutants contained in sediments by desorption, oxidation or by sintering or melting the sediments to change their composition with the goal to reduce metal mobility, thus decreasing their potential toxicity. However, thermal processes are characterised by being highly energy-consuming (Kou et al., 2021; Pal and Hogland, 2022). Hence, operating conditions (i.e., temperature and time of calcination) must be optimised to decrease costs, impacts and minimise undesirable phenomena that commonly occurs in these kinds of processes such as the reduction of internal porosity, and granulometry modification.

In comparison to chemical processes, thermal treatments usually obtain sediments with better characteristics such as improved studied the pozzolanic reactivity (Amar et al., 2021; Benzerzour et al., 2017; Snellings et al., 2016). However, the gases produced during the process have to be treated to degrade the pollutants that were removed from the sediments, increasing the complexity and costs of the treatment. Furthermore, thermal technologies are prone to breakdowns, which reduces the process reliability (Mulligan et al., 2001; Pal and Hogland, 2022; Todaro et al., 2016). Obviously, water content of sediments will decrease thermal process performances. In fact, water content should not be higher than 30% to be thermically treated (Hakstege, 2007). Pre-treatment to reduce the water content of sediments could be therefore needed for thermal treatment. Not considering pre-treatment, the costs of thermal processes can rise up to 50 – 70 €·t^{-1} (ref. year 2007) (Hakstege, 2007). For all these reasons, thermal processes should be carried out only in specific occasions, being thus minimised in circular management schemes (Section 6).

5.3.1. Thermal desorption

Thermal desorption has been traditionally used to remove volatile organic compounds (VOCs) and other (light) organic harmful compounds due to the differences between the volatility of the pollutants and of dredged sediments (Hakstege, 2007). By applying moderately high temperatures, compounds such as mineral oil, aromatics, PAH's, PCB's, cyanides, chlorinated solvents and TBT can effectively move from sediments' surface to the gas phase (Hakstege, 2007; Todaro et al., 2016). The maximum temperature achieved during thermal treatment will depend on the compounds that are wanted to be removed. For instance, to remove heavy mineral oil, temperatures higher than 450ºC are needed while they need to be raised to 550ºC when the main pollutant is PCB (Bortone and Palumbo, 2007). On the other hand, metals such as mercury, arsenic and cadmium could be removed from sediments at temperatures around 800 ºC (Mulligan et al., 2001). Thermal desorption can be combined with other thermal processes such thermal oxidation and thermal immobilisation (Bortone and Palumbo, 2007).

5.3.2. Thermal oxidation and immobilisation

When high temperatures are applied to sediments, organic pollutants will not only volatilise but will be also burned (thermal oxidation) (Pal and Hogland, 2022). With respect to the inorganic materials contained in sediments, the application of temperatures in the range 820-1100 ºC make compounds like SiO_2, Al_2O_3, CaO, Fe_2O_3, etc., melt. With the subsequent solidification step, metals remain fixed to sediments (thermal immobilisation) (Snellings et al., 2016; Zhao et al., 2022).

Thermal immobilisation can be used to obtain several products like bricks, light weight aggregates (LWA) and others that could be used to implement the circularity of the process (Section 6). In this respect, heating and cooling times are important parameters that have to be controlled to maximise the quality of the final by-product and minimise the energy consumption (Zhao et al., 2022).

5.3.3. Vitrification

Vitrification consists of the use of electricity to destroy and/or immobilise pollutants on sediments. High voltage is applied to the sediments to reach very high temperatures (up to 3000 ºC) in order to turn the sediments into liquid phase. Then, sediments are solidified after cooling down creating a vitreous material where the pollutants that were not destroyed during heating remain fixed (Mulligan et al., 2001; Todaro et al., 2016). This process can be very effective in removing pollutants. However, it is very energetically and environmentally costly, and generally requires intensive pre-treatment. It must be also considered that toxic gases, that have to be treated, can be also produced during vitrification (Colombo et al., 2012; Pal and Hogland, 2022; Todaro et al., 2016).

5.4. Biological processes

Biological processes for sediment treatment (also known bioremediation) consists of the oxidisation of biodegradable pollutants such as PAHs, mineral oil and hydrocarbons to transform them into non-hazardous compounds by the action of microorganisms or plants (Amar et al., 2021; Doni et al., 2015; Feng et al., 2022). This can occur by the natural capacity of indigenous microbes, often requiring huge land surfaces; by biostimulation, which entails the introduction of nutrients and oxygen into the polluted sediments to boost microbial activity; or even by bioaugmentation, i.e., the addition of external microorganisms to the sediments (Doni et al., 2018; Fodelianakis et al., 2015; Wu et al., 2014).

It must be considered that these techniques are generally extensive, so that they present much lower operating costs, but the times required for the degradation are noticeably high (Pal and Hogland, 2022). By way of example, Doni et al. (2018) observed a removal of 46% of total petroleum hydrocarbons (TPH) from raw sediments after three months of biological treatment by adding a bioactivator (a mix of microorganisms, enzymes, and nutrients), while it fell to 15% when natural attenuation was carried out. On the other hand, Liu et al. (2017a) achieved PAH removals of 42-83% after 91 days of incubation by adding acetate as co-substrate and nitrate as electron acceptor. In addition, bioremediation tend to be less predictable than other processes (Maletic et al, 2019). Consequently, industrial applications of biological processes to sediment treatment are limited.

Table 3 shows a summary of the sediment treatment operations and processes described in Section 5. Although the maturity and technical applicability of the single units described is currently reached, they are not usually effective nor efficient (in terms of economic and environmental impacts) to decontaminate toxic sediments on their own. It can be summarised that, generally, physical treatments are mostly used as pre-treatment to separate sediment fractions and reduce water, salinity and pollutant contents to facilitate downstream processes. Chemical treatments are usually effective in terms of reducing toxicity of sediments but depending on the case, large amounts of reagents could be needed. Regarding thermal processes, they are normally effective but highly energy and environmentally demanding, while biological treatments present low impacts in terms of carbon footprint but are not usually cost-effective. Consequently, these treatment operations and processes must be combined to form relatively complex sediment treatment schemes with the goal to look for feasible options to reduce sediment toxicity.

[TABLE 3 NEAR HERE]

6. Circular sediment management and resource recovery

As aforementioned in Section 5, conventional sediment treatment operations and processes usually require high amounts of resources and present significant operating costs and environmental impacts. Hence, a linear sediment treatment strategy which only focus on pollutant removal and toxicity reduction (Figure 1) is often not sustainable (Maletic et al., 2019; Spadaro and Rosenthal, 2020; Zheng et al., 2019).

[FIGURE 1 NEAR HERE]

With the goal to maximise the performance of the sediment treatment, different operations and processes have been traditionally combined in order to adapt the treatment scheme to the sediments' characteristics and composition: water content, metals and organic concentrations, size distribution, etc. The quantitative and qualitative variability of the sediments' characteristics together with the specificity of the treatment objectives of each processing unit make sediment treatment configurations noticeably different from each other. By way of example, the BioGenesis^{SM} sediment washing technology (Figure 2a) was developed in the 1990s and uses physical operations and chemical processes to wash and decontaminate the sediments. This technology has been already implemented in the ports of Venice, New York and New Jersey. The pollutant removal normally varies in the range 60-80%, depending on the sediment matrix, pollutant types and concentrations and the extent of treatment (Hakstege, 2007). The treatment costs are scale-dependant, being reported as 55-80 €·m^{-3} when treating 300,000 $m^3 \cdot y^{-1}$, and 90-130 €·m^{-3} when treating 50,000 $m^3 \cdot y^{-1}$ (Hakstege, 2007). On the other hand, METHA treatment (Figure 2b) for the sediment separation was developed in the Port of Hamburg (Germany) (Bortone and Palumbo, 2007). It is based on a two-stage separation. In the first one, the 63 µm silt is separated from the sand by hydrocyclones and upstream flow classifiers. When the particle size distribution is predominant in the range of 20 - 100 µm, it is advised to also separate the 20 µm fraction. In this case, hydrocyclones and downstream spiral concentrators are used. The products obtained during the separation steps are dewatered using different systems such as draining screens and vacuum filters. The dewatering of the flocculated silt suspension is done in two stages with the application of a belt filter press and a high-pressure press or alternatively with a membrane filter press.

[FIGURE 2 NEAR HERE]

As an alternative to these conventional linear sediment treatment schemes, and (mainly) due to more restrictive environmental regulations and increasing public awareness of sustainability, the way dredged sediments are managed is currently changing to move towards the Circular Economy (CE) principles (Laboyrie et al., 2018; Yoobanpot et al., 2020).

Within the CE concept, the economic activities become auto-regenerative by converting materials traditionally considered as waste into raw matter. This substitutes the conventional model into a more efficient paradigm (Puyol et al., 2017; Spadaro and Rosenthal, 2020). In the case of dredged sediments since 90% of them are considered useful for certain applications (Bhairappanavar et al., 2018), many authors have focused on obtaining by-products from these treated sediments to increase the feasibility of these processes (Section 6.1). This way, an added value can be obtained from a “waste” while the volume of sediments disposed in landfills is reduced (Amar et al., 2021; Mehdizadeh et al., 2021; Yang et al., 2020). But the CE concept not only focuses on taking advantage of the materials contained in wastes. It is also related to the improvement of all production chains to make them more sustainable and environmentally friendly, increasing their energy and cost efficiency and reducing carbon, water and land footprints (Figure 3). In this respect, substituting reagents and binding materials from non-renewable sources (such as cement, sand, lime, clays and others) by those traditionally considered as wastes presents multiple benefits in terms of environmental sustainability since both resource consumption and the impacts of wasting materials are reduced. However, a sustainable and economically efficient scheme to manage polluted dredged sediments have not been yet established (Todaro et al., 2016; Yang et al., 2020). Moreover, most of the processes and configurations aimed at applying CE principles still present low Technological Readiness Level (TRL), most of them having been only tested in lab conditions without evaluating the product(s) obtained on site. Data at industrial or semi-industrial scale and tests in-situ of the obtained by-products are needed to widely implement sustainable dredged sediment management. In this respect, innovative projects such as *SEDI.PORT.SIL* and

ECOSEDRA have being recently developed to enhance circularity in sediment management.

[FIGURE 3 NEAR HERE]

The SEDI.PORT.SIL project was developed to treat sediments from the Port of Ravenna (Italy). The entrance to the pilot plant consists of a mixed of sediment and water which is submitted to washing and dewatering operations. The coarser material (i.e., sand) returned clean and ready to be reused while the fine clay fraction can undergo two different treatments. In the first, a certain amount of clay goes to a plasma fusion plant to extract silicon iron alloys and slag (inert material). The latter is obtained by vitrification through rapid cooling. Heat is also produced (in the form of water at 90ºC). This can be sold to surrounding industrial plants that can benefit from this energy contribution in an industrial symbiosis (IS) approach (Domenech et al., 2019). The second process is fed with the material exceeding the inlet capacity of the furnace, which is finally reclaimed for landfarming. The material is submitted to biological degradation, which make the material be free of organic pollutants (Figure 4a).

On the other hand, *ECOSEDRA* Project emerges by the need for a shift in the dredged sediment management system in the Port of Ancona (Italy). Traditionally, these sediments have been directly disposed in the port’s dumpsite, which is a non-sustainable practice which is no longer permitted. *ECOSEDRA* aims at circular management of these dredged sediments to obtain materials for road construction in a sustainable and efficient way (Figure 4b). The management scheme used in *ECOSEDRA* is formed by widely consolidated processes and operations (for instance, acid washing, dewatering, etc.). The innovation, sustainability and circularity of the Project mainly lies on the following:

i) *ECOSEDRA* develops a sediment management scheme prototype (mid-scale) where the product obtained (road construction material) will be tested on-site to analyse its characteristics as commercial by-product. This will allow obtaining transferable results to large scale.

ii) The management scheme of *ECOSEDRA* will be developed according to the latest Industry 4.0 paradigms, i.e., it will be automatised by a control system in order to look for the most efficient operation and for the minimum carbon footprint during the process.

iii) The application of the CE principles to toxic sediments is usually complicated since many sediment treatment processes require the use of considerable amounts reagents, water and energy (as explained in Section 5). *ECOSEDRA* evaluates the most efficient environmental use of resources, for instance, by reusing the process wastewater and other by-products such as gravel fraction, following a zero-pollution approach (Čavoški, 2020).

iv) The toxic sediments from Ancona port will be used for producing road materials since the human exposure to these materials is much lower than if they were used in the construction sector (Section 6.1.2). In consequence, possible long-term risks (that have not been yet tested) would be minimised. Other uses such as sediment-based embankments will be also evaluated.

v) To approach circularity during the management process, it is important to mix the treated dredged sediments with low-carbon recycled materials such as those exposed in Table 4, or others like road milling material or demolition rubble.

vi) The results will be analysed considering all the relevant parameters of the sediment management process, i.e., economic, environmental, technical, social and risk-based factors.

[FIGURE 4 NEAR HERE]

6.1. Products obtained from sediments

6.1.1. Sediment fractions: sand, silt and clay

When dredged sediments present good quality, i.e., they contain low amounts of pollutants (according to corresponding legislation), they do not need to be treated or just need to be submitted to slight treatment. In these cases, sediments can easily be fractioned in sand, silt and clay. Sand appears as a valuable material in construction and can be also used as filling or filtering material. It is the most profitable fraction of sediments and normally present the best quality in terms of contamination (Doni et al., 2018). Silt has a market as sealing material or secondary raw material in the ceramic industry; while clay, apart from being used in the ceramic industry, can also act as filling material (Hakstege, 2007; Wang et al., 2018a). However, as commented before, vast amounts of dredged sediments worldwide contain pollutants that usually hinder the direct use of sediment fractions, especially in the case of the smallest ones (silt and clay) as they tend to accumulate most of the pollutants (Doni et al., 2018; Zentar et al., 2009).

6.1.2. Construction materials

Due to their mineral compositions, sediments can be used as a binder in the production of construction materials such as bricks, LWA, and cement with the goal to save construction costs (Loudini et al., 2020b; Maletic et al., 2019; Mehdizadeh et al., 2021;

Zhao et al., 2022). To do this, sediments generally need to be submitted to a combination of processes and operations since the presence of pollutants or the inappropriate granulometry could significantly decrease the quality of the final product by reducing its strength and durability (Ferrans et al., 2019). In this respect, S/S (Section 5.2.2) is a common process to obtain building materials (Kou et al., 2021).

LWA are commonly classified as particles with density lower than 2.0 $g \cdot cm^{-3}$ (Ayati et al., 2018). Conventional LWA are built by low-dense natural minerals which are becoming limiting. Silt and clay contained in dredged sediments are suitable material to substitute these minerals due to their high content in quartz, feldspar and aluminosilicate minerals, which are essential to produce LWA (Liu et al., 2017b; Wan et al., 2022; Wei et al., 2014; Zhao et al., 2022). This way, dredged sediments can be recycled while the natural mineral consumption is decreased. Sand fraction can be also used, but at lower proportion than silt and clay (Hakstege, 2007). Many authors have developed technical ways to obtain LWA from dredged sediments during last decades. It basically consists of a thermal immobilisation process where the dredged sediments and the material/s used as blinder/s are aggregated and, due to heat, expanded to reduce its total density. The heat also helps to immobilise the pollutants in the aggregates (as previously explained in Section 5.3), although it must be assured that the final product present low leachability and complies with the regulations (Bortone and Palumbo, 2007). The production costs for LWA from sediments in a new installed plant vary in the range 23 - 41 $€ \cdot m^{-3}$ (Hakstege, 2007), although this cost is very dependent on the water content of sediments. Considering that the import price of LWA is usually in the range of 80 -150 $\$ \cdot t^{-1}$ (Lim et al., 2020), it can be assumed that LWA production from sediments can be a profitable process. However, the main limitation of sediment based LWA lies on the need to satisfy the quality standards of commercial products since sediment-based LWA often present poorer properties such as water absorption, particle density, compressive strength, shrinkage, and microstructure in comparison to conventional ones. This is highly influenced by the distribution of size fractions, the type of metal that the sediments contain as well as their concentrations (Amar et al., 2021; Beddaa et al., 2020; De Gisi et al., 2020; Liu et al., 2017b).

To improve these characteristics, binders are usually added to the aggregate, although this activity can significantly increase the impacts of the process (Kou et al., 2021; Wang et al., 2019; Zhou et al., 2022). For instance, the use of ordinary Portland cement (OPC), which is one of the most common binders to produce LWA (Wang et al., 2018b), implies a generation of 842 $kgCO_2 \cdot t^{-1}$ of clinker (Amar et al., 2021). Many authors are doing noteworthy efforts to reduce the economic and environmental impacts of this process. To this aim, renewable binders obtained from wastes such as sludge, fly ash, slags, glass, and even blue-green algae have been tested (Table 4) with the goal of reducing resource consumption and the amount of materials to be landfilled, thus improving circularity. However, a feasible scheme to manage polluted sediments has not been yet established (Todaro et al., 2016; Yang et al., 2020).

Dredged sediments can be also successfully recovered by including them in cementitious material thanks to their chemical characteristics (Amar et al., 2018; Dang et al., 2013); or alternatively, in concretes (Achour et al., 2019; Rozière et al., 2015; Yang et al., 2020) or mortars (Benslafa et al., 2015; Mehdizadeh et al., 2021; Zhao et al., 2018). In this respect, De Gisi et al. (2020) reported a high sediment recovery efficient by producing 974 kg filling materials per ton of dredged sediments. The production process is mainly affected by the temperature and time of calcination, and the substitution rate of sediment. In this respect, Benzerzour et al. (2017) obtained best mortar properties at 850 ºC for 1 h, and a substitution rate of 8%. The substitution rate will vary according to the mineral characteristics of the sediments, which have significant influence on the final product's properties (Benzerzour et al., 2017). For instance, Mehdizadeh et al. (2021) obtained improved packing density and fluidity when 5% of cement mortar was replaced by fine sediments; while the 60-min static yield stress decreased when sediment substitution was in the range 5%–15%. On the other hand, Zhao et al. (2018) reported sediment content in cementitious material up to 20% without negatively influencing the mortar's compressive strength.

When sediments present considerable amounts of metals, organics and salts, the cementitious matrix is often negatively affected. In fact, organic compounds and metals such as zinc, cadmium or chromium can decrease the mortar structural strength by modifying the hydration reactions. Consequently, high-polluted sediments need to be either subjected to intensive pre-treatment to remove their pollutants or added large amounts of binders to immobilise them (Benzerzour et al., 2017; Wang et al., 2018a; Yang et al., 2020), which would decrease the feasibility, sustainability and commerciality of these sediment-based by-products (Mymrin et al., 2017). In this respect, calcination and grinding costs are generally around 45€ - 105 €·t^{-1} (Benzerzour et al., 2017).

Dredged sediments have been also used to manufacture bricks and blocks, which present higher added value than cementitious materials (Lafhaj et al., 2008; Said et al., 2015). The process is similar to other construction products, i.e., combinations of physical operations to pre-treat the building materials followed by a thermal process to synthesise the bricks (Samara et al., 2009). In this respect, Samara et al. (2009) operated a full-scale industrial plant which produced 15,000 bricks with a sediment substitution ratio of 15%. According to their study, these sediment-amended bricks presented 63% and 40% higher compressive strength and firing shrinkage, respectively; and 10% and 13% lower porosity and water absorption, respectively, than those obtained using quartz sand. Consequently, these bricks were sturdier and lesser prone to decomposition. In addition, Slimanou et al. (2021) reported sediment-based bricks to have low thermal conductivities in the range of 0.21 - 0.46 $W \cdot m^{-1} \cdot K^{-1}$, thus these bricks could act as insulator materials. However, the production costs as well as the carbon footprint of sediment-based bricks are usually high, which limits the process feasibility (Yang et al., 2020). Furthermore, as bricks are aimed at structural purposes in construction, their standard quality required is much higher than for other less valuable products such as LWA. In relation to this, the market is still reluctant to accept bricks (and other construction materials) made from polluted sediments due to the possible risks associated (Lim et al., 2020; Spadaro and Rosenthal, 2020). Higher efforts should be thus made to assure the long-term safety use of sediments in these products to improve the public acceptance.

[TABLE 4 NEAR HERE]

6.1.3. Road material

Another option to take advantage of dredged sediments is using them in the road sector (Kasmi et al., 2017; Loudini et al., 2020b; Siham et al., 2008). By means of example, Loudini et al. (2020b) used dredged sediments with 7% OPC to obtain road foundation layers at lab-scale. Since the final product is not used for structural purposes, the characteristic requirements of these sediment-based products are usually less strict, and, as the contact with human beings is much shorter than for building materials, the long-term risks are noticeably lower. This not only enables to use dredged sediments in higher percentages than in construction materials but could also ease their social acceptance. However, to the best of our knowledge, the characteristics of sediment-based road materials have not been widely tested at large scale.

It must be considered that sediments dominated by fine particles hinder their application as road materials since they need higher chemical and granulometric stabilisations, which implies the use of higher amounts of binders such as cement, lime, etc. (Miraoui et al., 2012). This, together with other sediment characteristics such as high metal content, hampers the elaboration of the product and reduces its application possibilities. Further research to study the mechanical behaviour and functionality of sediment-based road materials is needed to overcome these barriers, which is one of the goals of ECOSEDRA project.

6.1.4. Other products

There are other processes that are being currently evaluated with the aim to use dredging sediments to obtain by-products. For instance, since polluted sediments contain significant amounts of metals (Table 1), metal extraction from them appears as a sustainable alternative to traditional metal production from mines (Ferrans et al., 2021; Hasegawa et al., 2019). However, there is still a long way to make this process feasible at large scale since metal extraction is complex and dependant on multiple factors such as the level of pollution in the sediments, the metal speciation and the properties of the sediment matrix. Moreover, metal extraction processes usually present high risks of environmental pollution (Akcil et al., 2015; Norén et al., 2020).

To sum up, despite some authors have reported theoretical cost-benefits in the production of sediment-based materials (Yang et al., 2020; Zhao et al., 2022), their characteristics and properties such as the flexural and compressive strength, dry density, thermal conductivity, and water absorption tend to show lower quality in comparison to those obtained from non-renewable sources, especially when recycled materials are used as binders (Spadaro and Rosenthal, 2020; Wang et al., 2018a; Yang et al., 2020). Consequently, the products obtained from circular management configurations usually present limited applications. In addition, depending on the recycled material employed as binder, the percentage of sediments in the products is sometimes as low as 14-20% (Kou et al., 2021; Wang et al., 2019, 2018b), which can limit the quantities of sediments to be applied in the construction industry. Furthermore, the development of large-scale configurations to obtained sediment-based products that are technically, economically and environmentally feasible and able to compete in the current market with the conventional ones is still lacking (Norén et al., 2020; Spadaro and Rosenthal, 2020).

6.2. Socio-political barriers for Circular Economy application in sediments

Apart from the technical and operational issues described in Section 6.1, one of the main barriers for the wide commercialisation of sustainable and circular sediment-based products lies on their poor social acceptance since project managers, regulators, and contractors could perceive long-term risks associated to their use (Spadaro and Rosenthal, 2020; Zheng et al., 2019). Legislation is a powerful tool to improve people’s awareness by assuring the safety and quality of sediment-based products, but it has to be specific, clear and standard. Too generic and not numerically quantified limits could create controversies between regulating administrations and sediment-based producers, who could slow down the transition to circularity. In addition, differences in national regulations (Section 4) could hinder international markets of by-products due to legal discrepancies between countries. In the case of sediment-based products, it is also important to adapt the legislation to the final use. As explained in Section 6.1, the level of risk is not the same for building than for road construction purposes. On the other hand, regulations that are too conservative, even if they are case specific, can also hinder the production of circular sediment-based materials as they could limit the economic, technical and environmental feasibility of the management process. For instance, in France, dredged sediments are treated as waste by law, even though 90% of them were inert in 2017 (Beddaa et al., 2020).

Extra efforts to overcome technical and socio-political barriers are being developed by researchers and environmental stakeholders. In this respect, the European Commission has recently approved *PROMISCES* project, which aims to identify and overcome bottlenecks to deliver the ambitions of the European Green Deal (COM, 2019) and the Circular Economy Action Plan (COM, 2020). Other efforts are being doing on the alignment between the objectives of researchers, technological developers and other stakeholders such as regulators and market distributors. In order to boost circular products on the market, producers have to be aware of the market needs and consumers and contractors should be more flexible to consider not only the final characteristics of the by-products but also the overall steps in production chains. Similarly, regulators could play a key role in the transition to CE practices. First, the biggest efforts should be made in reducing the pollution in origin with the goal to maintain the quality of marine sediments. Moreover, by penalising more significantly non-circular products and practices such as landfilling, regulators can motivate the transition to the sustainable production of sediment-based products (Norén et al., 2020). Initiatives like the EU taxonomy (Lucarelli et al., 2020) are expected to increase CE applications (in the sediment sector and beyond) by encouraging companies to acquire more environmentally sustainable practices and motivating consumers to demand for greener products. However, more data regarding large-scale circular sediment management should be generated to improve social acceptance and give security to producers, policy makers and the general public.

Conclusions

Dredged sediments management is an issue of increasing concern. As they are characterised by containing high proportions of silt and clay fractions, they tend to adsorb and accumulate toxic pollutants such as heavy metals and organic contaminants, which makes them be considered as toxic wastes. Physical operations and/or chemical, thermal and biological processes are needed to reduce their pollutant content and the risk of pollutant resuspensions to the environment. The type and degree of treatment is mainly determined by the sediments' characteristics, including not only their pollutant concentrations but also their water (free and interstitial) content and their size distribution. The treatment scheme is also influenced by legal requirements, although there is a lack of uniformity and coherence in national regulations in terms of the sediment characteristics to be analysed, the pollutant concentration limits and the categories for the classification of sediments.

Conventional dredged sediment treatment techniques, which goal is mainly the reduction of the toxicity of sediments, are usually economically and environmentally demanding. However, with the increasing people's awareness in environmental issues and the more restrictive regulation that hinder certain classical applications of sediments, more sustainable and circular approaches for the management of dredged sediments are being sought.

Some authors have evaluated the possibility to take advantage of the mineral composition of sediments to obtain by-products that can be used for building and road construction purposes with the goal to decrease the need of raw materials in these sectors and reduce the amounts of sediments wasted. In fact, the production of sediment-based light-weight aggregates, concrete, mortar, bricks, and road foundation layers are being increasingly assessed (mainly at lab scale). However, the production of these sediment-based materials is not always feasible and, in most cases, they present lower quality in terms of structural properties in comparison to conventional non-circular products. The transition to circular economy in sediment management not only implies to obtain by-products from sediments, but also to decrease the overall impacts of the sediment management scheme, for instance by using recycled materials as binders instead of raw resources, minimising transport costs, and reducing overall waste production to approach zero-pollution ambition. However, the commercialisation of sustainable and circular sediment-based products has to deal with socio-political barriers such as low social acceptance, insufficient support from administrations and stakeholders, lack of standard regulations and the limited alignment between the objectives of circular sediment-based producers, consumers and regulators. Significant efforts are being made to overcome these barriers through initiatives such as the EU taxonomy, and the development of innovative projects like *PROMISCES* and *ECOSEDRA*.

However, useful results that could be extrapolated to industrial applications of circular economy in the sediment sector are still scarce and deserved to be further investigated.

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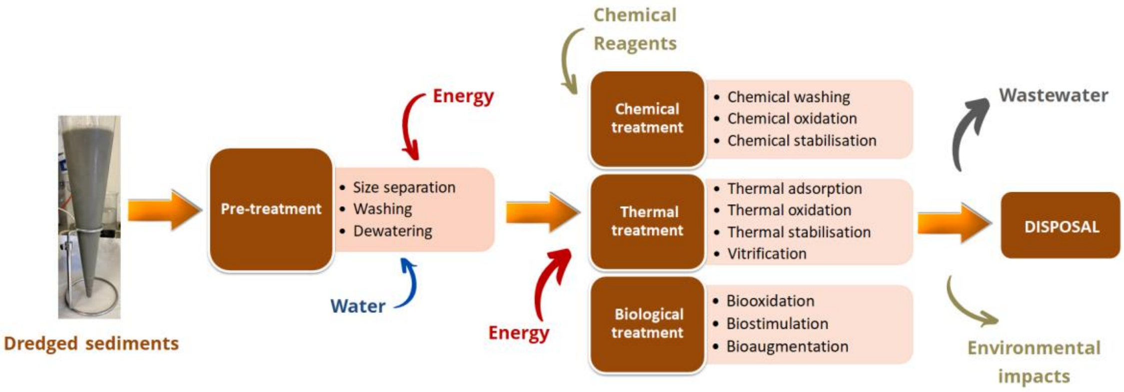

Figure 1. General configuration of linear sediment treatment.

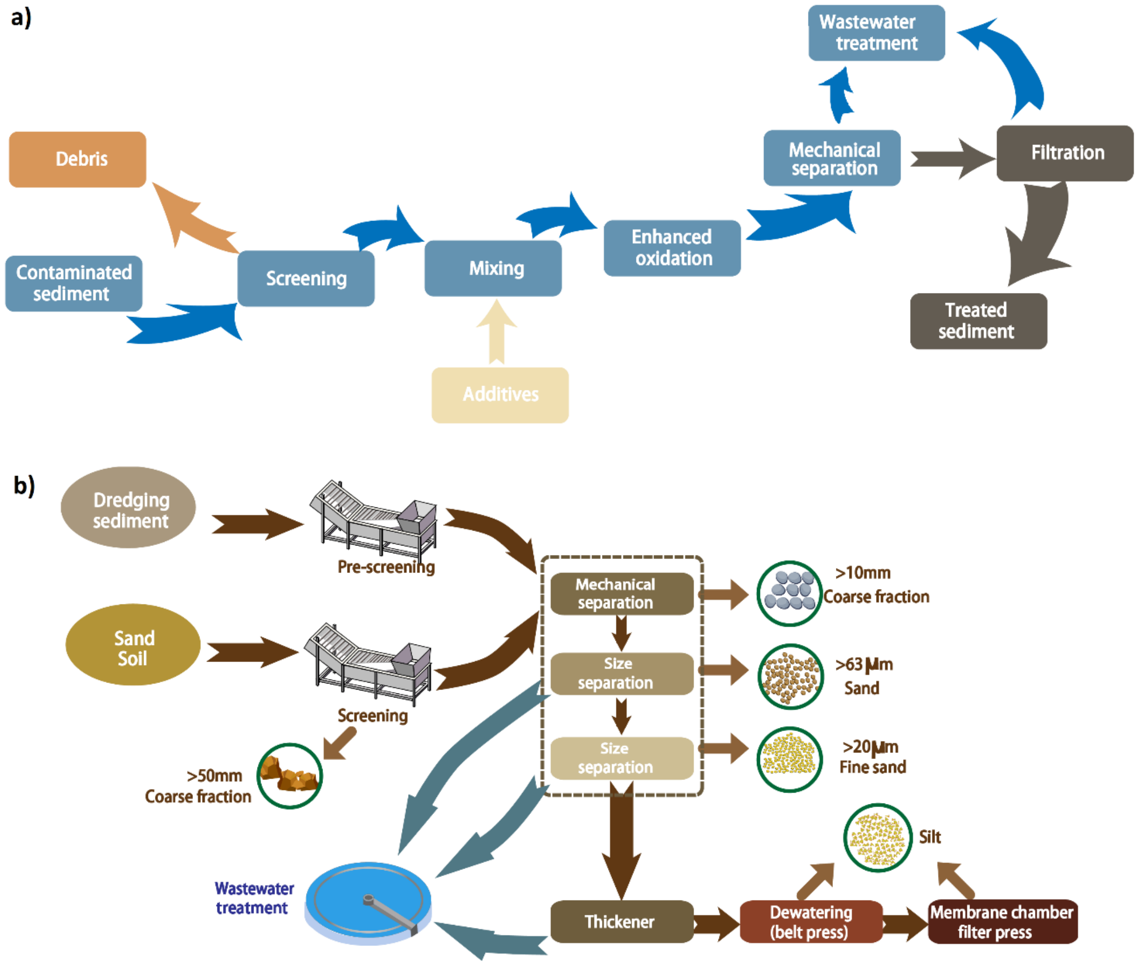

Figure 2. Simplified scheme of: a) the BioGenesis^{SM} sediment treatment scheme; b) the METHA sediment treatment scheme.

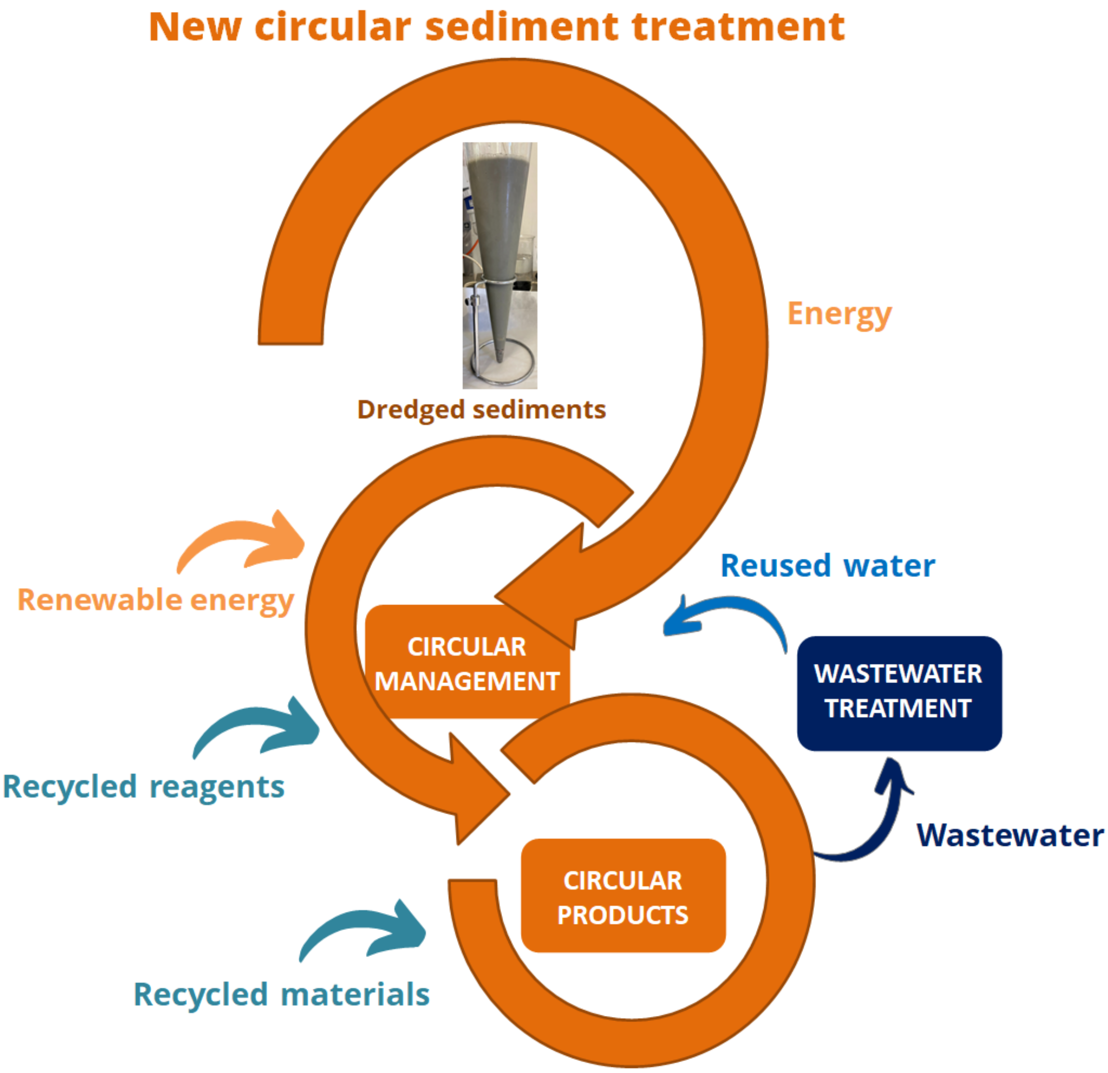

Figure 3. General configuration of circular sediment management strategy.

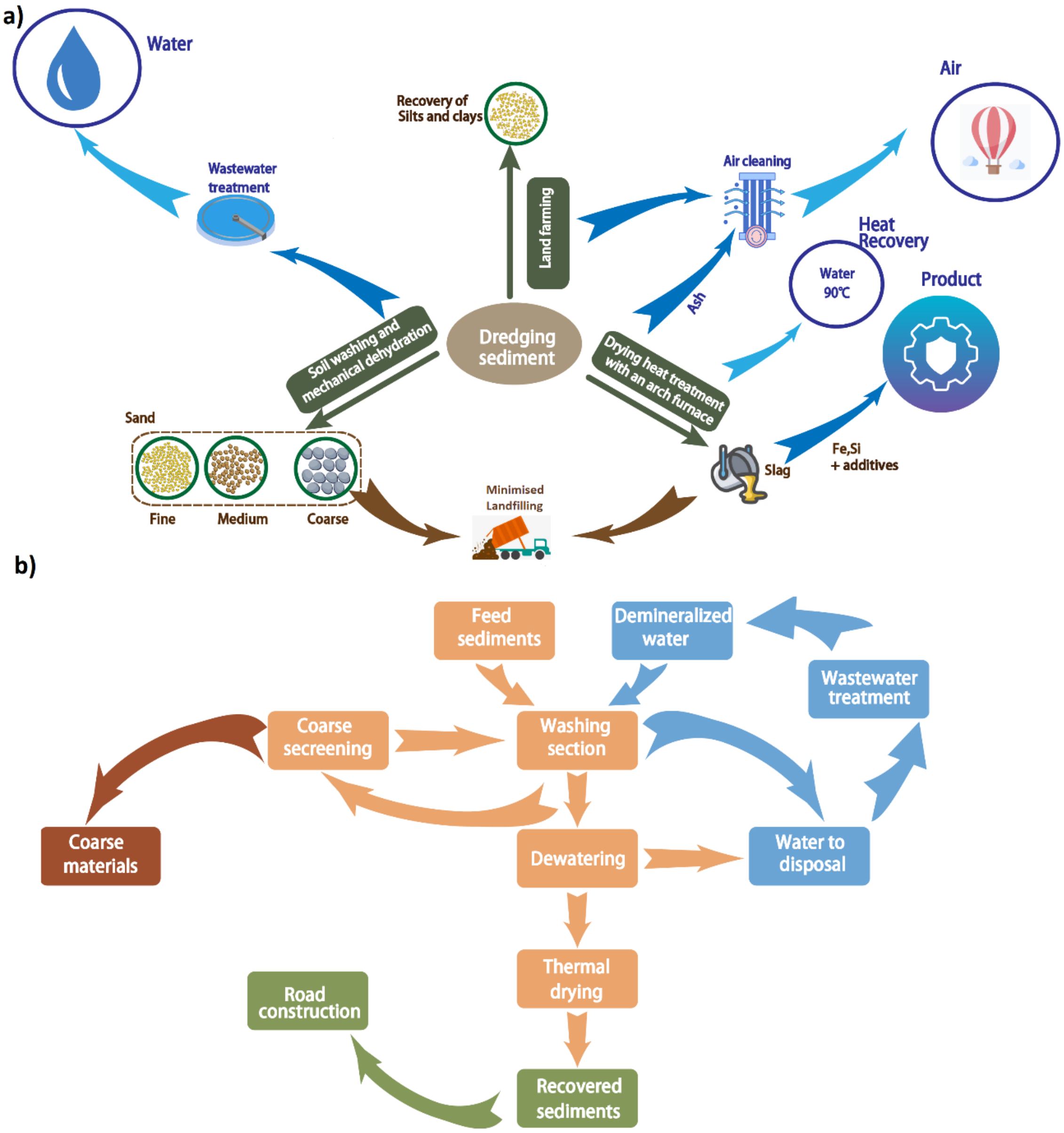

Figure 4. Simplified sediment management scheme of: a) the SEDI.PORT.SIL project, b) ECOSEDRA project.

Table 1: Main characteristics of dredged sediments from different locations.

Reference		Fonti et al. (2013)	Ben Hadj Ali et al. (2014)	Ben Hadj Ali et al. (2014)	Doni et al. (2018)	Ferrans et al. (2021)	Lopez et al. (2022)	Loudini et al. (2020a)	Loudini et al. (2020b)	Norén et al. (2020)	Norén et al. (2020)	Todaro et al. (2018)	Wang et al. (2018 a)	Wang et al. (2018 b)	Zhao et al. (2022)	Zhou et al. (2022)
Location		Port of Ancona (Italy)	Rades harbour (Tunisia)	Gabes harbour (Tunisia)	Livorno harbour (Italy)	Malmfjärden bay (Sweden)	Galveston bay (USA)	Safi harbour S1 (Morocco)	Safi harbour S2 (Morocco)	Gothenburg port (Sweden)	Oskarshamn port (Sweden)	Gulf of Taranto (Italy)	Hong Kong coast	Kwun Tong Typhoon Shelter (Hong Kong)	Tai Lake (China)	Lamma Island (Hong Kong)
Granulometry (%)	Gravel	-	<1	<1	N.D.	-	-	1	0	-	-	-	10	-	-	48.7
	Sand	-	≈18	≈44	63	10-20	-	41	99.5	-	-	19.4	30	6.1	-	51.1
	Silt- Clay	80	≈81	≈55	37	80-90	-	58	0.5	-	-	80.6	60	93.9	-	0.2
Physico-chemical characteristics	Humidity (%)	40	65-80	-	-	70-78	-	-	-	-	-	44.8	42	58.9	89.9	23.9
	pH	-	-	-	8.3	6.7	-	-	-	-	-	8.8	8.0	7.2	-	8.1
	Conductivity ($\mu S \cdot cm^{-1}$)	-	-	-	16,800	-	-	-	-	-	-	4,100	-	-	-	-
Nutrients ($mg \cdot Kg^{-1}$)	TN	-	-	-	3,840	8563-9488	-	-	-	-	-	-	-	-	-	-
	NH_4	-	-	-	14.1	-	-	-	-	-	-	-	-	-	-	-
	NO_3	-	-	-	20.0	-	-	-	-	-	-	-	-	-	-	-
	TP	-	-	-	650	740-1159	-	-	-	-	-	-	-	-	-	-
Organic compounds ($mg \cdot Kg^{-1}$)	Total organics	28,000	67,600-68,300	70,300-109,700	19,300	129,000	-	50,000	0	-	-	123,000	-	64,000	5741	25,500
	TBT	-	-	-	-	-	-	-	-	150	N.D.	-	-	-	-	-
	TPH	-	-	-	5447	-	-	-	-	-	-	-	-	-	-	-
	PAHs ($\mu g \cdot Kg$)	-	-	-	-	-	-	-	-	-	-	5,389	-	-	-	-
	PCBs ($\mu g \cdot Kg$)	-	-	-	-	-	-	-	-	-	-	1,669	-	-	-	-
	Cu	33	-	-	123	40	0.8-351.8	-	-	50	1100	10.5	-	1700	70.85	378.5

Metals (mg·Kg^{-1})	Zn	83	-	-	240	120	4.3-336.6	-	-	200	400	16.6	-	410	104.3	138.5
	Cd	0.5	-	-	N.D.	N.D.	0.02-0.5	-	-	0.4	4	-	-	2.6	-	1.8
	Cr	70	-	-	64.0	24	1.5-70.8	-	-	40	50	54.0	-	240	84.49	29.7
	Ni	40	-	-	49.1	23	1.1-30.3	-	-	20	60	38.2	-	68	-	23.6
	Pb	12	-	-	59.7	44.9	1.8-29.2	-	-	40	560	87.4	-	120	65.06	101.1
	As	10	-	-	-	8.3	1.3-20.0	-	-	-	-	12.3	-	-	-	7.8
	Co	-	-	-	-	-	-	-	-	-	-	7.1	-	-	-	-
	Al		-	-	-	-	1,6-54,003	-	-	-	-	-	-	-	-	-
	Fe	22,000	-	-	-	-	636-23,655	-	-	-	-	-	-	-	-	-
	Sb		-	-	-	-	0.1-1.5	-	-	-	-	-	-	-	-	-
	Mn		-	-	-	-	6.1-694.5	-	-	-	-	-	-	-	-	-
	Hg		-	-	-	-	0.01-0.4	-	-	-	-	-	-	-	-	-
	V		-	-	-	-	-	-	-	-	-	53.3	-	-	-	-

N.D.: non detected; NH_4: ammonium; NO_3: nitrate; PAHs: polycyclic aromatic hydrocarbons; PCBs: poly chlorinated biphenyls; TBT: tributylin; TN: total nitrogen; TP: total - phosphorus; TPH: total petroleum hydrocarbon.

S1 of Safi harbour corresponds to the basin zone of the port and S2 corresponds to the channel zone of the port.

Table 2: Characteristics of the regulations relative to dredged sediment management in Italy, France and Spain.

	ITALY	**FRANCE**	**SPAIN**
Reference regulation related to handling and management of marine dredged sediments.	Ministerial Decree n° 173 of 15th July 2016. DM 5th February 1998.	Environmental Code and the circular of 4/07/2008.	Guidelines for the characterisation of dredged material and their relocation within waters of the maritime-terrestrial public domain (2014). Law 22/2011 on Contaminated Wastes and Soils.
Parameters to be evaluated	• Macroscopic description; • Physical parameters; • Chemical parameters; • Ecotoxicological analysis: battery of 3 tests on all samples.	• Physical parameters; • Chemical parameters; • Biological characterisation: only in the cases where any of the compounds exceeds the N2 threshold.	• Physical parameters; • Chemical parameters (only for those potentially toxic according to previous testing); • Biological parameters (only for non-toxic sediments, quality C).
Other requirements	• Microbiological investigations for sea diving and coastal nourishment in proximity to bathing and aquaculture areas. • Mineralogical investigations in case of coastal nourishment. • Study of benthic communities for immersion in the sea over 3 nmi and nourishment.	• Determination of nitrogen and phosphorus for sediment discharged in sensitive areas; • Faecal contamination to avoid impacts on shellfish, mariculture, or bathing areas. • Biological characterisation of the affected site in case of sediment immersion, if at least one element	• Previous toxicity test (PTT), consisting of the inhibition of the luminescence of the bacterium *Vibrio fischeri* by direct contact of sediment suspensions and measured as EC50 concentration. • Bio-essays.

	- Release tests in water of the matrices for a period of 24 hours. The leachates from these tests cannot overpass the pollutants thresholds reported in DM 5th February 1998.	exceeds the N2 threshold. - The particle size has to be considered in case of reuse for nourishment.	
Management options	Based on the sediment quality class determined by a weighted integration criteria regarding the ecotoxicological hazard class and chemical hazard class.	Based on concentration limits of some physical and chemical parameters such as particle size, metals, organic micro-pollutants.	Based on two-step characterisation: one preliminary step to analyse their potential toxicity and a second step based on chemical and biological characterisation.
Classification	5 sediment quality classes: A > B > C > D > E.	Categories: N1 (low pollutant concentrations); N2 (mid pollutant concentrations).	Classified as non-toxic or toxic (when overpasses some legal limits). 3 quality classes for non-toxic sediments: A > B > C.
References	Bortone and Palumbo (2007).	Mugnai et al. (2018); Tessier et al. (2011)	Buceta et al. (2015)

nmi: nautic miles.

Table 3: Main characteristics of conventional dredged sediments treatment processes.

	Treatment	**Goal of the process**	**Target wastes**	**Performance**	**Main impacts**	**Costs**	***References***
Physical operations	Mechanical separation	To divide sediments in sand, silk and clay fractions	Sediments	Generally high performance	Energy consumption (M). Water consumption (L).	3 - 11 € (2007)·m^{-3}	Bortone and Palumbo (2007) Kim et al. (2016) Mulligan et al. (2001) Pal and Hogland (2022)
	Dewatering	To remove intersticial water	Water	Up to 80% of water removal (improved with chemical conditioners)	Energy consumption (M) Use of chemicals (L)	10-35 € (2007)·m^{-3}	Bortone and Palumbo (2007) Chi et al. (2018) Zhao et al. (2022)
	Washing	To reduce pollutant and salt concentration in sediments	Metals, salts.	Depending on the reagent, dosage, contact time, temperature and number of extractions (each case must be evaluated). Worse performance for smaller fractions.	Use of chemicals (M) Water pollution (M). Water consumption (H) Energy consumption (L).	40-73 € (2009)·m^{-3}	Beiyuan et al. (2018) Doni et al. (2018) Ferrans et al. (2021) Jeon et al. (2015) Pal and Hogland (2022) Papadopoulous et al. (1997) Polettini et al. (2009) Tsai et al. (2009)

Chemical treatment	Chemical oxidation	To degrade pollutants into non-hazardous compounds.	Organic pollutants	Depending on the chemical reagent, type of pollutant and water content. Poor metal removal.	Use of chemicals (H). Air treatment (L). Water consumption (L). Ecotoxicity (M) Environmental impacts (M-H)	Depending on the process conditions.	Bortone and Palumbo (2007) Debnath et al. (2021) Ferrarese et al. (2008) Maletic et al. (2019) Todaro et al. (2016)
	Solidification/Stabilisation (S/S)	To immobilise pollutants to reduce their mobility and bioavailability.	Metals.	Good performance with As, Pb, Cr(VI), Hg, Cd, Cu, Zn. Poor performance with high water, organics, salts contents and with certain metals. Lower impacts than thermal treatment	Use of chemicals (H). Energy Consumption (M) Risk of pollutant release (M) Ecotoxicity (M) Environmental impacts (M-H)	10 – 41 € (2007)·m^{-3}	Amar et al. (2021) Barjoveanu et al. (2018) Hossain et al. (2020) Kou et al. (2021) Maletic et al. (2019) Mulligan et al. (2001) Norén et al. (2020) Radenovic et al. (2020) Tang et al. (2020) Todaro et al. (2020) Wang et al. (2015)
Thermal treatment	Thermal desorption	To separate VOCs from sediments.	Organic pollutants	High for VOCs (T = 450-500ºC); for some metals (T = 800 ºC).	Energy consumption (M-H). Air pollution (M). Ecotoxicity (M)	50 – 70 € (2007)·t^{-1}	Bortone and Palumbo (2007) Mulligan et al. (2001) Pal and Hogland (2022) Todaro et al. (2016)

				Environmental impacts (M-H)		
Thermal oxidation	To degrade organic pollutants.	Organic pollutants.	High performance if temperature is adequate.	Energy consumption (H). Air pollution (M). Environmental impacts (M-H)	50 – 70 € (2007)·t^{-1}	Amar et al. (2021) Bortone and Palumbo (2007) Pal and Hogland (2022) Snellings et al. (2016)
Thermal immobilisation	To immobilise metals to reduce their mobility and bioavailability.	Metals.	Depending on the sediment characteristics. Generally higher performance than chemical S/S. Bricks, aggregates and other materials can be produced.	Energy consumption (H). Air pollution (M). Ecotoxicity (M) Environmental impacts (H)	50 – 70 € (2007)·t^{-1}	Snellings et al. (2016) Zhao et al. (2022)
Vitrification	To destroy and/or immobilise pollutants.	Metals.	Very effective.	Energy consumption (HH). Air pollution (M). Ecotoxicity (M) Environmental impacts (H)	Depending on the process conditions.	Colombo et al. (2012) Mulligan et al. (2001) Norén et al. (2020) Todaro et al. (2016)

Biological processes	To degrade pollutants by microbial activity.	Biodegradable pollutants.	Low degradation rates.	Land requirements (H).	Depending on the process conditions.	Amar et al. (2021) Liu et al. (2017a) Maletic et al. (2019) Doni et al. (2018) Feng et al. (2022) Fodelianakis et al. (2015) Wu et al. (2014)

L: Low impact; M: Medium Impact; H: High Impact.

€/$ (xx): value of money in €/$ referred to year xx; CDF: confined disposal facilities; VOCs: volatile organic compounds.

Table 4: Circular production of construction materials obtained from dredged sediments.

Product	**Sediment content (%)**	**Binders**	**Scale**	**Treatment operation and processes**	**Product characteristics**	**Positive impacts**	**Costs**	***References***
Sediment-based zeolite	-	Sodium metaaluminate, cadmium chloride, sodium silicate, sodium chloride, hydrochloric acid, sodium hydroxide.	Lab	Mechanical separation, Hydrothermal treatment	High specific surface: 516.36 m^2/g.	Capacity of Cd removal from wastewater.	-	Chen et al. (2021)
Filling material	85-90	Quicklime (10% wt), activated carbon, organocaly	Pilot	S/S	Unconfined compressive strength: 18.2-28.1 KPa. Suitable for environmental enhancement.	No pre-treatment of sediment needed (lower costs and impacts). High sediment recovery: 97.4%.	-	De Gisi et al. (2020)
LWA	66	Steel slag (27%) and waste glass (7%).	Lab	Thermal stabilisation	Reduced water absorption:2.2%. Increased compressive strength: 23.1 MPa.	Alternative for the disposal of dredged sediments. Waste reduction.	-	Lim et al. (2020)
LWA	50	Sewage sludge	Lab	Thermal stabilisation	Density: 836 kg m^{-3} Compressive strength: 13.7 MPa. Water absorption < 12%.	Reduction of toxicity of sediments. Waste reduction.	-	Liu et al. (2018)

Mortar	20	OPC (60%); CCS (10 %), GGBS (10%).	Lab	Stabilisation	Lower flexural and compressive strength when only sediments were used as binder. Low pozzolanic activity.	Reduction in the use of cement and consumption of mining materials.	-	Kou et al. (2021)
Bricks	15-20	Clay	Lab	Thermal stabilisation	Increase of mechanical properties: 20.6-32.5% Low thermal conductivity: 0.21-0.46 $W \cdot m^{-1} \cdot K^{-1}$ (insulating material)	Resource recovery. Possible economic and environmental benefits.	-	Slimanou et al. (2020)
Blocks	15	CCR (5%), OPC (20%), ISSA (5%), fly ash (5%), GGBS (20%).	Lab	S/S	Addition of CCR, GGBS and fly ash provided relative high strength.	Carbon sequestration: 4.3 %wt Reduction of sediment toxicity. Waste reduction.	-	Wang et al. (2018b)
Fill material/ Blocks	14-47.5	OPC, biochar (1-2%), ISSA	Lab	Stabilisation	Improvement of immobilisation of pollutants due to biochar.	Alternative for disposal in landfills for toxic wastes.	-	Wang et al. (2019)
LWA	80-90	Slag waste containing dioxines (10-20%)	Lab	Thermal stabilisation	No presence of dioxins nor metal leaching	Dioxins removal. Waste reduction.	-	Wei et al. (2014)
Foamed concrete	30	OPC, foam (0-80% v/v) and silica fume (10%).	Lab	Stabilisation	Compressive strength: 2.1-18.8 MPa. Dry density: 620.6-1442 $kg \cdot m^{-3}$.	Without heating nor pressurising. Reduction of wastes.	40.98-50.88 $\$ \cdot m^{-3}$	Yang et al. (2020)

					Thermal conductivity: 0.173-0.516 W$\cdot$m$^{-1}\cdot$K^{-1}. Water resistance coefficient: 0.43-0.73.			
Fill material	82.5-88.5	OPC (1.5-7.5%) and fly ash (10%).	Lab	Stabilisation	Large quantities of calcium silicate hydrate.	Improved properties in comparison with stabilising only with cement or fly ash	-	Yoobanpot et al. (2020)
Road construction material	94-98	CSA cement	Lab	S/S	Compressive strength: 0.5-1.91 MPa. Splitting tensile strength: 0.057-0.119 MPa. Elastic modulus: 0.41-1.07 GPa.	CSA is a greener binder than OPC.	-	Zentar et al. (2021)
Ceramsite (LWA)	60	Blue-green algae (20%); 20% of additives (fly ash:calcium oxide:kaolin)	Lab	Dewatering; thermal stabilisation	Product accomplished the National Standard as building material.	Simultaneous reduction of lake pollutants: toxic dredged sediments and algae. Economical profit: 2.3\$$\cdot$m^{-3}.	18.8 \$$\cdot$m^{-3}	Zhao et al. (2022)
Fill material	70	ISSA (20%); OPC (10%)	Lab	S/S	Low leachate concentrations even under simulative acidic rainfall conditions.	Decrease of cement consumption.	-	Zhou et al. (2022)

CCR: calcium carbide residue; CCS: calcium carbide slag; GGBS: ground granulated blast-furnace slag; ISSA: incinerated sewage sludge ash; LWA: lightweight aggregates; OPC: ordinary Portland cement; S/S: Stabilisation-solidification.