Phytomanagement of Metal(loid)-Contaminated Soils: Options, Efficiency and Value

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The growing loss of soil functionality due to contamination by metal(loid)s, alone or in combination with organic pollutants, is a global environmental issue that entails major risks to ecosystems and human health. Consequently, the management and restructuring of large metal(loid)-polluted areas through sustainable nature-based solutions is currently a priority in research programs and legislation worldwide. Over the last few years, phytomanagement has emerged as a promising phytotechnology, focused on the use of plants and associated microorganisms, together with *ad hoc* site management practices, for an economically viable and ecologically sustainable recovery of contaminated sites. It promotes simultaneously the recovery of soil ecological functions and the decrease of pollutant linkages, while providing economic revenues, *e.g.* by producing non-food crops for biomass-processing technologies (biofuel and bioenergy sector, ecomaterials, biosourced-chemistry, etc.), thus contributing to the international demand for sustainable and renewable sources of energy and raw materials for the bioeconomy. Potential environmental benefits also include the provision of valuable ecosystem services such as water drainage management, soil erosion deterrence, C sequestration, regulation of nutrient cycles, xenobiotic biodegradation, and metal(loid) stabilization. Phytomanagement relies on the proper selection of (i) plants and (ii) microbial inoculants with the capacity to behave as powerful plant allies, *e.g.*, PGPB: plant growth-promoting bacteria and AMF: arbuscular mycorrhizal fungi. This review gives an up-to-date overview of the main annual, perennial, and woody crops, as well as the most adequate cropping systems, presently used to phytomanage metal(loid)-contaminated soils, and the relevant products and ecosystems services provided by the various phytomanagement options. Suitable bioaugmentation practices with PGPB and AMF are also discussed. Furthermore, we identify the potential interest of phytomanagement for stakeholders and end-users and highlight future opportunities boosted by an effective engagement between environmental protection and economic development. We conclude by presenting the legal and regulatory framework of soil remediation and by discussing prospects for phytotechnologies applications in the future.

Keywords: phytoremediation, cash crops, cropping systems, PGPB, bioeconomy, bioinoculants, AMF

INTRODUCTION

Soil contamination with metal(loid)s, termed also 'trace elements' in Biogeochemistry and Life Sciences (hereafter referred as TE) is a global environmental issue that poses serious risks for ecosystem integrity and human health (Joimel et al., 2016; Hou et al., 2017; Pérez and Eugenio, 2018; Bagherifam et al., 2019; Mench et al., 2020; Haller and Jonsson, 2020). Although background and bioavailable TE levels are generally low in soils, except at geochemical anomalies with either deficiency or exceedance, anthropogenic activities such as industry, mining, smelting and metallurgy, intensive agriculture, e-wastes, traffic, use of fossil fuels, etc. have caused an increase in soil TE concentrations reaching hazardous levels (Alloway, 2013; Kumpiene et al., 2017; Petruzzelli et al., 2020). Furthermore, anthropogenically-released TE tend to have higher availability and mobility than those resulting from natural processes (Kabata-Pendias, 2010), raising environmental and human health concerns. In the European Union (EU), there are at least 2.8 million potential contaminated sites and 650,000 registered sites where polluting activities occurred or are occurring (Pérez and Eugenio, 2018). Such high number of contaminated sites, coupled with the threat posed by contaminants to environmental and public health, pushed policymakers and legislators to 1) set soil protection as a strategic priority and 2) encourage the remediation, reclamation, restoration and recovery of those sites (Reinikainen et al., 2016; Castelo-Grande et al., 2018; Ramon and Lull, 2019). Consequently, nowadays, numerous countries have specific legislation and guidelines to deal with contaminated sites and are committed to their remediation, based on either regulatory values or site-specific risk assessment, which further depends on contemplated future land use (Mench et al., 2020). This has boosted the number of initiatives in the EU aimed at recovering contaminated sites (Pérez and Eugenio, 2018). However, despite this positive trend, these numbers fall short given the extent of the problem.

The recovery of TE-polluted land is critical to enhance soil ecosystem services, as well as to decrease the contamination of the soil matrix itself, together with that of recipient waterbodies and food crops, thus ensuring human welfare. Additionally, the reclamation of contaminated sites through phytomanagement creates a set of opportunities to comply with the net-zero carbon emissions targets, by generating areas that act as carbon sinks and by implementing programs for valuing land that commensurate with the current sustainability paradigm (Bardos et al., 2016, 2020; Cundy et al., 2021). Furthermore, the use of contaminated lands for bioenergy production can importantly reduce the clearing of agricultural/fertile areas for this purpose, leading to greenhouse gas emission savings (Mellor et al., 2021). In the last years, stakeholders, including site owners, local population, investors and public authorities, are increasing their environmental awareness and gradually recognizing that recovered sites can provide values, goods and services, and then demanding the management and recovery of contaminated land. In the last 2 decades, environmentally-friendlier, greener technologies have paved the way to become reliable alternatives to previously favored disruptive

methodologies (Kidd et al., 2015). Amongst these, phytotechnologies stand out as a cost-effective, sustainable option for the recovery of TE-contaminated areas (Mench et al., 2010, 2018; Herzig et al., 2014; Thijs et al., 2018; Kolbas et al., 2020). Phytotechnologies involve a set of techniques that exploit plants and sustainably manage "soil-plant-microbial" systems to recover polluted sites, especially those with low and medium levels of soil pollution (Vangronsveld et al., 2009; Mench et al., 2010, 2018). Regarding TE-pollution, phytotechnologies can also be an effective choice for highly contaminated sites when the goal is stripping the soil of its bioavailable metal(loid) fractions, or TE-stabilization (thus, decreasing potential TE toxicity). This can be tackled through TE uptake and accumulation in harvestable plant parts (*e.g.,* phytoextraction), or by progressively promoting *in situ* inactivation by combining the use of TE-excluding plants, soil amendments and/or microbial inoculants (assisted phytostabilization), respectively (Mench et al., 2010, 2018; Epelde et al., 2014; Kidd et al., 2015; Burges et al., 2016, 2017, 2018). Phytoextraction options can also be assisted by soil amendments, chemical agents and soil microorganisms (assisted phytoextraction) (Vangronsveld et al., 2009; Mench et al., 2010; Kidd et al., 2015; Wang et al., 2019; Kolbas et al., 2020). Within this context, microbial bioinoculants and site-tailored cropping systems are most useful tools to help plants cope with soil contamination. Not surprisingly, the research on microbial bioinoculants (*e.g.,* plant growth-promoting bacteria - PGPB and arbuscular mycorrhizal fungi - AMF), soil amendments (organic and/or inorganic), and suitable cropping patterns (*e.g.,* intercropping, winter cropping) escalated in the last decade, yielding very positive and encouraging outcomes.

In the recent past, phytotechnologies were combined with sustainable site management practices, giving birth to a wider approach—*phytomanagement*—where environmental benefits are allied with financial returns for stakeholders, and/or wider social and economic benefits to the surrounding community (Robinson et al., 2009; Cundy et al., 2016; Burges et al., 2018; Li et al., 2018; Bardos et al., 2020). Beside presenting a plethora of ecological benefits such as the progressive reestablishment of soil health and decreased TE run-off risks, phytomanagement places emphasis on obtaining economic profits by using cash crops to produce biomass for renewable energy and valuable materials (Evangelou et al., 2015; Kidd et al., 2015; Cundy et al., 2016; Šimek et al., 2017; Thijs et al., 2018; Xue et al., 2018). Phytomanagement is in line with the objectives of the European Green Deal roadmap that aims at turning climate and environmental challenges into opportunities, restoring biodiversity and decreasing pollution. Using contaminated sites, for instance, for bioenergy production supports the growing demand for energy sources alternative to fossil fuels, while reducing the prevailing pressure on the use of forests or agricultural productive land for the productions of biofuels (Edrisi and Abhilash, 2016; Sytar and Prasad, 2016).

Under this scope, this review aims to: 1) provide an overview of the main traits and potential economic applications of the most widely used TE-tolerant cash crops that better fit phytomanagement goals; 2) summarize selected field phytomanagement experiments (especially those published in

2010–2020) using these plants; 3) indicate suitable crops that can be coupled with cash crops to improve site management and phytomanagement effectiveness; 4) address the importance of cropping patterns and the use of amendments in the remediation of TE-contaminated sites; and 5) explore the use of microorganisms, namely PGPB and AMF, as probiotics for soil health recovery and plant performance (establishment, survival, growth, and physiological traits). Cash crops were selected based on their ability to provide sufficient harvestable biomass that can be processed, especially for the local bioeconomy, and of which there is a significant body of literature based on empirical research, especially from field trials. We conclude by highlighting the current status of phytomanagement and the legal and regulatory frameworks for its implementation across Europe.

The type of amendments, cropping systems and their combination provides a myriad of possible scenarios. This topic is a matter of thorough scrutiny elsewhere (*e.g.,* Kidd et al., 2015), and therefore it will not be exhaustively addressed here. Genetic-engineered cultivars are also beyond the scope of this review (for information on this topic, see for instance Gunarathne et al., 2019 and Sebastian et al., 2019).

PHYTOMANAGEMENT BENEFITS AND CONSTRAINTS—BRIEF OVERVIEW

Soil contamination reduces site's economic, ecological and social values. As addressed, a suitable and cost-effective option to remediate such contaminated sites and restore land values is the phytomanagement, a multi-objective management strategy that reconciles economic and social returns with ecological gains (**Figure 1**) (Nsanganwimana et al., 2014; Kidd et al., 2015; Burges et al., 2018). Phytomanagement can provide financial benefits through planting valuable crops that serve as feedstock for multiple industries and end-products such as furniture, pulp and paper, biochemicals (adhesives and detergents), insulation and building materials, composites and plastic alternatives, food additives, animal feeding and bedding, etc. Some of these crops can also be used as bioenergy crops yielding high-quality biomass (Nsanganwimana et al., 2014, 2015; Burges et al., 2018; Lacalle et al., 2018; Mench et al., 2018; Thijs et al., 2018) to produce renewable energy (electricity, heat and biofuels) (Gonsalvesh et al., 2016; Pandey et al., 2016; Rizwan et al., 2018; Grottola et al., 2019; Pogrzeba et al., 2019; Rusinowski et al., 2019; Sidhu et al., 2020; Tran et al., 2020). Most importantly, these crops offer the possibility to combine the production of biomass for energy production and/or other end-products (Grisan et al., 2016; Barla and Kumar, 2019) with TE phytoextraction or phytostabilization (Thijs et al., 2018; Chalot et al., 2020). Likewise, they can simultaneously promote the biodegradation of soil organic contaminants. Economic revenues can also be obtained through phytomining, a phytotechnology focused on the recovery of valuable TE (*e.g.,* Co, Ni, and Re) from the TE-rich biomass of hyperaccumulators (also called bio-ores) (Remigio et al., 2020; Chaney et al., 2018). TE-rich phytomass can be pyrolysed/calcinated and the resulting biochar and/or

ashes used as ecocatalysts in biosourced fine chemistry (Clavé et al., 2016; Quintela-Sabarís et al., 2017; Mench et al., 2018; Xue et al., 2018; Bihanic et al., 2020; Kolbas et al., 2020). The production of high-value products fits the Circular Economy (CE) and Bioeconomy paradigms, both highly promoted within the EU and other strong economies such as China, United Kingdom, Canada and Japan (Korhonen et al., 2018). The CE paradigm promotes economic development through a cyclical flow of materials that spill over as direct and indirect benefits to the environmental and social dimension of our society (Korhonen et al., 2018). However, CE requires a shift in the society´s mindset and requires, among other aspects, the design of new business models and robust networking and innovation in production processes and commercial products (Prieto-Sandoval et al., 2018).

Regarding the potential environmental benefits, at the local level, phytomanagement improves soil health and fertility (Herzig et al., 2014; Kidd et al., 2015; Touceda-González et al., 2017; Xue et al., 2018), soil organic matter (OM) quantity and quality (Mench et al., 2018; Risueño et al., 2020; Álvarez-Rogel et al., 2021) and soil biodiversity, both faunal (Chauvat et al., 2014) and microbial (Foulon et al., 2016; Šimek et al., 2017; Durand et al., 2018; Xue et al., 2018; Garbisu et al., 2020; Kidd et al., 2021). At the large scale, phytomanagement can enhance carbon sequestration, mitigate the emission of greenhouse gases, and reduce and/or prevent TE dispersion (Evangelou et al., 2012a; Kidd et al., 2015; Cundy et al., 2016; Šimek et al., 2017; Xue et al., 2018).

As a risk-based approach, prior to its implementation, phytomanagement requires an initial risk assessment to evaluate pollutant linkages (source-pathway-receptor) (**Figure 1**) (Cundy et al., 2016; Reinikainen et al., 2016). Upon this risk evaluation, an option appraisal must be conducted by weighting several variables such as feasibility, time, economic viability, legal requirements, social approval, etc. to properly outline the best way of handling the abovementioned pollutant linkages. Option appraisal is a baseline cornerstone for the successful design of any restoration plan (Mench et al., 2020). In addition, for phytomanagement to be fully operational at a given site, an optimization stage before full-scale implementation is required. At this stage, issues related to edaphoclimatic conditions are addressed to guide the selection of the most appropriate crops. Potential edaphic constrains typically include physical and chemical characteristics of the soil. Physical constrains typically regard to compaction, reduced water holding capacity and low aeration (Shrestha and Lal, 2011), whereas chemical properties are usually related to low (or high) pH, high TE concentrations, mixtures of TE with other pollutants (*e.g.,* mineral oils, polycyclic aromatic hydrocarbons, pesticides, polychlorinated biphenyls, organochlorines, per- and polyfluoroalkyl substances), low nutrient and OM contents, etc. (Kidd et al., 2015). These soil characteristics depend on ongoing and/or past polluting activities, which in the case of industrial contaminated sites depend on the type of industry and its products (Alloway, 2013). The effects of high TE concentrations depend on their intricate reactions with soil phases, namely through, *e.g.,* dissolution, sorption,
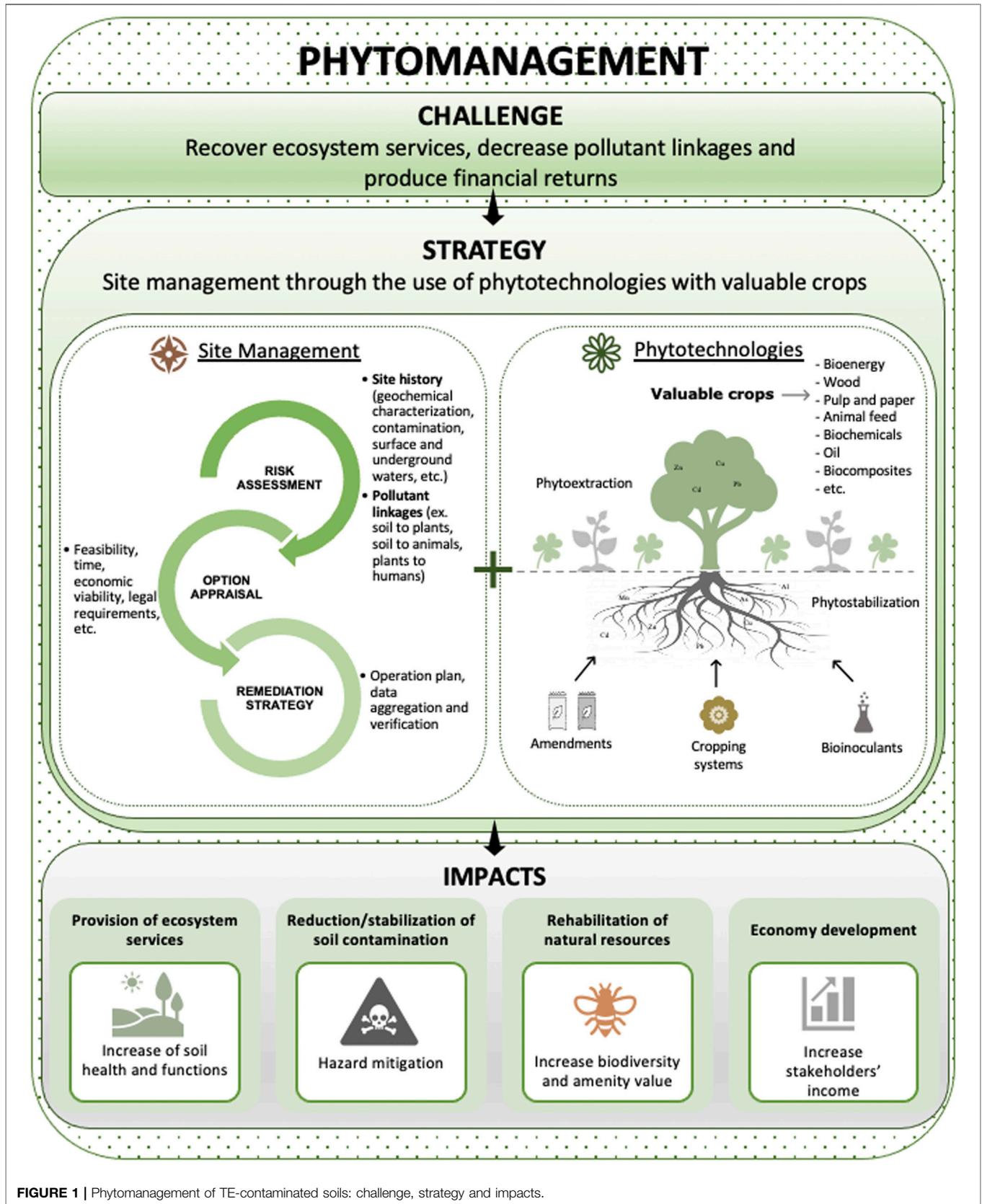
FIGURE 1 | Phytomanagement of TE-contaminated soils: challenge, strategy and impacts.

complexion, precipitation, which are a function of soil properties (Kabata-Pendias, 2010). For instance, a soil with low pH can increase the ion species of some TE (e.g., metallic cations) dissolved in soil solution, rendering them more bioavailable (Young, 2013). Similarly, climatic conditions also pose critical limitations to phytomanagement success. Temperature shapes plant transpiration, growth and metabolism, as well as water chemistry, thus directly affecting both contaminant uptake and its fate in plant parts and other ecosystem compartments (Bhargava et al., 2012). Likewise, moisture affects plant growth, faunal and microbial activity, and contaminant transport within soil. Prolonged drought induces plant stress, enhancing plant sensitivity to pathogens and herbivory and, notably, reducing plant growth (Kidd et al., 2015). Hence, water management needs to be carefully considered, especially in arid and semi-arid areas that undergo relatively long periods of drought and/or heatwaves, and soils with low water holding capacity (Kidd et al., 2015). Therefore, only plants that successfully withstand all these conditions can be used to phytomanage a contaminated site, especially in the current scenario of climate change. In addition, under such conditions, the selected crops need to achieve economically-viable yields and their products should be non-potentially toxic and fit to the quality standards. A large proportion of the aboveground biomass harvested at phytomanaged sites is from meta(loid)-excluders and therefore free of metal(loid)s excess. Also, root-to-shoot transfer of organic contaminants is frequently low, except for some xenobiotics such as organochlorines. Conversely, some elements (e.g., Ca, Si, Mg, etc.), as well as biomass moisture, may be of concern for the energy sector (Nsanganwimana et al., 2013; Nsanganwimana et al., 2014). Regarding combustion and pyrolysis, both intrinsic alkali metals (e.g., K and Ca) and silica present in the material (which can react to form alkali silicates), and contamination of the harvested biomass by soil, may induce operational problems such as 'slag' formation. In particular, for some crops, such as Miscanthus x giganteus and Arundo donax, a delayed harvest can reduce undesirable components (K, Ca, P, S, and N) in the biomass.

According to some studies, TE excess in the biomass can induce changes in heavy hydrocarbons present in tars, bio-oil yield, ash content, and relative evolution of CO_2 and H_2 in volatiles (reduced CO content) (Edgar et al., 2021). For poplar and willow short rotation coppice (SRC), Zn and Cd concentrations are higher in bark than in wood, decreasing in older branches and trunk. In addition, foliar Zn and Cd concentrations can decrease with growth and successive cuts. Therefore, the selection of the harvested shoot parts and their age are an important factor (Grignet et al., 2020; Grzegórska et al., 2020). Trees growing at brownfield and landfill sites can exhibit higher lignin content than those cultivated in uncontaminated soils due to abiotic stresses, e.g., drought-stress, leading to lower glucose yield (Edgar et al., 2021). In contrast, vetiver plants exposed to Cu excess can display a decrease in lignin and an increase in hemicellulose and cellulose

contents, leading to a higher production rate of bioethanol (Geiger et al., 2019).

Metal(loid)-enriched biomass can be processed by torrefaction and pyrolysis for producing biofuels and tars (Bert et al., 2017). Accordingly, the potential emission of volatile TE chemicals at high pyrolysis temperatures, the potential leaching of minerals and organics from chars, and the product quality of the products deserve attention. The TE fate depends on complex and multifactorial processes for all technologies based on thermal conversion (e.g., incineration, pyro-gasification, and pyrolysis) (Edgar et al., 2021). Oxide-forming elements and refractory compounds are often found in ashes and tars. Capture of volatile Cd and As chemicals depends on the filter quality. Ecocatalysts prepared from hyperaccumulators are used in various ways but preparing them with the most essential elements (e.g., Zn, Cu, Co, and Mn) and the least non-essential elements (Cd, As) requires a strong selection of plant species (Clavé et al., 2016). Besides the energy sector (e.g., bioethanol, biodiesel, biogas, and heat), many chemical, physical and biological biomass-processing technologies are reported as pre-treatment and conversion technologies. In the case of anaerobic digestion, some TE excess (e.g., >500 mg Zn kg^{-1}, 20,000 mg Mn kg^{-1}) can decrease the methane content in biogas and daily methane production (Edgar et al., 2021). Essential oils from aromatic plants harvested at phytomanaged sites also did not show TE contamination (Raveau et al., 2020). Similarly, oilseeds from sunflower, hemp, and most Brassicaceae harvested at phytomanaged sites generally do not present TE excess, nor the oil for biodiesel production and other uses.

Several pre-treatments can separate the metal(loid)s from the biomass fraction of interest or on the contrary avoids their release during the process and limit their bioavailability in the biochars produced (Edgar et al., 2021): pre-mixing with chemicals (e.g., $MgCO_3$, $FeCl_3$ and $Fe(NO_3)_3$, CaO) before biomass pyrolysis (He et al., 2020); composting (except the methylation of Hg-chemicals); for anaerobic digestion and fermentation, pre-treatment with NaOH enhances the release of biogas and metals from straw; biomass pretreatments with either ethanol organosolv, soda or dilute acid (Asad et al., 2017) and steam explosion (Ziegler-Devin et al., 2019) to release TE before bioethanol production. Post-treatment of conversion products and platform chemicals is also an option (e.g., sorption of arsenicals by Fe hydroxides after solvolysis of Pteris vittata fronds (Carrier et al., 2011). Overall, selection of plant species and cultivars, agricultural practices, harvest timing, etc., can also improve the quality of the harvested biomass compared to the required standards of the biomass-processing technologies.

Phytomanagement of TE-contaminated sites is certainly expanding (Pérez and Eugenio, 2018), but it is still rarely chosen as a remediation technology when compared to conventional physicochemical methods of soil remediation (Kidd et al., 2015; Quintela-Sabarís et al., 2017). As a matter of fact, once a contaminated site is targeted for recovery, the most typical procedure is to engage in faster and more drastic solutions, generally involving the use of physicochemical techniques (e.g.,

soil replacement and soil washing) and civil engineering works (Ashraf et al., 2019), which often remove the target pollutants at the expense of the destruction of soil integrity and functionality. This is, at least partly, due to technical issues related to the implementation of phyto-based strategies, as well as to the perception of many stakeholders who have a low confidence on the reliability of phytotechnologies (Cundy et al., 2013, 2015; Reinikainen et al., 2016; Ramon and Lull, 2019). Contributing to this is the lack of 1) convincing pilot field applications of plant-based options and 2) specific legal frameworks (Cundy et al., 2016). In any event, phytomanagement can also be handled as a holding-strategy for unused and vacant contaminated sites (Moreira et al., 2021).

In summary, environmental and socioeconomic benefits of phytomanagement options largely depend on specific site requirements, such as the need for amendments and irrigation, specific agronomic techniques, maintenance costs, presence of biomass processing units nearby to decrease costs, etc. Economic benefits obtained from harvested biomass and from other potential end-products can be easily valued, but social and environmental benefits (e.g., ecosystem services) are much more difficult to calculate (Bardos et al., 2016; Kuppens et al., 2018). In any case, different sites with different contamination histories frequently require different technologies for their remediation and recovery, for achieving the desirable goals and end-uses.

PLANT SELECTION

Phytomanagement uses plants that can withstand moderate or high bioavailable levels of TE (and of organic compounds), as well as other abiotic stresses while offering financial returns and environmental gains. The selection of the most suitable plant species for a contaminated site is therefore a critical point and depends on several factors, namely: 1) type, concentration, chemical speciation, bioavailability and location of soil contaminants; 2) physicochemical soil properties (e.g., structure, compaction, fertility, moisture, pH, OM, etc.); 3) water availability; and 4) climatic conditions (e.g., temperature, precipitation, wind, and altitude) (O'Connor et al., 2019); and 5) combined life-cycles of pests and biological auxiliaries. In particular, crops chosen for TE phytostabilization purposes should present a TE-excluder phenotype and avoid TE dispersion by leaching, water and wind erosion (Mench et al., 2007; Ruttens et al., 2006a,b; Mench et al., 2010; Vangronsveld et al., 2009). Plants stabilize TE by root uptake and accumulation, precipitation and adsorption, and by changing their chemical form through pH or redox potential modifications around roots (Mench et al., 2010; Burges et al., 2018; Yan et al., 2020). Conversely, plants meant for TE phytoextraction translocate the TE from roots to shoots and accumulate them in their aboveground tissues (Robinson et al., 2015; Remigio et al., 2020).

Traits such as the level of TE tolerance, growth rate, crop yield, type of life cycle (perennial, annual, and biennial), leaf habit (deciduous, evergreen), growth habit (grasses, shrubs, and

trees), water requirements, root depth, susceptibility to diseases/pests, etc. must be taken into account (Kidd et al., 2015). Also, the potential for volatilization (Hg and Se) (Ali et al., 2013) should be carefully anticipated. Importantly, for TE phytostabilization, plants should contain low levels of TE in the harvestable biomass (unlike for phytomining and bioavailable contaminant stripping). In any case, TE uptake and accumulation are TE- and host-specific, can be highly variable within plant species and their populations (different variants or cultivars) (Ruttens et al., 2011), and depends on site specificities. Plants should also be resilient to other abiotic factors often related with contaminated areas, which can include soil nutrient deficiencies, salinity, compaction, etc. (Kidd et al., 2015).

For TE-contaminated sites, local colonizing florae, described as metallophytes (endemic plants found in TE-rich soils) and pseudo-metallophytes (facultative metallophytes, i.e., plants with abilities to grow in both TE and non TE-rich soils) (Favas et al., 2014), should be firstly considered. They present specific traits resulting from the adaption to local harsh conditions that grant them advantages in plant establishment and growth. Furthermore, the use of local colonizing florae prevents ecological site-disturbances that potentially invasive/aggressive species may trigger, by competing with local species and/or acting as ecological disruptors. For instance, Jatropha curcas L., a TE-tolerant bioenergy crop is native in Mexico but in some countries, such as Indonesia, Australia and South Africa, is registered as invasive. This issue can be attenuated by using sterile cultivars, if available, to avoid further colonization, although propagation by stem cuttings is more difficult to prevent. The use of metallophytes in phytomanagement is usually thwarted by their typical low biomass, slow-growing nature and reduced economic value for stakeholders, except when used for phytomining (e.g., Ni by Allyssum species) (Chaney et al., 2007; Remigio et al., 2020). However, they could still be used, for example, as cover crops, intercrops and plant borders in phytomanagement initiatives. Conversely, some pseudo-metallophytes fit the phytomanagement purposes by presenting high biomass and growth rate, while also overcoming constraints posed by abiotic and biotic stresses.

In the past 10–15 years, some energy crops have arisen as most promising in adding value to TE-contaminated areas by generating biomass-based products (gaseous, liquid, and solid) (Grzegórska et al., 2020) that can be converted into different kinds of energy (heat, electricity, and fuel for transportation), while attaining environmental goals. Other industrial crops (e.g., fiber crops), aromatic plants, ornamental plants and biofortified crops are also perfectly suitable for phytomanagement, as one of the commitments of this phytotechnology is to deliver economic benefits for the end-users (Gonsalvesh et al., 2016; Pandey et al., 2016; Rizwan et al., 2018; Grottola et al., 2019; Sidhu et al., 2020).

Greenhouse and mesocosm experiments are often used to test promising plants and favorable clones and cultivars. Besides traditional breeding, mutation and somatic embryogenesis

TABLE 1 | Examples of field trials with economically valuable annual crops (sunflower, maize, tobacco).

Plant	Species/Cultivars/ Clones/Hybrids	Site	Duration	TE concentration in soil (mg kg^{-1})	Main results	Suggested uses for plants	References
Sunflower (*Helianthus annuus* L.)	*n.p.*	Former waste incineration plant (Czech Republic)	3 years	Cr (39–141), Ni (23–122), Cd (1.6–6.8), Pb (44–193)	– ↑ TE concentration in roots, followed by leaves – ↑ TF for Cd and Pb – ↓ Cd and Pb concentrations in grains	*n.p.*	Kacálková et al. (2014)
	′Peredovick′	Uranium-mining-influenced area (Germany)	34–170 days	Cd (0.62), Co (23.8), Cr (50.9), Cu (45.2), Fe (42,700), Mn (989), Ni (64.8), Pb (15.9), Th (9.05), U (7.75), Zn (79.4)	– ↑ Growth up to 140 days (in acidic soils) – ↑ TE extraction over time – BCF > 1 for Cd – BCF > 1 for all elements, except for U when considering the ratio with bioavailabe TE in soil	Biomass for energy	Kötschau et al. (2014)
	Mutant line families (resulting of chemical mutagenesis for TE tolerance of inbred line IBL04)	Farmland near a former Zn-smelter (Belgium)	3 years	Cd (3.9–4.6), Zn (234), Pb (142)	– Similar DW over time for each mutant – Distinct DW among mutants – ↓ Aboveground yields – BCF > 1 for Zn, BCF < 1 for Pb for all mutants – BCF > 1 or < 1 for Cd depending on the mutant – ↑ Zn extraction – Pb < detection limit in seeds	Biomass conversion	Thijs et al. (2018)
	40 oilseed cv. (*e.g.,* ′S-9178′)	Farmlands contaminated by mining and other anthropogenic activities (China)	126 days	Cd (0.24/0.85)	– ↑ Biomass of several cv. – ↑ Grain yield in several cv. – Cd concentration ranked as follows: shoots > seeds > roots – TF > 1 in several cv., especially in acidic soils – ↓ Cd content in oil – ↑ Amount of oil in seeds of some cv. (especially ′S-9178′) with a very good amount of oleic acid (55.6–77.1%) and suitable for human consumption	Edible oil	Zehra et al. (2020a)
	40 germplasms (*e.g.,* ′G.P:8585′)	Farmlands contaminated by mining and other anthropogenic activities (China)	126 days	Pb (106.5)	– ↑ Biomass of several cv. – ↑ Pb content in plant tissues, ranked as follows: shoots > seeds > roots – TF > 1 in most cv. – ↓ Pb content in oil – Pb content of sunflower meal under the legal limits – ′G.P:8585′ coined as best germplasm for restoring moderately Pb-contaminated sites, with oil's Pb concentration below Chinese Food Safety Standards	Edible oil Sunflower meal	Zehra et al. (2020b)
Maize (*Zea mays* L.)	Z1-Z6	Area affected by former smelters activities (Belgium)	6 months	Cd (5.4–8.9), Zn (266–398), Pb (135–189)	– ↑ Biomass of some cv. – ↓ TE concentration in grains – ↑ Cd concentration in shoots (over European standards for maize fodder) – ↓ Zn concentration in topsoil	Animal feed (grain) Biogas Digestate	Meers et al. (2010)

(Continued on following page)

TABLE 1 | (*Continued*) Examples of field trials with economically valuable annual crops (sunflower, maize, tobacco).

Plant	Species/Cultivars/ Clones/Hybrids	Site	Duration	TE concentration in soil (mg kg^{-1})	Main results	Suggested uses for plants	References
	CT38 and HZ (sweet maize)	Field contaminated with disposal of city wastes and batteries (China)	75 days	Cd (1.4)	– Regular growth – Cd concentrations ranked as follows: sheath > root > lamina > stem > grain – ↓ Cd concentration in grain (< China legal threshold for human consumption)	Human feed	Xu et al. (2013)
	Z1-Z7 (energy maize)	Site affected by former smelters activities (Belgium)	5 months	Cd (4.7–6.5), Zn (210–339)	– ↑ DW variability among cv. – ↓ Cd and Zn concentrations in plant tissues ranked as follows: leaves > stem > grains – Average total TE concentrations in maize cultivars of 0.96 ± 0.29 mg Cd kg^{-1} and 219 ± 39 mg Zn kg^{-1} – Potential Cd and Zn removal of 19 ± 6 g Cd ha^{-1} y^{-1} and 4.3 ± 0.9 kg Zn ha^{-1} y^{-1}	Animal feed (grain) Biogas	Van Slycken et al. (2013b)
	n.p.	Former waste incineration plant (Czech Republic)	3 years	Cr (39–141), Ni (23–122), Cd (1.6–6.8), Pb (44–193)	– ↑ TE concentration in roots, followed by leaves – ↑ TF of Cd and Pb – ↓ Cr, Cd and Pb concentrations in grains	*n.p.*	Kacálková et al. (2014)
	'Bright Jean No. 7'	Pb-contaminated agricultural area (Taiwan)	1 year and 11 months	Pb (5844)	– ↑ Growth and biomass production – ↓ Pb concentrations in roots and shoots over planting periods – TF < 1 – ↑ BCF in roots followed by leaves > stems > bracteal leafs > cob > kernel – ↓ Pb content in grain (< EU legal threshold for animal feed)	Animal feed (grain) Thermal energy	Cheng et al. (2015)
	19 cv. of southern China	Cd-contaminated paddy field (China)	*n.p.*	Cd (1.64)	– Cd accumulation in plant tissues ranked as follows: roots > shoots > grain – DW, Cd accumulation and grain yield varied according to the cv. – ↑ Cd in aboveground tissues and high biomass of 2 cv. – ↓ Cd in grains of 3 cv. (< EU legal threshold for human consumption)	Human feed	Wang et al. (2016)
Tobacco (*Nicotiana tabacum* L.)	*n.p.*	Agricultural field (Thailand)	37 and 57 days	Cd (87)	– ↑ Cd concentration – TF > 1 – ↑ BAF – ↑ Total Cd uptake from day 37 to day 57 – Hyperaccumulation values for Cd (based on extractable TE)	*n.p.*	Khaokaew and Landrot (2015)
	'Virginia'	3 contaminated sites (Bulgaria)	3 years	Pb (4–116), Cd (0.4–3), Cu (15–399), Zn (35–280)	– ↑ Pb and Cd concentration in leaves in sites with high concentration of these TE – ↑ Cu concentration in leaves – Zn concentration ranked as follows: Leaves > blossoms > roots > stems	*n.p.*	Zaprjanova et al. (2010)
	'Virgina' and 'Burley'	Mine tailings (Serbia)	3–4 months	U (15.3)	– ↑ Concentration in lower leaves than upper leaves – ↑ Accumulation in leaves followed by stems – Hyperaccumulation values for U in both cv.	*n.p.*	Stojanović et al. (2012)

(Continued on following page)

TABLE 1 | (*Continued*) Examples of field trials with economically valuable annual crops (sunflower, maize, tobacco).

Plant	Species/Cultivars/Clones/Hybrids	Site	Duration	TE concentration in soil (mg kg^{-1})	Main results	Suggested uses for plants	References
	n.p.	Two agricultural fields (China)	60/80 days	Cd (0.59/1.38)	– ↑ BCF in lower leaves – Cd concentrations in tissues ranked as follows: Lower leaves > middle leaves > upper leaves > root > stalk (or > stalk > root) Cd concentration in lower and middle leaves after coppice – Leaves DW: 4.24 tonne ha^{-1} to 5.03 tonne ha^{-1} and stalks DW: 1.41 tonne ha^{-1} to 1.48 tonne ha^{-1} – Theoretical percentages of extracted Cd: 132.06 /203.91 g ha^{-1} – Theoretical extraction efficiency: 10%/6.7% – ↓ Total and available Cd in soil	Animal feed	Yang et al. (2017)
	Somaclonal tabacco variants (*e.g.*, ′NBCu104′ and ′NFCu719′)	Farmland contaminated by Zn-smelter fallout (Belgium)	3 years	Cd (3.9–4.6), Zn (234), Pb (142)	Variable biomass and TE accumulation over the years – Efficient Cd extraction – BCF > 1 for Zn and Cd in aerial parts – BCF < 1 for Pb in aerial parts	Biomass conversion	Thijs et al. (2018)

Abbreviations are as follows: n.p., not provided; TF, Translocation factor; BCF, Bioconcentration factor; BAF, Bioaccumulation factor; DW, dry weight; cv., cultivar; ↑, high/higher/increase; ↓, low/lower/decrease.

Values for TE concentrations are given for the upper topsoil layer (<25–30 cm); Soil TE concentration correspond to total/pseudototal in soil, unless indicated otherwise.

TF, BCF and BAF are calculated as follows: TF = TE shoot/TE root; BCF = TE plant part/TE soil (total); BAF = TE plant part/TE soil (available).

(somaclones) have been used to produce plants with higher tolerance to TE (Herzig et al., 2014; Kolbas et al., 2020). Nonetheless, scarce information still persists regarding the plant performances under natural field conditions. Indeed, TE uptake and accumulations patterns can vary between crops grown in pots and in field. Screening a large range of crop species and cultivars, under different edaphoclimatic conditions, is a prerequisite for assessing robust candidates with ecological restoration capacities for phytomanagement (Zapata-Carbonell et al., 2019). As aforementioned, the plants' life cycle is a factor that has to be taken into account when phytomanaging a contaminated site. Relying on annual plants can entail shortcomings as they have yearly costs related with, *e.g.*, soil ploughing/tillage, irrigation, fertilization and harvest. Additionally, tillage can increase TE dispersion (*e.g.*, through wind erosion and water runoff) and be potentially destructive for soils' surface microbial communities, mainly fungal (Pandey et al., 2015; Burges et al., 2018). Annual plants require rotations or other cropping patterns that, despite bringing benefits to soil health, can further increase the cost of the remediation strategy. Conversely, perennial crops can convey more advantages for remediation. They tend to have deeper roots, which contribute to soil stability, especially when remediating slopes and riverbanks, or when contamination reaches deeper soil layers. Perennial life-cycle plants also tend to be more nutrient-efficient than annual plants, yielding higher biomass and energy, and granting the additional benefit of providing higher carbon sequestration (Burges et al., 2018).

The following sections describe economically valuable crops that can be used for phytomanagement purposes, and their use in selected field trials reported in the period 2010–2020 is summarized in **Tables 1** and **2**.

Annual Crops
Sunflower (*Helianthus annuus* L.)
Native from northeastern United States (Harter et al., 2004), sunflower is an edible summer annual crop of the Asteraceae family widely cultivated across the globe. It prefers moist and well-drained soils but can maintain a relatively good performance throughout heatwaves and droughts. Alongside its usage for human and livestock consumption (Zehra et al., 2020a,b), sunflower is also a relevant raw feedstock for several technologies depending on the specific plant part under consideration: stalks can be used as insulating materials (Eschenhagen et al., 2019), to produce thermal energy (Wang et al., 2020), bioethanol (Ruiz et al., 2013; Camargo and Sene, 2014), biogas (Hesami et al., 2015), briquettes (Alaru et al., 2013), activated charcoal (Kolbas et al., 2011), and a wide variety of other biocomposites (Mathias et al., 2015; Eschenhagen et al., 2019); seed husks can be employed in the reinforcement of plastic products (Carus and Partanen, 2018); and seed oil can be used for biodiesel production (Kolbas et al., 2011; Ziebell et al., 2013)

TABLE 2 | Examples of field trials with economically valuable perennial and woody crops (miscanthus, poplar and willow).

Plant	Species/Cultivars/Clones/Hybrids	Site	Duration	TE concentration in soil (mg kg⁻¹)	Main results	Suggested uses for plants	References
Miscanthus (*Miscanthus* spp.)	*M. sinensis*	Tailing dam (China)	n.p.	As (418.7), Cd (8.01), Cr (85.1), Cu (15.6), Zn (92.5), Ni (43.5), Pb (15.1)	– TE under the normal range for terrestrial plants (except for As); – ↑ As concentration in roots (98% of total As concentration in plants); – TF < 1; – BCF <1	n.p.	Zhu et al. (2010)
	M. floridulus	Uranium mill tailings repository (China)	n.p.	Ba (4.43E3), U (23.3), Th (8.84), Ni (24.7), Sr (119), Pb (344)	– ↑ Ba and Ni concentrations in plant tissues; – TF < 1	n.p.	Li et al. (2011b)
	M. floridulus	Pb–Zn Mine (China)	n.p.	Pb (5 215), Mn (438), Cu (67), Zn (13,352), Fe (24,755), Cr (68)	– BCF < 1 for all TE; – TF > 1 for Pb and Cr; – TF < 1 for Zn, Fe, Mn and Cu	n.p.	Wang et al. (2012)
	Miscanthus sp.	Site near a former Pb and Zn smelter (France)	3 years	Zn (334.3), Pb (219.5), Cd (4.5), Cu (20.6)	– Soil EDTA-extractable TE ranked as follows: Pb > Cd > Cu > Zn; – ↓ Cu and Pb soil bioavailability when compared to the reference soil cultivated with an annual crop; – ↑ Soil organic C in the 20–50 and 200–2000 μm soil fractions; – TE soil availability was correlated with TE and carbon localizations in soil fractions	n.p.	Iqbal et al. (2013)
	M. x giganteus	2 sites in the vicinity of a former smelter processing sulphidic ores (Romania)	3 years	Pb (802/683), Cd (26/13.5), Zn (242/147), Cu (224/213)	– ↓ Cd and Pb concentration in aboveground tissues; – Good combustions properties of biomass (low ash content and no sulfur present)	Thermal energy biofuel	Barbu et al. (2013)
	M. x giganteus	Contaminated agricultural soil near a former Pb-Zn-Cd ore mining and processing plant (Poland)	6 years	Pb (547), Cd (20.8), Zn (2175)	– Potential production of 30 tonnes of biomass ha⁻¹; – ↑ TE concentration in aboveground parts; – ↓ TE in biomass over the experimental period; – Total TE content in aboveground tissues: 2.2–4.8 kg ha⁻¹ for Pb, 0.055 to 0.12 kg ha⁻¹ for Cd and 9 to 13 kg ha⁻¹ for Zn	Biomass for energy	Pogrzeba et al. (2013)
	M. x giganteus	Agricultural contaminated site (France)	8 months	Cd (3.6/8.8), Pb (226.4/486.3), Zn (301.2/511.8)	– Similar shoot yields to those of plants grown in uncontaminated soil (US); – ↑ Cd and Zn concentrations in all organs than those of plants grown in US; – ↑ Pb and Zn concentrations in leaves than in stems; – ↓ Soil bioavailable Pb concentration; – TE concentrations depended on element, organ and growth phase; – Nutrient concentration ranked as follows: K > N > P ≥ Ca > Mg > Na; – ↑ Nutrient concentration (except Na) in leaves than in stems at the maximum DW	Biomass for thermochemical conversion	Nsanganwimana et al. (2016)
	Miscanthus hybrids (*M. sinensis x sacchariflorus* 'GNT41' and 'GNT34'); *M.xg*	Arable land contaminated by smelting activities (Poland)	2 years	Pb (690), Cd (27), Zn (3,028)	– ↑ Biomass of all miscanthus (13–16 tonnes ha⁻¹ y⁻¹); – ↑ Shoot height of miscanthus hybrids when compared to *M.xg*; – ↑ Leaf Area Index and stem number of miscanthus hybrids; – ↓ Shoot Pb and Cd concentrations of miscanthus hybrids when compared to *M.xg*; – ↑ Water-conserving behavior of miscanthus hybrids	Biomass for energy	Rusinowski et al. (2019)
Poplar (*Populus* spp.)	*P. trichocarpa × deltoides*	Alluvial soil polluted by river sediments (new plantation) and polluted dredged sediment soil (old plantation) (Belgium)	7 years	Cd (4.4/9.4), Zn (272/986), Cu (25/1,219), Mn (510)	– ↓ Growth rate; – ↓ Transpiration rate; – ↑ Cd, Zn and Mn foliar concentrations over time in the new plantation; – ↑ TE concentration in leaves; – ↑ Cu concentration in leaves; – No changes in Cd, Zn and Mn foliar concentrations over time in the old plantation	n.p.	Lettens et al. (2011)
	P. deltoides x (P. trichocarpa x deltoides) ('Grimminge'); *P. trichocarpa x P. deltoides* ('Hoogvorst'); *P. deltoides x P. nigra* ('Muur')	Former maize field near a metal smelter (Belgium)	2.5 years	Zn (343), Cd (5.7), Cu (35), Pb (185),	– ↓ Poplar biomass; – ↑ Plant TE concentrations; – Chlorosis of leaves observed in some clones	Biomass for energy	Ruttens et al. (2011)
	P. euramericana (hybrids); *P. trichocarpa*	Sites near Zn and Pb smelters (France and near a Zn smelter (Germany))	> 18 years	Cd (0.8–69), Cu (17–94), Pb (110–3,500), Zn (540–11,400)	– ↓ Pb uptake; – ↑ Zn and Cd concentrations in leaves; – ↓ Cu and C concentration in bark and wood; – ↑ Ca and K concentration in bark and wood	Biomass for energy	Evangelou et al. (2012b)
	P. nigra x P. maximowiczii; P. nigra	Area of former waste incineration plant (Czech Republic)	2 years	Cd (0.53–1.3), Zn (96.1–174), Cu (24.4–80.8)	– Regular growth and no visuals symptoms of toxicity; – Cd and Zn concentration ranked as follows: Leaves > twigs > roots; – Cu concentration ranked as follows: twigs > leaves > roots; – BCF > 1 for Cd and Zn in leaves (in general); – BCF < 1 for Cu in leaves	n.p.	Kacálková et al. (2015)
	14 genotypes, e.g.: *P. trichocarpa x P. maximowiczii* ('Skado', 'Bakan'); *deltoides x nigra* ('Vesten', etc); *P. trichocarpa* ('Trichobel', etc)	Soil formerly irrigated with raw wastewaters (France)	4–5 years	Pb (74.7–484), Cd (1.98–4.44), Zn (314–692), Cu (69–218),	– ↑ variability in TE concentration in leaves among genotypes; – TE concentration in leaves ranked as follows: Mg > Zn > Mn = Fe > Cu > Cd; – ↑ Fe and Cu mean concentrations at the beginning of leaf development, remaining constant until leaf abscission; – Steady increase of Mn, Cd and Zn from leaf emergence until the end of the summer, and damped just before abscission; – ↓ Mg concentrations just before leaf abscission; – Fe, Mn, Zn and Cd were not remobilized during leaf senescence in any genotype	n.p.	Pottier et al. (2015)
		Farmland near a former Zn-smelter (Belgium)	3 years	Cd (4.1–8.4), Zn (234–488)	– Variable biomass between clones	Biomass for conversion	Thijs et al. (2018)

(Continued on following page)

TABLE 2 | (*Continued*) Examples of field trials with economically valuable perennial and woody crops (miscanthus, poplar and willow).

Plant	Species/Cultivars/Clones/Hybrids	Site	Duration	TE concentration in soil (mg kg⁻¹)	Main results	Suggested uses for plants	References
	e.g.: *P. trichocarpa* clones; *P. trichocarpa* x *trichocarpa*; *P. trichocarpa* x *P. maximowiczii*				– ↓ Biomass productivity (< 6 tonnes ha⁻¹ y⁻¹) – Variable biomass and TE accumulation over the years – Variable Zn and Cd concentrations between poplar clones		
	Populus deltoides x *P. nigra* ('Orion')	Area formerly used to electronic waste disposal (Italy)	4 years	Cd (0.8), Cu (64.5), Ni (26.8), Pb (75), Zn (286)	– TE concentration differed between bark and wood – ↑ Zn accumulation in the largest poplar plants (60 mm DBH) – ↑ Cd, Cu and Ni accumulation in slightly smaller plants (50 mm DBH) – ↑ Pb accumulation in smaller plants (40 mm DBH)	n.p.	Nissim et al. (2019)
	14 genotypes; ex.: *P. trichocarpa* x *P. maximowiczii* ('Skado', 'Bakan'); *P. deltoides* x *P. nigra* ('Vesten', etc); *P. trichocarpa* ('Trichobel', etc)	Area formerly irrigated with raw wastewaters (France)	4, 7, 10 years	Pb (74.7–484), Cd (1.98–4.44), Zn (314–692), Cu (69–218),	– ↑ Biomass of *P.* 'Vesten' – ↑ Variability in TE concentration among genotypes – ↓ Soil TE (Cd, Cu and Zn) concentration after 10 years of poplar growth – No change or increase of macronutrients concentrations after 10 years of poplar growth – ↑ TE accumulation in bark – ↑ TE accumulation in branches, namely Cd, Cu and Zn than in wood	Biomass for energy	Chalot et al. (2020)
Willow (*Salix* spp.)	*S. fragilis* ('Belgisch Rood'); *S. viminalis* ('Jorunn'); *S. viminalis* x *viminalis* ('Christina')	Former maize field near a metal smelter (Belgium)	2.5 years	Zn (343), Cd (5.7), Cu (35), Pb (185)	– Moderate biomass production – ↑ Plant TE concentration – No changes in soil TE concentration (either total or available) – Chlorosis of leaves observed in some cv.	n.p.	Ruttens et al. (2011)
	S. caprea; S. alba	Sites near Zn and Pb smelters (France) and 1 near a Zn smelter (Germany)	> 18 years	Cd (0.8–69), Cu (17–94), Pb (110–3,500), Zn (540–11,400)	– ↓ Pb uptake – ↑ Zn and Cd concentrations in leaves – ↓ Cu and C concentration in bark and wood – ↑ Ca and K concentration in bark and wood	n.p.	Evangelou et al. (2012b)
	e.g., *S. alba* 'Alba'; *S. viminalis*; *S. dasyclados; S. schwerinii* x *S. viminalis*)	Area contaminated by former smelter activities (Belgium)	4 years	Cd (6.5), Zn (377)	– ↑ Productivity during the first SR cycle (3.7 tonnes DW ha⁻¹ y⁻¹) – Potential removal of 72 g Cd and 2.0 kg Zn ha⁻¹ – ↑ Removal of Cd and Zn when leaves are included in the harvest	Biomass for energy	Van Slycken et al. (2013a)
	S. x *smithiana, S. rubens*	Area of former waste incineration plant (Czech Republic)	2 years	Cd (0.53–1.3), Zn (96.1–174), Cu (24.4–80.8)	– Regular growth and no visuals symptoms of toxicity – ↑ Cd and Zn in leaves – No differences in Cd and Zn accumulation between species – BCF > 1 for Cd and Zn in leaves (in general) – BCF < 1 for Cu	n.p.	Kacálková et al. (2015)
	S. triandra x *S. viminalis* ('Inger')	Former industrial landfill area (Sweden)	3 years	Cu (28–240), Pb (22–1,000), Zn (72–530)	– good growth, with yields in line with the expected productivity in commercial applications – ↓ Pb and Cu accumulation in leaves and twigs – ↑ Soil organic matter content – No significant differences in soil TE concentrations before and after willow cultivation (except for Pb in one plot, where a decrease was found) – ↑ Ecological conditions for nematodes – ↓ TE concentration in porewater during cultivation	Biomass for energy	Enell et al. (2016)
	S. purpurea ('Fish Creek'); *Salix* x *dasyclados* ('SV1'); *Salix miyabeana* ('SX67')	Military landfill (Canada)	3 years	NH₄NO₃-extractable TE: As (1.16), Cd (0.56), Cu (59), Ni (8.8), Pb (21), Zn (89)	– TE (except Cd and Zn) had a significant impact on biomass production – Low to moderate biomass production at the third year – TE in the whole plant ranked as follows: Zn > Cu > Cd > Ni, Pb > As – ↑ Zn and Cd concentration in leaves – ↑ Cu and Pb concentration in roots – General differences of TE contents in leaves among cv. – Stable TE concentration over time in shoots and leaves	n.p.	Courchesne et al. (2017)
	e.g., *S. alba; S. alba* x *alba; S. viminalis; S. viminalis* x *S. viminalis*	Farmland near a former Zn-smelter (Belgium)	8 years	Cd (4.1–8.4), Zn (234–488)	– Variable biomass between clones – Variable biomass and TE accumulation over the years – ↓ Biomass productivity (< 6 tonnes ha⁻¹ y⁻¹) – ↑ Extraction of Cd and Zn of some clones	Biomass conversion	Thijs et al. (2018)
	S. viminalis	Urban area (France)	3 years	Cd (1.66), Zn (617)	– Zn and Cd soil bioavailability was maintained over the 3 years – Regular growth of willows, with good health status – ↑ Survival rate – ↑ Foliar concentration of Zn and Cd in leaves – BCF > 1 for Cd and Zn in leaves	Green chemistry to produce Zn-ecocatalysts	Grignet et al. (2020)

Abbreviations are as follows: n.p., not provided; TF, Translocation factor; BCF, Bioconcentration factor; BAF, Bioaccumulation factor; DW, dry weight; cv., cultivar; ↑, high/higher/increase; ↓, low/lower/decrease; DBH, diameter at breast height; SR, shot rotation.

Values for TE concentrations are given for the upper topsoil layer (<25 cm); Soil TE concentration correspond to total/pseudototal in soil, unless indicated otherwise.

TF, BCF and BAF are calculated as follows: TF = TE shoot/TE root; BCF = TE plant part/TE soil (total); BAF = TE plant part/TE soil (available).

and medicinal purposes (Bashir et al., 2015). Sunflower can also be used to produce hydrogen fuel (Antonopoulou et al., 2016).

Besides these established industrial, commercial and medicinal applications, its high yields, TE-tolerance and capacity to generate a cluster of biomass-based products support the great potential of sunflower as an attractive crop for phytomanagement. Also, the low TE contents usually present in the seeds and oil from sunflower plants grown in contaminated areas points out to a limited risk of contaminating the food chain (Angelova et al., 2016; Mench et al., 2018), and fosters their use for, e.g., feeding livestock (Mench et al., 2018). Several sunflower cultivars show tolerance to As (Piracha et al., 2019), Cd (Thijs et al., 2018; Zehra et al., 2020a; **Table 1**), Cu (Kolbas et al., 2011, 2014; Mench et al., 2018), Cr (Aslam et al., 2014), Ni (Ahmad et al., 2011), Pb (Kacálková et al., 2014; Zehra et al., 2020b; **Table 1**), Zn (Herzig et al., 2014; Marques et al., 2013), U (Kötschau et al., 2014 - **Table 1**; Meng et al., 2018), Cs and Sr (Brooks, 1998). They are also able to successfully grow in multi-metal(loid) contaminated areas, as is the case of mining (Kötschau et al., 2014; **Table 1**), industrial (Herzig et al., 2014; Mench et al., 2018) and agricultural sites (Thijs et al., 2018; **Table 1**). Regardless of their tolerance, sunflowers' response to bioavailable TE excess can be highly variable among cultivars, with some showing higher TE extraction efficiency (Nehnevajova et al., 2005), which can limit crop yield. Therefore, efforts have been made to implement breeding programs aimed at developing new cultivars/hybrids with increased TE-tolerance. For instance, Cd and Zn concentrations contrasted by a factor of ≈2 in 14 sunflower cultivars grown in spiked solutions, with differences being mainly due to uptake and translocation patterns (Laporte et al., 2015). Likewise, attempts to develop variants with increased drought and salt tolerance have been performed (Kane et al., 2013), as these abiotic stressors are frequently found in TE-contaminated soils. Sunflower variants with high TE extraction/stabilization capacity can be obtained by chemical mutagenesis (EMS–ethyl-methane-sulfonate), with some of them being more effective in phytoextracting TE (Cd, Cu, Pb and Zn) in field trials, compared to their mother lines (Herzig et al., 2014; Kolbas et al., 2011; 2018; 2020). These variants also showed a higher activity of antioxidant enzymes (Nehnevajova et al., 2012). In any event, cropping systems are critical drivers of sunflower performance (Kolbas et al., 2011; Rizwan et al., 2016; Mench et al., 2018). For instance, crop rotation can reduce allelopathy and the spread of fungal diseases (Markell et al., 2015). Selected sunflower field trials are described in **Table 1**.

Maize (*Zea mays* L.)

Maize is an edible annual plant of the Poaceae family native to southwestern Mexico, where it was domesticated from Balsas Teosinte (Van Heerwaarden et al., 2011). Maize plants prefer fertile, well-drained and moisty soils, and are sensitive to frost, water logging and drought (Wuana and Okieimen, 2010). These liabilities, alongside the high water demand, pose limitations for the use of this crop to recover contaminated areas. Several

varieties and cultivars of maize are widely cultivated across the globe for food or livestock forage (Wuana and Okieimen, 2010). However, other valuable economic products can be obtained from maize such as bioethanol (Meers et al., 2010), biogas (Thewys et al., 2010; Van Slycken et al., 2013a - **Table 1**), digestate for soil conditioning (from which TE can also be recovered) (Meers et al., 2010; **Table 1**), biomass for production of electrical and thermal energy (Schreurs et al., 2011; Witters et al., 2012a,b; Van Slycken et al., 2013a; Cheng et al., 2015; Rizwan et al., 2017), sweeteners (Ranum et al., 2014), and starch for food and industrial applications (De Vasconcelos et al., 2013). Maize can also be used for the production of charcoal/biochar (Břendová et al., 2015).

Maize has been reported to be tolerant to several TE, e.g., Cd (Van Slycken et al., 2013a; Xu et al., 2013; Kacálková et al., 2014 - **Table 1**; Moreira et al., 2014; Rizwan et al., 2016), Cu (Jarausch-Wehrheim et al., 1996; Karczewska et al., 2009), Zn (Moreira et al., 2016a,b; Van Slycken et al., 2013a - **Table 1**), Pb (Cheng et al., 2015; **Table 1**), and Cr and Ni (Kacálková et al., 2014; **Table 1**). In fact, maize is usually recognized as a root TE accumulator (Li et al., 2009; Meers et al., 2005). However, this root-accumulating phenotype largely depends on soil properties and genetic variability. Its capacity to withstand high TE levels in soil (Wuana and Okieiman, 2010) coupled with its potential for the generation of biomass-based products, makes maize an interesting annual crop for phytomanagement. Kernels tend to have a lower TE content than stems and roots (Putwattana et al., 2015; Wang et al., 2016; Xu et al., 2013 - **Table 1**), making them potentially suitable for animal feeding if legal TE thresholds are not surpassed (Meers et al., 2010; Van Slycken et al., 2013a; Cheng et al., 2015; **Table 1**). Maize cultivars (e.g., Bright Jean Number 7) have been bred in the past decades to cope with rising temperatures and low water supply (Cheng et al., 2015; **Table 1**). Nonetheless, the use of maize for phytomanagement remains a challenge under the current climate change scenario, especially in dry areas, since it can imply higher costs to stakeholders due to its water requirements or lower yields in arid or semi-arid areas (Meers et al., 2010). High biomass producing cultivars have been developed (Meers et al., 2005) to meet the need for renewable energy sources. These high biomass cultivars are most adequate for biomass production in polluted areas, while mitigating the risk of the spread of contaminants to other environmental compartments. For instance, maize grown in a heavily Pb-contaminated area (6,000 mg kg^{-1}) in Taiwan could produce about ≈1545 GJ ha^{-1} y^{-1} of thermal energy or the combination of 25 tons of grain for livestock feeding plus the production of 1172 GJ ha^{-1} y^{-1} of thermal energy from the remain plant parts (Cheng et al., 2015). Meers et al. (2010) estimated that as much as around 119 to 166 GJ ha^{-1} y^{-1} of electrical and thermal energy could be generated from maize grown in a site contaminated with Cd, Zn, and Pb in Flanders (Belgium). This approach can reduce the emission of CO_2 by 21 tons ha^{-1} y^{-1}, when compared to the use of fossil fuel to generate the same amount of energy. Moderately contaminated agricultural areas could also be used to grow energy maize instead of fodder maize, without loss of income to farmers (Meers et al., 2010; Van Slycken

et al., 2013a; **Table 1**). Some selected field trials demonstrate the feasibility of growing of maize in TE-contaminated sites and are summarized in **Table 1**.

Tobacco (*Nicotiana tabacum* L.)

Tobacco is an annually grown herbaceous plant of the Solanaceae family native to South America (Goodspeed, 1954), which is mostly known for cigar production (Popova et al., 2018). Tobacco is a fast-growing crop that yields high biomass (Yang et al., 2017), has a relatively deep rooting system, and prefers well-drained and fertile moisty soils to grow. The optimal conditions for tobacco growth involve 12–13 h of light during its vegetative growth and a temperature around 23.5°C (Yang et al., 2018), which is why most of its production is found in tropical climates (Barla and Kumar, 2019). However, tobacco is a widely adaptable crop, with several cultivars planted in over 120 countries under different climatic conditions (Barla and Kumar, 2019). In the past ten years, tobacco has gathered interest as a bioenergy crop, as it has the potential to produce up to 170 tons of biomass per ha (Barla and Kumar, 2019). However, tobacco can have other applications: 1) its seeds yield up to 30–40% of its dry weight in oil, which can then be used for biodiesel production (Grisan et al., 2016; Barla and Kumar, 2019); 2) seed cakes, an oil extraction by-product rich in proteins and other nutrients (Popova et al., 2018), can be used as a source of protein to feed livestock (Rossi et al., 2013; Serrapica et al., 2019), for human dietary supplements and for cosmetic applications; 3) tobacco leaves can be used as source of protein for humans and animals, and as feedstock of amino acids and other for industrial applications (Yang et al., 2017; **Table 1** and 4) the biomass of whole aerial parts can be collected for biogas production (González-González and Cuadros, 2014), bioethanol (Asad et al., 2017), pellets (Rossi et al., 2013; Grisan et al., 2016), and the production of thermal energy through combustion. These important economic applications have fostered tobacco and tobacco-hybrid breeding programs, aimed at delivering high-biomass varieties with greater number of seeds with high oil content to improve its profitability.

Tobacco plants can tolerate and accumulate several TE, including Cd (Fässler et al., 2010a, b; Yang et al., 2017; Thijs et al., 2018—**Table 1**), which can be accumulated in high amounts in its aerial parts (Khaokaew and Landrot, 2015; **Table 1**), Zn (Herzig et al., 2014; Lyubenova et al., 2009), Cu (Keller et al., 2003; Kolbas et al., 2020); Pb (Yuan et al., 2011; Zaprjanova et al., 2010—**Table 1**), U (Stojanović et al., 2012; **Table 1** (Wu et al., 2020). However, tobacco can volatilize elemental Hg into the atmosphere (Mani and Kumar, 2014), which raises concerns about the environmental implications of using this plant species in areas rich in this non-essential element. TE-resistant somaclonal variants of tobacco have been selected and tested in Switzerland, Belgium, France, and Spain (Lyubenova et al., 2009; Vangronsveld et al., 2009; Herzig et al., 2014; Kidd et al., 2015;

Kolbas et al., 2020). The best-performing variants have shown higher shoot TE concentrations, i.e. 5-7-fold increase for Cu, 2-5-fold increase for Cd, and 1.5-fold increase for Zn, in hydroponics when compared to their mother lines, while differing in their antioxidant content (Guadagnini, 2000; Lyubenova et al., 2009). Such tobacco variants are strong candidates for non-food crop rotations in TE-contaminated soils, but their productivity depends on soil properties (Kolbas et al., 2020). Gonsalvesh et al. (2016) suggested that plant biomass with high TE concentrations should be subjected to pyrolysis, instead of direct combustion, when used for energy production, in order to avoid the release of TE into the environment and to reduce plant volume. These authors further suggested that Cd- and Zn-enriched tobacco biomass could be converted through slow pyrolysis and steam activation to biochar and activated carbon, which could then be used as effective adsorbents for Cr (VI) removal or other applications. Pretreatment (soda, organosolvents, and diluted acid) of metal-rich tobacco shoots is another option for producing bioethanol (Asad et al., 2017). Some field studies describing the potential of tobacco for the phytomanagement of TE-contaminated soils are summarized in **Table 1**.

Perennial Grass Crops
Miscanthus (*Miscanthus* spp.)

Miscanthus refers to a genus within the Poaceae family, native to eastern and southeastern of Asia and South Pacific, and comprises approximately 11–12 wild rhizomatous species, and several hybrids and cultivars (Hodkinson et al., 2015). This crop prefers well-drained soils and grows worldwide over tropical and moderate cold temperatures. *Miscanthus* have an excellent ability to adapt to a wide range of environmental conditions and several genotypes can thrive under low temperatures (Clifton-Brown et al., 2001); in saline (Chen et al., 2017) and dry areas (Van der Weijde et al., 2017), marginal lands and TE-contaminated soils (Li G. Y. et al., 2011; Nsanganwimana et al., 2014, 2015; Rusinowski et al., 2019; Tran et al., 2020). However, frost (Clifton-Brown et al., 2001; Zub et al., 2012), and drought can impair *Miscanthus* establishment and survival, especially during the first year (Arnoult and Brancourt-Humel, 2015). The most important features of this C_4 grass are its high biomass production capacity under low nutrient inputs (Lewandowski et al., 2016, 2018), reaching up 7 m height (Hodkinson et al., 2015), and its water-use efficiency (Heaton et al., 2010), both relevant traits for phytomanagement purposes. By the end of the growing season, *Miscanthus* can accumulate a large amount of nutrients in its rhizomes, which may reduce the use of fertilizers in the next growing season (Cadoux et al., 2012). Having a lifetime of 20–25 years (Lewandowski et al., 2003), this crop has long been used for several purposes, namely, for preventing wind and water erosion of soils due its extensive root system, and for preventing surface and groundwater contamination by controlling TE leaching (McCalmont et al., 2017). From an economic standpoint, *Miscanthus* can be used: 1) to produce pulp, fiber (Nsanganwimana et al., 2014; Acikel, 2011) and paper (Xue et al., 2015); 2) as thatching material (Stewart

et al., 2009); 3) as bedding for animals (Caslin et al., 2010); and 4) as feedstock for fibers (Acikel, 2011), particle composite boards (Park et al., 2012), polyethylene composites (Chupin et al., 2017), concrete and insulation applications (Eschenhagen et al., 2019), biochar (Houben et al., 2014) and activated carbons (Lim et al., 2019). Apart from these uses, *Miscanthus* is largely known as an important second-generation bioenergy crop (Yan et al., 2012; Pidlisnyuk et al., 2016), used to produce thermoelectric energy (Brosse et al., 2012; Barbu et al., 2013 - **Table 2**) and biofuels, such as bioethanol (Van der Weijde et al., 2017) and biogas (Thomas et al., 2019). In addition, it is known due to its contribution to increase C stocks and greenhouse gas mitigation (Zang et al., 2018).

Miscanthus plants are known to tolerate and/or accumulate TE such as As (Zhu et al., 2010; **Table 2**), Pb (Pogrzeba et al., 2013; Nsanganwimana et al., 2016; **Table 2**), Sb (Wanat et al., 2013), Cr, Al (Stewart et al., 2009), Cu, Sn, Cd, Zn (Galende et al., 2014; Bang et al., 2015), Ba and Ni (Li G. Y. et al., 2011; **Table 2**). These TEs are usually accumulated in higher concentrations in roots and rhizomes, followed by stems and leaves (Nsanganwimana et al., 2014). By preferentially accumulating TE in their belowground tissues (Bang et al., 2015; Pandey et al., 2016), *Miscanthus* biomass should *a priori* be safe for energy production (Pidlisnyuk et al., 2014). Among *Miscanthus*, *M. floridulus* (Li G. Y. et al., 2011; Barbu et al., 2013; **Table 2**), *M. sinensis*, *M. sacchariflorus* and *M.* x *giganteus* (sterile triploid hybrid from *M. sacchariflorus* and *M. sinensis; M.xg*) are well known to grow in contaminated soils (**Table 2**). Hybrids have been developed in breeding programs specifically for phytoremediation purposes, favoring traits such as resistance to drought and to high TE concentrations. Rusinowski et al. (2019) established two plantations of *Miscanthus* hybrids and one of *M.xg* in a TE-contaminated arable area and showed that hybrids had reduced transpiration rates and lower Pb and Cd concentration in tissues under drought stress than *M.xg* (see **Table 2** for more details). These results highlight the potential benefit of plant breeding programs in improving key plant attributes for phytoremediation purposes. However, plant yields can be highly variable among species, hybrids and clones of *Miscanthus* (Lewandowski et al., 2016). As *M.xg* usually express the highest biomass production (Pogrzeba et al., 2013; Arnoult and Brancourt-Humel, 2015; Smith et al., 2015), it is one of the most selected candidates for phytomanagement (Burges et al., 2018). Additional advantages of *M.xg* include its susceptibility to only a few pests and sterility, which limits its invasiveness capacity. Depending on the clone, *M.xg* and *M. sacchariflorus* may have high lignin content, making them suitable for thermochemical conversion processes. Positive effects of *Miscanthus* on contaminated lands include the increase of: 1) soil microbial biomass and activity (Al Souki et al., 2017), 2) density and diversity of soil macroinvertebrates (Hedde et al., 2013), 3) carbon sequestration (Christensen et al., 2016), and 4) long-term TE phytostabilization (Pavel et al., 2014) with few TE inputs from senescent leaves incorporation into the soils (Al Souki et al., 2020). An additional benefit of *Miscanthus* is the reduction of soil disturbance, as no tillage is needed for its

implementation and maintenance (Nsanganwimana et al., 2014). Nonetheless, the entrance of TE into food chains should be carefully considered. For instance, Zhu et al. (2010; **Table 2**) addressed potential threats involving *M. sinensis* in restoring TE-contaminated soils, as As can enter the food chain through its consumption by livestock. Further examples of *Miscanthus* use in phytomanagement are illustrated in **Table 2**.

Vetiver (*Chrysopogon zizanioides* (L.) Roberty)

Vetiver is a high-biomass perennial C_4 grass of the Poaceae family (Danh et al., 2009) native to India (Lavania, 2000) that can reach 2–3 m height (Truong, 2002). Vetiver has a dense, thick and deep-rooting system (3–4 m depth in the first year of growth) (Truong, 2002; Truong et al., 2008), making it extremely effective for land stabilization, water filtering and erosion control (Truong et al., 2008; Vargas et al., 2016; Gnansounou et al., 2017). It is widely cultivated in tropical areas but can be found worldwide because of its high adaptability to a wide range of temperatures (−14°C to +55°C), low nutrient requirements, and tolerance to extreme soil conditions, including pH (3.3–12.5) (Truong et al., 2008) and salinity (it can survive in soils with an electrical conductivity up to 47.5 dS m^{-1}) (Danh et al., 2012). Furthermore, vetiver tolerates high TE concentrations, especially Pb, Zn and Cr (Antiochia et al., 2007; Danh et al., 2012). However, it is sensitive to shading, which can limit its establishment and survival (Truong et al., 2008). One of the main applications of vetiver relates to the extraction of essential oil (known as Khus) from its roots, which can then be used in medicinal practices, aromatherapy, cosmetics and as food additive (Prakasa Rao et al., 2008; Lal, 2013). Khus has biocide properties, protecting other crops from detrimental fungi such as *Rhizoctonia solani* (Dubey et al., 2010), and can also act as termite repellent (Zhu et al., 2001). Apart from its ecological and economic benefits, vetiver can further be used as fodder for livestock (Falola et al., 2013), roof covering (Gnansounou et al., 2017), and to produce ropes, several types of handicrafts (Danh et al., 2012; Gnansounou et al., 2017) and pulp and paper (Darajeh et al., 2019). Furthermore, having a potential biomass production in optimal conditions of over 100 tonnes ha^{-1} yr^{-1}, vetiver can be used for *e.g.,* producing electricity, while providing environmental benefits through soil carbon sequestration (Danh et al., 2012). The combustion of two tonnes of dry vetiver is estimated to be similar to that of one tonne of coal, resulting in a very attractive financial return as vetiver production is cheaper than extracting and processing coal (Danh et al., 2012). Bioethanol and cellulolytic enzymes are other products that can be obtained from vetiver biomass (Subsamran et al., 2019). Generally, only sterile cultivars are used, including for phytomanagement purposes, due to its non-invasive nature (Wilde et al., 2005).

Vetiver is tolerant to Pb (Danh et al., 2012; Pidatala et al., 2016; Bahraminia et al., 2016), Cu and Zn (Hego et al., 2009; Vargas et al., 2016), B (Angin et al., 2008), Cr (Shahandeh and Hossner, 2000; Danh et al., 2012), Al (Truong, 1999; Danh et al., 2009), Ni (Prasad et al., 2014), Fe (Banerjee et al., 2016, 2019), As (Datta et al., 2011), Cd (Ondo Zue Abaga et al., 2014), and Hg (Danh et al., 2009; Lomonte et al., 2014). TE are usually accumulated in

vetiver roots (Phusantisampan et al., 2016), although some ecotypes show higher concentrations in shoots (e.g., Zn in the Ratchaburi ecotype) (Roongtanakiat and Sanoh, 2011), depending on site conditions and time. Moreover, vetiver plants showed the ability to accumulate ^{134}Cs, especially in roots, when grown in ^{134}Cs spiked solutions (Roongtanakiat and Akharawutchayanon, 2017). Therefore, the physiological and morphological characteristics of vetiver, coupled with the profitability of its biomass and derived products, make it an excellent candidate for the phytomanagement of TE-contaminated sites (Danh et al., 2012; Banerjee et al., 2016; 2019; Phusantisampan et al., 2016—**Table 4**; Gnansounou et al., 2017). Actually, vetiver has been advocated useful for revegetating Pb/Zn mine tailings (Wu et al., 2010) and other mine wastes (Danh et al., 2012). Arochas et al. (2010) showed that vetiver plants were able to grow in the tailings and waste rocks of a high-altitude Cu mine (3,500 m above sea level) and some could survive the cold winter for several years. However, these authors cautioned that higher survival rates required plants to be fertilized, irrigated and protected from grazing. Vetiver has shown potential for Cd phytoextraction (Ondo Zue Abaga et al., 2014) and for its use in gold tailings, to prevent dust storm and wind erosion, and mine rehabilitation (Danh et al., 2012). It was efficiently cultivated in Cu/PAH-contaminated soils at a wood preservation site for more than 10 years (Hego et al., 2009).

Woody Crops
Poplar (Populus spp.)
Poplars are typically deciduous trees of the Salicaceae family (Eckenwalder, 1996), native to North America (Dickmann and Kuzovkina, 2014), that comprise ca. 22–45 species (although this range is not consensual), as well as numerous cultivars and hybrids (Dickmann and Kuzovkina, 2014). Despite being mostly found in alluvial and riparian areas of the Northern Hemisphere, their high genetic diversity makes poplars very adaptable to other areas (Pandey et al., 2016). Currently, poplars are disseminated worldwide and can be found from the tropical regions of South America (e.g., Mexico) to central Asia (e.g., India). Nevertheless, poplars prefer temperate areas and moisty, well-drained and aerated alluvial soils (Stanturf and van Oosten, 2014) with a pH ranging from 5.0 to 7.5 (Baker and Broadfoot, 1979). Poplar trees have a deep and extensive rooting system (Guerra et al., 2011) and are one of the fastest growing trees, with some specimens reaching over 40–50 m height. For this reason, they provide an important contribution for global CO_2 emissions mitigation (Krzyżaniak et al., 2019) and soil carbon sequestration (Pandey et al., 2016). Poplar trees are usually sensitive to waterlogging (unless temporary), intensive shading and salinity (Stanturf et al., 2001; Stanturf and van Oosten, 2014), although some species and ecotypes can be tolerant to these conditions. For instance, P. euphratica Olivier, which is considered an important species for foresting saline and alkaline arid areas, is able to thrive under salt stress by changing leaf morphology, developing succulence and presenting a plethora of physiological and molecular adaptations (Chen and Polle,

2010). Poplars can be easily propagated by stem cuttings, which benefits its plantation under SRC where poplars are grown over a period of around 2–5 years cycles. As a bioenergy crop, poplar can be grown for the production of bioethanol (Littlewood et al., 2014) and hydrocarbon biofuel (Crawford et al., 2016), and is used to generate heat and electricity through combustion or gasification processes. A good quality gas (syngas) was produced from the gasification of the biomass of the genotype Monviso–P. generosa x P. nigra, grown in a soil contaminated with a mixture of organic and inorganic contaminants (Aghaalikhani et al., 2017; Ancona et al., 2017). These trees can also be grown over a larger period, e.g., 10–30 years, and then be used to produce biomass for other applications (Barontini et al., 2014), including timber, veneer, pulp and paper (Crawford et al., 2016), resin adhesives (Yang S. et al., 2015), biopolymers and phytochemicals (Devappa et al., 2015). The production of biomass is very variable and depends on poplar genotype, climate and soil conditions (Searle and Malins, 2014).

Poplars have the ability to grow in poor and TE-contaminated soils. They can tolerate TE such as As (Vamerali et al., 2009), Cr (Chandra and Kang, 2016), Cd and Pb (Ruttens et al., 2011 - **Table 1**; Redovniković et al., 2017), B (Robinson et al., 2007), Zn (Lettens et al., 2011; Evangelou et al., 2012b; Thijs et al., 2018; Nissim et al., 2019; **Table 2**), Cu (Borghi et al., 2008; Kacálková et al., 2015 - **Table 2**), Ni (Nissim et al., 2019), Fe (Baldantoni et al., 2014) and Hg (Assad et al., 2016). However, poplar species, cultivars and hybrids vary widely in TE tolerance, accumulation and translocation patterns (Migeon et al., 2009; Ruttens et al., 2011). Baldantoni et al. (2014) grew two TE-tolerant clones, P. nigra ′N12′ and P. alba ′AL22′, in a multi-contaminated soil and showed that the former had a significantly higher Cd concentration in all sampled organs, as well as a Pb translocation factor 50-fold higher, than the latter. Conversely, ′AL22′ showed greater Cu accumulation ability. Although certain TEs, such as Cd and Zn, are preferentially accumulated in poplar shoots (notably in the leaves and young bark, making them useful for phytoextraction), poplars tend to follow the pattern observed for other woody crops of retaining TE in roots (useful for phytostabilization), followed by leaves and stems (Shi et al., 2011; Baldantoni et al., 2014). Poplar species, hybrids and clones can differ in their TE tolerance, which can be detected by a reduction in photosynthesis. As an example, Chandra and Kang (2016) showed a reduction in the photosynthetic rates and pigment contents of poplar hybrids grown in Cd, Cu, Zn and Cr contaminated soil. Only one hybrid (′Eco 28′, P. x euroamericana) showed higher photosynthetic rate, chlorophyll and carotenoid contents at the highest TE concentration tested (500 mg kg^{-1}).

Based on field experiments, poplar SRC emerges as a promising option to phytomanage TE-contaminated soils, yielding high biomass production (Mench et al., 2010; Chalot et al., 2020; Zalesny et al., 2019; Padoan et al., 2020). As an example, a SRC of poplar cv. ′Skado′ was able to produce a stem biomass of over 35 tonnes DW ha^{-1} over a 4-years growth period when grown in a TE-contaminated site (Ciadamidaro et al., 2017). Poplars may also improve organic carbon content and

microbial activity in multi-TE contaminated soils (Ancona et al., 2017). Despite all its positive features, the deciduous nature of poplar trees results in TE-rich leaves to spillback into soil. Pottier et al. (2015) reported that although Mg is remobilized in poplar leaves, the concentration of Fe, Mn, Zn and Cd increases until abscission, which can be reintroduced in soil by OM decomposition (see **Table 2** for more details). Poplar hybrids are the most used for phytomanagement as these can be easily developed to exhibit the required traits, such as high biomass production, and tolerance to specific TE and local harsh conditions (Lettens et al., 2011; Ruttens et al., 2011; Evangelou et al., 2012b; Kacálková et al., 2015; Thijs et al., 2018). The use of poplar benefits from well-established sylviculture techniques (Zalesny et al., 2016), which can be translated or adapted to TE-contaminated areas. Some examples of poplar uses in TE-contaminated soils are given in **Table 2**.

Willow (*Salix* spp.)

Native to eastern Asia, willows belong to the Salicaceae family (Dickmann and Kuzovkina, 2014) and comprise *ca.* 450 species and over 200 hybrids (Argus, 1986; Newsholme, 1992). This genus is spread worldwide, although mostly found in the Northern Hemisphere (Argus, 1986). The general attributes and growth requirements of willow trees resemble those of poplars, but they are able to tolerate higher soil moisture, colder temperatures and higher salinity (Mirck and Volk, 2010). Besides, some willow species (*e.g., S. repens* L. and *S. caprea* L.) are adapted to thrive in arid areas (Dickmann and Kuzovkina, 2014). Willows have extensive and deep rooting systems (Licht and Isebrands, 2005), prefer a soil pH of 5.5–8.0 (Abrahamson et al., 2002), and can easily establish in floodplains and sandbars. They are regularly used to stabilize stream banks and prevent erosion (Ens et al., 2013; Karp, 2014). The commercial value of willow is often related to the production of several end-products which can be obtained from its biomass, such as timber, fibers for basketry, charcoal (Karp, 2014), biochar (Rasa et al., 2018), fodder (Lira et al., 2008), pellets, whips and wood chips (Stanturf and van Oosten, 2014), pulp and paper and wood flour (Barton-Pudlik and Czaja, 2018). Willow biomass can also be exploited to produce industrial chemicals, such as polymers and resins (Karp, 2014). Willows are easy to propagate by stem cuttings and can rapidly grow in SRC as poplars (Ens et al., 2013). This capacity has fostered plantations of fast-growing willows to provide biomass and lignocellulosic feedstock for renewable energy production (Mleczek et al., 2010), such as for heating and electricity, as well as for bioethanol (Ziegler-Devin et al., 2019) and biosynthetic natural-gas production (Norman and McManus, 2019). Short-rotation coppices can yield up to 30 tonnes ha^{-1} y^{-1}, depending on genotype, climate and soil conditions (Stolarski et al., 2015). Usually, a cycle of 2–5 years is applied in commercial plantations (Van Slycken et al., 2013b).

Willows are tolerant to a wide range of TE such as Sn (Dagher et al., 2020), Pb (Evangelou et al., 2012b; **Table 2**), Cd (Witters et al., 2009; Mleczek et al., 2010), Cu, Zn (Witters et al., 2009; Labrecque et al., 2020; Evangelou et al., 2012b - **Table 2**), Cr (Quaggiotti et al., 2007), Fe, Ni, As (Jha et al., 2017) and Hg (Mleczek et al., 2010). Analogously to poplars, the willow cultivars and clones used in phytoremediation can

display considerable differences in terms of biomass production, and TE tolerance and accumulation (Pulford et al., 2002; Migeon et al., 2009; Ruttens et al., 2011; Van Slycken et al., 2013b; **Table 2**). French et al. (2006) showed that five willow clones grown in TE-contaminated brownfields highly differed in their biomass production and TE (Cu, Cd and Zn) uptake. Two of those clones were particularly efficient at harvesting Cd, showing a 7 to 9- and 9 to 10-fold increase in TE accumulation in stems and leaves, respectively, compared to EDTA-extractable soil concentrations. Clones can also perform differently according to soil characteristics. Puschenreiter et al. (2013) tested a *S.* x *smithiana* Willd. clone in seven TE-contaminated soils (mainly with Zn, Cd and Pb) for *ca.* 2 years and, although this clone revealed to be a relevant Zn/Cd phytoextractor, its biomass and TE uptake decreased in soils with high bioavailable Zn and Cd levels, especially in low pH and carbonate-free soil. Given the wide trait variability among willow clones, it is essential to select the potential best-performing clones based on their TE tolerance, uptake efficiency (TE accumulating clones for phytoextraction vs. TE excluding clones for phytostabilization), TE translocation from roots to shoots, and biomass production (Pulford and Dickinson, 2005; Unterbrunner et al., 2007; Wieshammer et al., 2007). On the other hand, TE accumulation in willow can differ between the plant parts (roots, twigs, leaves, etc.). The ability of willow trees to recover contaminated sites has successfully been tested in field trials (*e.g.,* Van Ginneken et al., 2007; Vangronsveld et al., 2009; Mench et al., 2010; Kacálková et al., 2015; Kidd et al., 2015; Enell et al., 2016; Courchesne et al., 2017; Thijs et al., 2018; Xue et al., 2018; Grignet et al., 2020) and some examples are given in **Table 2**.

Other Crops Suitable for Phytomanagement

Beside the above-described crops, other plant species can be used as cash crops for the phytomanagement of TE-polluted soils. Interesting alternatives include: i) industrial hemp (*Cannabis sativa* L.), ii) kenaf (*Hibiscus cannabinus* L.), iii) flax (*Linum usitatissimum* L.), iv) jute (*Corchorus capsularis* L.), v) switchgrass (*Panicum virgatum* L.), vi) tall fescue (*Festuca arundinaceae* Schreb.), vii) giant reed (*A. donax* L.); vii) colonial bentgrass (*Agrostis capillaris* L.), viii) perennial ryegrass (*Lolium perenne* L.) ix) sudangrass (*Sorghum bicolor* (L.) Moench), x) reed canary grass (*Phalaris arundinaceae* L.), xi) physic nut (*J. curcas* L.), xii) castor (*Ricinus communis* L.), xiii) common nettle (*Urtica dioica* L.)*,* xiv) virginia fanpetals (*Sida hermaphrodita* (L.) Rusby, xv) pongamia (*Milletia pinnata* (L.) Panigrahi), xvi) babul (*Acacia nilotica* L.), and xvii) sesban (*Sesbania sesban* (L.) Merr.) (**Table 3**). Regrettably, most of these species remain poorly investigated regarding their abilities to phytomanage TE-contaminated sites, especially under real field conditions. In spite of limited data to support its wide application in contaminated soils, industrial hemp is seen as one of the most economically and environmentally rewarding bioenergy crops. This plant species offers a range of relevant features for phytomanagement, such as tolerance to high concentrations of TE in soil (*e.g.,* Cu, Cd, Pb, Zn, Cr and Ni)

(Rheay et al., 2020) and, depending on the specific cultivars and site conditions, its potential for phytostabilization or phytoextraction. In addition, industrial hemp can be established and grown with very low agrochemical inputs. Remarkably, all hemp tissues can be used for a wide range of applications, including building materials, textiles, animal bedding, organic compost and biofuel production (Alaru et al., 2013; Crini et al., 2020). Finally, its high cellulose content is a key factor for bioethanol production (Crini et al., 2020).

CROPPING SYSTEMS

Although contaminated soils tend to be strictly interpreted as soils with above background pollutant levels, they often entail other issues that cannot be neglected such as nutrient deficiency, low OM, poor structure, low pH and/or high salinity (Kidd et al., 2015). To tackle these problems and fully grasp the benefits of phytotechnologies to soil remediation and recovery, tailored cropping systems are specifically required. Suitable cropping systems must address the best spatiotemporal organization of cash crops and management practices. Alongside crop selection, cropping systems take into account: 1) crop sequence across years (e.g., monocultures, crop rotation), 2) cropping patterns, i.e. crop spatial arrangement (e.g., intercropping or co-cropping, winter cropping), and 3) management practices (e.g., tillage/no tillage; nutrient management, namely by the use of amendments, fertilizers, foliar sprays; irrigation; weeding) (Bégué et al., 2018). Inoculation of crops with potentially beneficial microorganisms can also be used within cropping systems to optimize remediation processes (see "Bioinoculants" Section). Assessing and weighting the trade-offs among all available cropping systems can critically increase phytomanagement efficiency (Vangronsveld et al., 2009; Kidd et al., 2015). An optimal cropping system should improve crop growth through an efficient use of nutrients and water, increase soil fertility, improve soil ecological functions, and ensure minimal adverse effects of TE on biological recipients (Kidd et al., 2015).

Monocultural options should be avoided during the phytomanagement of contaminated sites. Monocultures of annual crops tend to deplete soil nutrients because they exploit the same nutrient pool across multiple growing seasons. Therefore, monocultural options typically require fertilization (Mench et al., 2010; Herzig et al., 2014; Kidd et al., 2015). Additionally, crops used in monocultural practices are frequently clones, and consequently have low genetic diversity which, when combined with spatial homogeneity, reduces resilience to pests and diseases. For instance, when grown in monoculture, sunflower is commonly affected by white mold caused by the soil-borne fungus *Sclerotinia sclerotiorum* (Debaeke et al., 2014), while willows and poplars tend to be severely impacted by the insect *Chrysomela* sp. (Georgi and Müller, 2015). However, the impact of phytopathogens on sunflower can be minimized or even prevented by using crop rotation strategies with non-host crops (Mench et al., 2018). Similarly, using a mixture of cultivars aimed at increasing the genetic pool is a suitable approach for

willows and poplars (Reiss and Drinkwater, 2018). This latter approach has the additional advantage of helping plants to cope with adverse edaphoclimatic conditions. Monoculture systems negatively affect biodiversity with concomitant adverse effects for ecosystem recovery and resilience. Crop rotations and/or cropping patterns, including with different cash crops, can improve the recovery of contaminated soils by influencing, for instance, TE bioavailability and crop production (Kidd et al., 2015). Greger and Landberg (2015) showed that growing willow (*S. viminalis* L.) for 4 years in Cd-contaminated soils led to lower Cd concentrations and increased nutrient content in wheat plants (*Triticum aestivum* L.) grown later at the same site.

Cover crops are also widely used under rotational and cropping patterns mainly to: 1) increase soil fertility, 2) enhance N and C sequestration (Kaye and Quemada, 2017; García-González et al., 2018); 3) decrease soil erosion (Gómez et al., 2018); 4) improve soil OM (Saleem et al., 2020); 5) improve soil moisture and water quality (García-González et al., 2018); and 6) control weeds and increase the yield of cash crops (Wittwer et al., 2017). Furthermore, cover crops can be used for establishing vegetated borders (Saleem M. et al., 2020). Some of the most used cover crops in contaminated sites are legumes (Fabaceae family), which can endure in N-depleted soils due to their ability to fix atmospheric N_2 through a symbiotic relationship with N_2-fixing bacteria (rhizobia) on their roots. Nitrogen compounds delivered to the plants are used for growth and development and, in return, rhizobia receive carbohydrates from the plants. At the end of the legume life cycle, N is released to soil. Legumes used in contaminated soils include fava bean (*Vicia faba* L.), common vetch (*V. sativa* L.), hairy vetch (*V. vilosa* L.), alfalfa (*Medicago sativa* L.), red clover (*Trifolium pratense* L.), white clover (*T. repens* L.), birdsfoot trefoil (*Locus corniculatus* L.) and false indigo (*Amorpha fruticosa* L). Saad et al. (2018) reported that, when growing in a co-cropping pattern, common vetch was responsible for the observed increase of biomass and Ni accumulation in the hyperaccumulator yellow tuft (*Alyssum murale* Waldst. & Kit.), as compared to fertilized and non-fertilized monoculture systems. Plants of the Brassicaceae family can also be used as cover crops; interestingly, some of them have potential to be used as cash crops. Examples include mustard (*Brassica juncea* (L.) Czern.), canola (*B. napus* L.) and forage radish (*Raphanus sativus* L.). *Brassica* species are fast-growers with high TE tolerance, including to Cd (Goswami and Das, 2015; Rizwan et al., 2018), U (Chen et al., 2020), Pb (Bouquet et al., 2017; Gurajala et al., 2019) and Zn (Belouchrani et al., 2016), and its biomass can be used for fuel production (Dhiman et al., 2016). Using a mixture of cover crops is often an efficient option for contaminated sites because they provide complementary benefits. However, the choice of compatible crops and the adjustment of seeding rates are key aspects to prevent overlapping.

Cropping patterns can enhance the recovery of microbial diversity and activity in contaminated soils. Gao et al. (2012) showed that the co-cropping of mustard and tall fescue mitigated the impact of Cd and Pb on soil microbial diversity and enzymatic activities, when compared to unplanted soils. This effect is related not only to the

secretion of multiple types of root exudates but also to a decrease in TE availability in soil driven by the accumulating capabilities of mustard (Gao et al., 2012). Likewise, Brereton et al. (2020) showed that the pairwise co-cropping of tall fescue, willow (*S. miyabeana* Seemen.) and alfalfa promoted higher diversity of rhizosphere bacteria than when each plant was grown in monoculture. Interestingly, this synergistic effect was not detected when all three crops were co-cropped.

Management practices strongly influence the effectiveness of phytotechnologies to recover polluted sites. Despite improving soil aeration and water drainage, tillage practices can affect TE availability. Vamerali et al. (2011) showed that ploughing a TE (As, Cd, Co, Cu, Pb, and Zn) contaminated pyrite waste, capped with non-polluted soil, increased TE availability and improved TE concentrations in sunflower, fodder radish, alfalfa and ryegrass. Tillage practices can also facilitate the proliferation of diseases. For instance, white mold sclerotia can decrease over time in non-till systems but remain constant in till-systems. No-tillage systems can also benefit the diversity and density of soil invertebrates in TE-contaminated soils (Hedde et al., 2013).

The management of TE contaminated sites may include the use of amendments to allow plant establishment and growth, decrease TE bioavailability, and improve soil properties and functions. Alternatively, amendments can be used to increase TE mobility and thus promote phytoextraction. Inorganic amendments (IA) comprise liming agents, phosphates (*e.g.,* H_3PO_4, KH_2PO_4, hydroxyapatite, and phosphate rock), Fe/Mn oxyhydroxides and iron grits, natural and synthetic zeolites, alumino-silicates, and cyclonic and fly ashes (Mench et al., 2007; Bolan et al., 2014; Kumpiene et al., 2008, Kumpiene, 2010, Kumpiene et al., 2019). Conversely, organic amendments (OA) include biosolids, green composts, livestock manures, biochar, etc. (Gómez-Sagasti et al., 2018; Kumpiene et al., 2019; Urra et al., 2019a,b). The advantages of OA over IA include their high accessibility and reduced costs. Organic byproducts and residues can be used as OA, although attention must be paid to their content in TEs, organic contaminants, microplastics and antibiotics. The use of OA can promote waste reuse, thus being aligned with the abovementioned CE paradigm. Additional benefits of OA are: 1) provision of macro- and micronutrients, which are commonly limiting factors for plant growth in many TE-contaminated sites; 2) improvement of soil structure and aeration; 3) enhancement of soil OM content; 4) stimulation of soil microorganisms and fauna (Burges et al., 2020); and 5) enhancement of soil moisture and water-holding capacity. Phusantisampan et al. (2016) showed that the application of a pig manure-derived OA to Cd contaminated soil improved its pH and increased its OM and macronutrient content, allowing a better growth of vetiver plants and their phytostabilization potential.

Other bench- and field-scale experiments have also described amendment-induced changes in TE availability and enhanced plant establishment and growth in TE-contaminated soils (Mench et al., 2010; Bolan et al., 2014;

Kumpiene et al., 2019). Pavel et al. (2014) added red mud to a soil contaminated with Zn, Cd and Pb to improve the growth of *M.xg* plants, and observed a significant reduction of soil TE concentration and plant TE uptake which benefited plant biomass (**Table 4**).

Other management practices to take into account are the adjustment of plant density, irrigation, harvest management, weed and pest control, and the use of pesticides. These practices were thoroughly reviewed in Kidd et al. (2015). However, most of the research has been carried out on the isolated effect of each of these practices, while their combination is only rarely investigated. Finally, the high variability of field site conditions and the wide panel of different amendment types and compositions, cropping systems and plant species complicate the extrapolation and generalization of results. Some examples of field trials with crops under different cropping patterns and/or ammendments application are given in **Table 4**.

BIOINOCULANTS

As stated, phytoremediation relies on the interactions between soil, plants, and associated microorganisms. Microorganisms are key players for plant growth under suboptimal soil conditions such as those found in soils with TE contamination. Therefore, the inoculation of beneficial microorganisms can help plant survival, establishment, and resilience, and they can also transform contaminants to less toxic forms or change their bioavailability, and consequently, plant uptake (Tiwari and Lata, 2018; Moreira et al., 2019). PGPB, including rhizosphere and endophytic bacteria, and AMF are known to effectively enhance plant growth in contaminated areas, fostering the success of phytomanagement. Some examples on the use of these bioinoculants in phytotechnologies are summarized below.

PGPB

PGPB help plants to cope with high concentrations of TE either by direct or indirect mechanisms acting simultaneously during the different stages of plant cycle (Glick, 2010, 2012; Saharan and Nehra, 2011; Kong and Glick, 2017). Direct mechanisms include 1) the synthesis of phytohormones, such as auxins (*e.g.,* IAA: indole-3-acetic acid), cytokinins, and gibberellins, which are pivotal in regulating plant growth and development (Glick, 2012), and 2) the activity of the enzyme 1-aminocyclopropane-1-carboxylate deaminase (ACC-deaminase). This enzyme hydrolyzes ACC (immediate precursor of ethylene) into α-ketobutyrate and ammonia, thereby reducing the endogenous levels of ethylene in plants, which is often induced by TE excess (Glick, 2014). Several authors have reported the beneficial effects of inoculation with ACC-deaminase and IAA-producing strains on plant growth under TE contamination. Shoot height (+40%) and root length (+100%) of sunflower plants grown on mining wastes increased when inoculated with the IAA and ACC-deaminase producing strain *Serratia* sp. k120 (Mendoza-Hernández et al., 2016). Likewise,

maize plants grown under increasing Pb concentrations and inoculated with bacterial strains having ACC-deaminase activity showed increased growth, photosynthetic pigments and proline contents, and improved antioxidative response (Hassan et al., 2014). Grobelak et al. (2018) also related the higher biomass of *B. napus* and *F. rubra* grown in a TE-contaminated soil to the high ACC-deaminase activity of the inoculated bacterial strains. PGPB can also directly favor plant nutrient uptake through the production of siderophores, solubilization of phosphorous, and biological N_2 fixation (Glick, 2012). Siderophores have the dual function of Fe chelation, facilitating its root absorption, and phytopathogen inhibition by limiting Fe availability (Ma et al., 2011a). Phosphorus is a key macronutrient for plant growth and development (Hawkesford et al., 2012) and hazardous TE concentrations may hinder P uptake by plants, limiting their growth. Phosphate solubilizing bacteria (PSB) can supply P to plants by converting insoluble inorganic phosphates into available forms through acidification, exchange reactions, and/or production of organic acids (Rodríguez and Fraga, 1999; Sharma et al., 2013). In turn, the mineralization of organic P can also occur by the action of phosphatases excreted by PSB (Richardson et al., 2009). Likewise, some PGPB (*e.g.*, rhizobia, free-living N_2-fixing bacteria) are able to transform atmospheric nitrogen into inorganic N forms that can then be taken up by the plants (Kong and Glick, 2017). Indirect plant growth-promoting traits are related to the PGPB ability to decrease or prevent the deleterious effects of phytopathogens on plants. Bacterial strains producing a plethora of compounds, namely antibiotics, hydrogen cyanide (HCN), and lytic enzymes are often used in biocontrol strategies for a broad spectrum of plant pathogens (Saharan and Nehra, 2011). Another positive effect of PGPB inoculation refers to the induction of systemic resistance (ISR) regulated by jasmonate and ethylene signaling pathways (Glick, 2012).

The deployment of efficient bacterial strains can greatly benefit phytomanagement strategies by reducing the harmful effects of hazardous TE levels on the growth of energy plants, including annual (Moreira et al., 2014, 2016a, 2016b, 2019; Gupta et al., 2018; Rosenkranz et al., 2018; Saran et al., 2020), perennial (Babu et al., 2015; Itusha et al., 2019) and woody crops (Wang et al., 2011; Cocozza et al., 2015; Janssen et al., 2015). A non-exhaustive list of the beneficial effects of PGPB, belonging to a large variety of genera (*e.g., Pseudomonas, Bacillus, Psychrobacter, Ralstonia, Chryseobacterium, Klebsiella, Aeromonas*, etc.), on energy crops grown in TE-contaminated soils is shown in **Table 5**. Bioinoculation can increase plant growth in contaminated soils, leading to biomass values similar to those reported for uncontaminated soils (Gupta et al., 2018). However, the effects of bacterial inoculation on plant growth and TE accumulation depend on many factors including strain selection, type and availability of TE, plant species/cultivar, and soil properties (Montalbán et al., 2017; Saran et al., 2020). Moreira et al. (2016a) reported that out of five TE-resistant PGPB inoculated in maize grown in a mine soil containing high Zn and Cd levels, only three efficiently promoted plant

growth. Moreover, bacterial inoculation influenced differently TE accumulation in roots, by decreasing Cd and increasing Zn. Likewise, Saran et al. (2020) observed that *H. petiolaris* plants inoculated with the strain *Bacillus proteolyticus* ST9 presented higher stem length and lower Pb accumulation than non-inoculated plants, whereas the inoculation of the strain *Bacillus paramycoides* ST29 decreased the accumulation of Cd and increased Pb content in roots. An alternative strategy to foster plant TE uptake is the combined use of ligands, such as ethylene diaminetetraacetic acid (EDTA), and PGPB. The synergistic use of EDTA and PGPB enhanced maize and sunflower dry biomass, as well as Cu accumulation (Abbaszadeh-Dahaji et al., 2019).

Some PGPB are effective at decreasing TE availability in soils through the release of extracellular polymeric substances (Guo et al., 2020) that can sequester TE and form stable complexes in soil. Moreira et al. (2016b) reported reduced Zn availability in the rhizosphere of maize inoculated with PGPB. Conversely, bioaugmentation-induced increases of TE availability in rhizosphere soils have also been reported (Sheng et al., 2012; Prapagdee et al., 2013; Arunakumara et al., 2014; Walpola et al., 2014; Babu et al., 2015; Gupta et al., 2018), which can be related to the secretion of siderophores by PGPB that bind TE enhancing their bioavailability and plant uptake (Rajkumar et al., 2010).

The production and accumulation of reactive oxygen species (ROS) in plant tissues triggered by TE hazardous concentrations may also have a negative impact on plant development (Fryzova et al., 2017). Plant responses to ROS lie on the activation of antioxidant defense mechanisms, including enzymatic and non-enzymatic strategies (Apel and Hirt, 2004). Several authors reported enhanced activities of ROS-scavenging enzymes, *e.g.,* superoxide dismutase (SOD), catalase (CAT), and ascorbate peroxidase (APx), when plants are exposed to high levels of TE (Ma et al., 2011b; Wang et al., 2011; Gupta et al., 2018; Saleem et al., 2018). The activity of these enzymes can be further improved by PGPB inoculation, which contributes to reduce oxidative damages in plants, as corroborated by the generalized decrease of malondialdehyde (MDA) contents often observed in inoculated plants (Wang et al., 2011; Gupta et al., 2018; Saleem et al., 2018; Jain et al., 2020).

AMF

AMF are soil-borne fungi that form symbiotic associations with most (80%) terrestrial plants (Smith and Read, 2008). These fungi form specialized structures in the cortical cells of the roots, called arbuscules, vesicles, and hyphae, constituting the intraradical mycelium (IRM). On the other hand, the external network of hyphae, also called extraradical mycelium (ERM) is responsible for translocating nutrients and water to the roots (Smith and Smith, 2011). In TE-polluted soils, AMF have a reinforced importance since they improve soil aggregation, reducing erosion and the risk of TE leaching (He et al., 2020a). Besides, TE can be deposited in the cell wall or the fungal vacuoles, preventing their uptake by plants (Agarwal et al., 2017; Gong and Tian, 2019). Likewise, AMF hyphae produce an insoluble

TABLE 3 | Other economically valuable crops that can be used in phytomanagement of TE-contaminated sites.

Plant	Family	Duration	TE tolerance/ accumulation	Potential products	References
Babul (*Acacia nilotica* L.)	Fabaceae	Perennial	Cd, U, Ni, Pb	Bioenergy; timber; livestock fodder; dye; gum	Sheokand et al. (2012); Fathi et al. (2014); Adhikesavan et al. (2015); Shabir et al. (2018)
Castor (*Ricinus communis* L.)	Euphorbiaceae	Perennial	Cd, Pb, Zn, Cu, Ni, As, Mn, Co	Bioenergy; biofuels; oil (with medical and industrial applications); fertilizers	Olivares et al. (2013); Bauddh et al. (2015); Yashim et al. (2016)
Colonial bentgrass (*Agrostis capillaris* L.)	Poaceae	Perennial	Pb, As, Fe, Zn, Cu	Livestock fodder or forage	Houben and Sonnet (2015); Rodríguez-Seijo et al. (2016); Touceda-González et al. (201; Lebrun et al. (2021)
Flax (*Linum usitatissimum* L.)	Linaceae	Annual	Zn, Cd, Cu, Pb, Ni	Fibers for paper and fabrics; oil; biocomposites; livestock fodder; bioenergy	Griga and Bjelková (2013); Saleem et al. (2020a)
Giant reed (*Arundo donax* L.)	Poaceae	Perennial	Zn, Cr, Cd, Pb, Ni, Cu, V, As, Hg	Bioenergy; fibers; biopolymers; paper and pulp; activated carbon	Sun et al. (2012); Nsanganwimana et al. (2013); Barbosa et al. (2015); Pirozzi et al. (2015); Christou et al. (2018); Cristaldi et al. (2020)
Hemp (*Cannabis sativa* L.)	Cannabaceae	Annual	Cd, Cr, Ni, Cu, Pb, Fe, Zn	Fibers for fabrics and textiles; livestock fodder or forage; animal bedding; biocomposites; paper, oil; cosmetics; paint; biofuels	Crini et al. (2020); Rheay et al. (2020)
Jute (*Corchorus capsularis* L.)	Malvaceae	Annual	Pb, Cr, Cd, Cu, Fe, Zn	Fibers; livestock fodder or forage; construction; furniture	Saleem et al. (2020b)
Kenaf (*Hibiscus cannabinus* L.)	Malvaceae	Annual	As, Cr, Pb, Cd, Zn	Fibers for paper, pulp, composite boards and textiles; edible oil; soap; paints; livestock fodder or forage; plastic compounds; wood products substitute; bioactive molecules; biofuels; bioenergy	Ho et al. (2008); Arbaoui et al. (2013); Ding et al. (2016); Ramesh (2016); Ayadi et al. (2017); Deng et al. (2017)
Perennial ryegrass (*Lolium perenne* L.)	Poaceae	Perennial	Cu, Cd, Ni, Hg, Pb, Zn	Bioenergy; livestock fodder or forage	Santibañez et al. (2012); Leudo et al. (2020); Li et al. (2020)
Physic nut (*Jatropha curcas* L.)	Euphorbiaceae	Perennial	As, Al, Cr, Zn, Cd, Pb, Ni	Bioenergy; oil	Marques and Nascimento (2013); Chang et al. (2014); Pandey et al. (2016)
Pongamia (*Milletia pinnata* (L.) Panigrahi)	Fabaceae	Perennial	As, Cr, Mn, Fe, Ni, Cu, Zn, Pb	Bioenergy; fibers for textiles and reinforced composites; oil cakes for enzyme production and animal feed; pesticides; phytochemicals	Mohankumara et al. (2020); Kumar et al. (2017); Sharma et al. (2020)
Reed canary grass (*Phalaris arundinaceae* L.)	Poaceae	Perennial	Zn, Pb, Cd, Co, Cr, Hg	Bioenergy; pulp and paper; forage	Polechońska and Klink (201; Lord (2015); Kolodziej et al. (2016); Vymazal (2016); Alhumade et al. (2019)
Sesban (*Sesbania sesban* (L.) Merr.)	Fabaceae	Perennial	Cr, Cd, Pb, Zn, Cu	Bioenergy; livestock fodder; green manure; phytochemicals; fibers	Gupta et al. (2011); Nigussie and Alemayehu (2013); Patra et al. (2020)
Sudangrass (*Sorghum bicolor* (L.) Moench)	Poaceae	Perennial	Al, Co, Mn, Ni, Pb, Zn, Cd, Sr, Cr, Cu, U	Bioenergy; livestock forage and fodder; fibers	Karimi (2013); Al Chami et al. (2015); Phieler et al. (2015); Wang et al. (2017)
Stinging nettle (*Urtica dioïca* L.)	Urticaceae	Perennial	Cr, Pb, Ni, Zn, Cd, Cu; Fe	Fibers for textiles and composite applications	Shams et al. (2010); Jeannin et al. (2020a), Jeannin et al. (2020b); Bislimi et al. (2021); Yung et al. (2021)
Switchgrass (*Panicum virgatum* L.)	Poaceae	Perennial	Cd, Pb, Cr, Cu	Fibers for pulp and paper; oil; thermoplastic composites; bioenergy	Keshwani and Cheng (2009); Juang et al. (2011); Li et al. (2011a); Pedroso et al. (2011); Arora et al. (2016); Guo et al. (2019)
Tall fescue (*Festuca arundinaceae* Schreb.)	Poaceae	Perennial	Pb, Cd, Zn, Ni	Livestock fodder or forage; bioenergy	Ferchaud et al. (2015); Lou et al. (2017); Steliga and Kluk (2020)
Virginia fanpetals (*Sida hermaphrodita* (L.) Rusby)	Malvaceae	Perennial	Zn, Pb	Fibers; livestock fodder; nectar for beekeeping; bioenergy	Kocoń and Matyka (2012)

glycoprotein, named glomalin, which forms complexes with TE and, ultimately, reduces their bioavailability (**Figure 2**) (Miransari, 2010; Gong and Tian, 2019).

AMF can remarkably promote plant growth by improving plant nutrition in TE-contaminated soils. The propagation of ERM in the vicinity of plant roots allows the exploitation of more soil volume, increasing nutrient and water supply to plants (Smith and Smith, 2011). Guo et al. (2013) found increases of 62, 359, and 99% in shoot N, P, and K contents, respectively, in AMF-inoculated maize plants grown in a mixture of mine tailings and topsoil (corresponding increases in the roots were 62, 256, and 141%). Remarkable increases in P accumulation (82% in roots and 300% in shoots) were also observed in AMF-

inoculated vetiver plants grown in a substrate containing coal mine wastes (Meyer et al., 2017). Plant growth promotion resulting from AMF inoculation has often been observed in energy crops growing in TE-contaminated soils (Merlos et al., 2016; Sun et al., 2018; He et al., 2020b, **Table 6**). Nonetheless, the effect of AMF inoculation on TE accumulation in plant tissues is highly variable (Wong et al., 2007; Guo et al., 2013; Meyer et al., 2017; Sun et al., 2018). For instance, the accumulation of TE in roots and shoots of vetiver plants varied according to the AMF species used for bioaugmentation (actually, some species increased TE accumulation while others decreased it) (Meyer et al., 2017). Guo et al. (2013) also reported that although AMF inoculation caused a generalized reduction of TE accumulation in maize tissues, *Glomus versiforme* was more effective than *G. mosseae* in this respect. Moreover, according to Sun et al. (2018), AMF inoculation has contrasting effects on TE accumulation in different switchgrass cultivars: inoculated highland cultivars showed a decrease of Cd accumulation, whereas lowland cultivars presented increased concentrations of Cd in their tissues. The effects of AMF on TE accumulation in plants is also highly conditioned by soil TE concentrations. Accordingly, Wong et al. (2007) observed greater accumulation of Pb and Zn in AMF-colonized plants grown under low soil Pb/Zn concentrations (0 and 10 mg kg^{-1}), while inoculated plants grown at higher concentrations (100 and 1,000 mg kg^{-1}) showed lower uptake values for both metals. AMF can also improve plant development under TE contamination by enhancing photosynthetic efficiency (Yang et al., 2015; He et al., 2020a) and antioxidative responses (Sharma et al., 2017).

Several studies demonstrate the efficiency of AMF bioaugmentation initiatives in the field. The addition of a commercial mycorrhizal inoculum (SolRize®) induced AMF root colonization and protected *Miscanthus × giganteus* against the detrimental effects of high concentrations of Cd, Pb, and Zn (Firmin et al., 2015). Mycorrhized plants showed reduced malondialdehyde (MDA) levels in leaves, suggesting that AMF colonization attenuated TE-induced oxidative stress. The same mycorrhizal inoculum (SolRize®) increased Cd and Zn accumulation in *M.xg* plants grown in TE-contaminated agricultural soils (Nsanganwimana et al., 2015). Likewise, AMF inoculation fostered TE phytostabilization by vetiver (Kafil et al., 2019). Interestingly, mycorrhizal inoculation (commercial inocula with a mixture of ecto- and endomycorrhiza fungi) enhanced, in average, 21% biomass yield of poplars cv. 'Skado' grown in two Cd, Cu, Pb, and Zn-contaminated fields (Ciadamidaro et al., 2017), and reduced Zn, Cu, Pb, and Cr translocation to leaves (Phanthavongsa et al., 2017). According to Guarino et al. (2018) the combined inoculation (PGPB and mycorrhyzal fungi) of poplar (*P. deltoides × P. nigra*) and willow (*S. purpurea* subsp. *lambertiana*) grown in TE-contaminated field led to a higher TE accumulation in its roots than in non-inoculated plants. Moreover, inoculated plants showed an increased antioxidant enzymatic activity.

Microbial Consortia

The inoculation of plants with a microbial consortia, comprising different groups of microorganisms, on plants exposed to different environmental stresses has deserved special attention in recent years (Oliveira et al., 2005a; Moreira et al., 2016b; Pereira et al., 2016; Mokarram-Kashtiban et al., 2019). Positive interactions can occur between PGPB and AMF, which may synergistically benefit plant growth under TE contamination. The interactive effects of PGPR and AMF on plant growth promotion involve a range of mechanisms (Artursson et al., 2006). AMF can benefit from the metabolites produced by bacterial strains, which increase root cell permeability and consequently root exudation, which in turn promotes hyphal growth and favors root colonization by fungi (Jeffries et al., 2003). Moreover, rhizosphere bacteria can support AMF by enhancing bioavailability of some macronutrients (*e.g.,* N and P), potentiating their role on nutrient acquisition by plants. For instance, N$_2$-fixing bacteria improve the N bioavailability to plants, which may be capitalized by AMF colonization (Barea et al., 2002). Similarly, in soils with low available P levels, PSB may support AMF by releasing phosphate ions from inorganic and organic compounds (Pereira and Castro, 2014), contributing to increase the available P pool in soil, and consequently P acquisition by plants (Smith and Read, 2008). Besides improving plant growth, positive interactions include the increase of their abundance in soil (Marschner and Timonen, 2005). AMF can directly influence the indigenous rhizobacterial community by promoting the production of root exudates, which supply compounds required for bacterial growth (Sood, 2003). On the other hand, AMF survival and root colonization may be improved by PGPB (Hildebrandt et al., 2006; Nadeem et al., 2014).

The application of microbial consortia to energy crops used for the phytomanagement of TE-contaminated soils is still underexplored. However, some encouraging results have been reported. Co-inoculation of the AMF *Rhizophagus irregularis* and the rhizobacterium *Pseudomonas fluorescens* in white willow plants exposed to Pb, Cu, and Cd greatly improved plant growth, TE uptake, and the general physiological status (Mokarram-Kashtiban et al., 2019). Likewise, the joint inoculation of two PGPB (*Chryseobacterium humi* ECP37 and *Pseudomonas reactans* EDP28) and the AMF *R. irregularis* increased the biomass of energy maize grown in a mine soil (Moreira et al., 2016b). These authors also found that microbial inoculation led to a general decrease in plant Zn accumulation, which was further evident in those treatments in which the AMF was present. This can be explained by the ability of AMF to adsorb and accumulate TE in their mycelia, limiting their absorption by plants.

Factors Affecting Bioinocula Performance

Once introduced into the soil, bioinocula have to face several constrains that can hinder their survival, persistence, and rhizocompetence (Bashan et al., 2014). Therefore, the selection of microbial strains well-adapted to the local edaphic conditions

TABLE 4 | Examples of field trials with economically valuable crops under different cropping patterns and/or amendments application.

Crop	Species/ Cultivars/ Clones/ Hybrids	Site	Duration	TE concentration in soil (mg kg^{-1})	Cropping patterns and/or amendments	Main results	Suggested uses for plants	References
Maize, sunflower and tobacco	*Zea mays* 'Magister'; *Helianthus annuus* 'Sanluca'; *Nicotiana tabacum* 'Burley 92'	Agricultural field (Switzerland)	6 years	Cd (1.35), Cu (477), Zn (652).	– Crop rotation; – Ammonium sulfate; elemental sulfur; nitrilotriacetic acid – Standard fertilization with N, P, K, Mg and S.	– Regular growth of crops, although variable over time, with no deficiency symptoms – ↑ Soluble Cd in soil in all treatments over time – ↑ Soluble Zn and insoluble Mg in plants under the elemental sulfur treatment – ↓ TE accumulation in plants of all treatments – ↑ Zn accumulation in all crops' shoots of all treatments – ↑ Cd in tobacco shoots in plants under the elemental sulfur treatment	Livestock feed, biofortification, biofuel production; green manure for micronutrient-deficient soils	Fässler et al. (2010b)
Sunflower	6 commercial cv. (ex. 'Countri', 'Energic') and 2 mutants (Mutant 1 and Mutant 2)	Former wood preservation site (France)	5 months	Cu (163–1,170), Cr (15.8–22.5), Zn (35–98), Ni (4.7–6.0), As (4.8–8.6), Co (1.5–2.0).	– Compost (poultry manure and pine bark chips (5% w/w) and dolomitic limestone (0.2% W/W)	– Neither sunflower cultivars nor mutants grew without the addition of the amendment – ↓ Cu soil bioavailability in the amendment's treatment – ↑ Sunflower growth in the amendment's treatment – Cultivars 'Energic' and 'Countri' with a comparable shoot and seed yields to other field trials in SW France, when grown at 200–400 mg Cu kg^{-1} in topsoil; – 'Energic' and 'Countri' cv. and mutant lines suitable for Cu phytoextraction	Livestock feed; biodiesel; hydrogen fuel; activated charcoal	Kolbas et al. (2011)
Miscanthus	*M.* x *giganteus*	3 sites in the vicinity of a former smelter processing sulphidic ores (Romania)	12 months	Cd (4.4–12.7), Zn (315–778), Pb (146–607).	– Red mud (1%)	– *M.xg* with high TE tolerance – ↑ Biomass production in less contaminated sites – ↑ *M.xg* growth in red mud amended soil when compared	Biomass production	Pavel et al. (2014)

TABLE 4 | (*Continued*) Examples of field trials with economically valuable crops under different cropping patterns and/or amendments application.

Crop	Species/ Cultivars/ Clones/ Hybrids	Site	Duration	TE concentration in soil (mg kg^{-1})	Cropping patterns and/or amendments	Main results	Suggested uses for plants	References
						to control (non-amended) – Red mud application increased soil pH and reduced TE bioavailability in soil – ↓ TE concentrations in plant tissues d with increasing distance from the source of pollution – ↓ TE accumulation in plant tissues ranked as follows: Zn > Pb > Cd in rhizomes > leaves > stems – BCF < 1 – ↓ TE concentrations in plant tissues under red mud treatment		
Poplar, Willow and Colonial bentgrass	*Populus nigra*; *Salix viminalis*; *Salix caprea* ′Mauerbach′; *Agrostis capillaris* ′Highland′	Cu-mine tailings (Spain)	3 years	Cu (308–1,155), Zn (95–366), Cr (68–140), Ni (27–181), Cd (0.8–2.3), Co (2.2–29.4), Mn (650–1,142), Al (33400–58900), Fe (81000–137000)	– Compost (municipal solid wastes and bark chippings)	– ↑ Soil pH over time – ↑ Soil CEC after 2 years – ↑ C and N content in soil over time – ↓ Soils' available Cu – ↑ Soil biochemical parameters and microbial parameters in vegetated areas – ↑ Soil enzyme activities over time – ↑ Soil enzyme activities, especially with *S. viminalis* – ↑ Cu concentration in leaves in the 3rd year – ↓ Zn concentration in leaves in the 3rd year	*n.p.*	Touceda-gonzález et al. (2017)
Vetiver	India and Sri Lanka ecotypes	Cd-contaminated area (Thailand)	9 months	Cd (0.9–35.7)	– Cow, pig, and bat manure and organic fertilizer	– No Cd toxicity symptoms were observed in both ecotypes – ↑ Plant height (specially in soils amended with organic fertilizer - India ecotype) – ↑ Biomass of Sri Lanka ecotype grown in soils	*n.p.*	Phusantisampan et al. (2016)

TABLE 4 | (*Continued*) Examples of field trials with economically valuable crops under different cropping patterns and/or amendments application.

Crop	Species/ Cultivars/ Clones/ Hybrids	Site	Duration	TE concentration in soil (mg kg^{-1})	Cropping patterns and/or amendments	Main results	Suggested uses for plants	References
						amended with bat, pig and cow manure – General improvement of soil physicochemical properties – ↓ Soil Cd concentrations – ↑ Cd concentrations in shoots and roots over time – ↑ Cd in roots (TF < 1) of Sri Lanka ecotype grown in amended soils (except in pig manure); – ↑ Cd in shoots (TF > 1) of India ecotype		
Sunflower	'LG545010 ES Ethic'	Former wood preservation site (France)	2008–2020	Cu (198–1,169)	– Compost of poultry manure + pine bark chips (OM); Dolomitic limestone (DL); Green waste compost (GW); P-spiked Linz-Donawitz basic slag (PLD); Carmeuse basic slag (CAR)	– ↑ Shoot DW yields, and Cu removals of plants grown in soils amend with compost (OM), alone and with DL than plants grown in PLD, CAR, GW treated soils – ↑ Organic matter content in compost-amended soils – ↓ Extractable Cu concentrations in compost-amended soils. – Shoot Cu concentration: 10–48 mg kg^{-1} – Shoot Cu removal: 26–88 ha^{-1} y^{-1}	Biomass-processing technologies	Mench et al. (2018)
Tobacco	Variant NBCu 10-8-F1 (C1), FoP clone (C2), BaG clone (C3)	Former wood preservation site (France)	2008–2010	Cu (21–1,290), Zn (35.2–98.4)	– Compost + dolomitic limestone (OMDL), compost + zerovalent iron grit (OMZ)	– Good survival rate and development of plants without visible toxicity symptoms – Shoot Cu removal: 13–173 g Cu ha^{-1} y^{-1} (cut 1) and 15–261 g Cu ha^{-1} y^{-1} (cut 2) – ↑ Shoot Cu removal in soils amended with compost + dolomitic limestone (OMDL) than in compost + zerovalent iron grit (OMZ) treated soils.	*n.p*	Kolbas et al. (2020)

TABLE 4 | (*Continued*) Examples of field trials with economically valuable crops under different cropping patterns and/or amendments application.

Crop	Species/ Cultivars/ Clones/ Hybrids	Site	Duration	TE concentration in soil (mg kg^{-1})	Cropping patterns and/or amendments	Main results	Suggested uses for plants	References
Sunflower	Three Eyebrown	Former mining area (China)	5 months	Pb (161), Cd (4.41), As (21.5), Zn (187).	– Lichee biochar (2.5, 5, and 10%)	– ↑ Sunflower biomass in 5% biochar treatment – Sunflower biomass increase was suppressed in 10% biochar treatment – ↑ Pb, Cd, and As accumulation in sunflower plants grown with biochar (5% biochar—highest Cd (42.3%) and As (110%) accumulation compared to control; 2.5% biochar - highest Pb accumulation (58.9%) compared to control – ↓ Zn accumulation in sunflower plants grown with 10% biochar – ↑ TE accumulation in leaves and receptacles in plants under biochar treatment – ↓ TE accumulation in roots, stems, and seeds in plants under biochar treatment – ↓ Pb, Cd, As, and Zn concentrations in the rhizosphere of sunflower plants under the 10% biochar treatment	n.p	Jun et al. (2020)

Abbreviations are as follows: n.p., not provided; TF, Translocation factor; DW, dry weight; cv., cultivar; ↑, high/higher/increase; ↓, low/lower/decrease; CEC, cation exchange capacity.
Values for TE concentrations are given for the upper topsoil layer (<25 cm); Soil TE concentration correspond to total/pseudototal in soil, unless indicated otherwise.
TF is calculated as follows: TF = TE shoot/TE root.

and their inclusion in an effective carrier are critical points for the success of phytotechnologies in the field (Oliveira et al., 2005b; Ortiz et al., 2015; Malusà et al., 2016). Microorganisms used as bioinoculants for phytomanagement strategies are generally retrieved from contaminated sites and then tested for their *in vitro* plant growth-promoting (PGP) traits (Pereira et al., 2015; Pires et al., 2017; Tirry et al., 2018). Usually, these PGP traits are evaluated in the absence of stress. However, PGP traits can be disrupted under TE hazardous concentrations, being of utmost importance to perform *in vitro* tests with relevant exposure to the stressors under consideration. Yet, only a few

studies have evaluated the ability of PGPB under TE stressful conditions (Moreira et al., 2016a; Mendoza-Hernández et al., 2016). The selection of PGPB should also have into consideration the TE-induced effects on *in vivo* seedling growth (Moreira et al., 2016a). Furthermore, the positive effect of PGPB on plant growth observed in TE spiked and/or sterilized soil can be overestimated, being possibly not transferable to real field conditions. In fact, it is recommended to use non-sterilized soil collected at the target site for the assessment of PGPB candidates (Moreira et al., 2019). This approach facilitates the identification of the most effective PGPB under more realistic conditions, including multi-element

contamination and competition with indigenous microbial communities.

The inocula size and its application rate are also relevant points. For instance, the application of low cell density inocula often results in low colonization rates, hampering competition with indigenous soil populations (Compant et al., 2010; Shabir et al., 2016). Furthermore, the size of the inoculum can influence TE accumulation in plants and should be considered in phytotechnologies where biomass is the most relevant outcome (Moreira et al., 2016a; Álvarez-López et al., 2016).

To maintain the viability of microbial inoculants and to ensure their efficiency, microbial strains should be incorporated in carriers. Although great progresses have been made in the last decade, the development of stable carrier formulations with ability to provide protection for bioinocula against abiotic and biotic threats is still a challenging task.

LEGAL AND REGULATORY FRAMEWORK

The legislation aimed at protecting soil from contamination mainly covers agricultural practices, *e.g.*, the use of sewage sludge and other organic wastes as amendments (Urra et al., 2019b, c), wastewater irrigation, application of pesticides and fertilizers, as well as industrial and commercial activities. At least 13 pieces of EU legislation apply to contaminated soils (Pérez and Sánchez, 2017). However, legislation/regulations for soil protection and contamination are frequently tailored to each country's views and needs (Ramon and Lull, 2019). As expected, many environmental, economic and social aspects influence the process of decision-making on contaminated land management (CLM) (Reinikainen et al., 2016; Castelo-Grande et al., 2018). These aspects must be rightly balanced in risk management strategies in order to properly define the sustainability of the outcome in policy frameworks. Policy frameworks and functional policy instruments (*e.g.*, inventories, surveys, regulations, guidelines, and funding mechanisms) are requested to reach sustainable practices and must involve environmental administrations and key stakeholders, *e.g.*, authorities, landowners, consultants, industries, communities and academics (Cundy et al., 2016; Reinikainen et al., 2016).

Many international policies have replaced the initial CLM option of a multifunctional land use (i.e., low risk tolerance, exhaustive remediation) by the concepts of risk-based and sustainable land management (SLM) (*e.g.*, Vegter et al., 2002; Paleari, 2017; MTES, 2018; Ramon and Lull, 2019). Risk-based land management is worldwide addressed in guidance documents, policy papers and case-studies (U. S. EPA, 2008; Ellis and Hadley, 2009; CL: AIRE, 2010; USACE, 2010; ITRC, 2011; Smith and Nadebaum, 2016). Sustainable land management is adopted in the CLM policy frameworks of many countries (*e.g.*, Europe, North-America, Australia and New Zealand) (AEP (Alberta Environment and Parks), 2017), targeted in the EU 7th Environment Action Program (Pérez and Sánchez, 2017), and present in international standards and guidance of ISO and ASTM International (Bardos et al., 2016). Several European countries (*e.g.*, Belgium, Estonia, Austria, Denmark, Sweden, Hungary, Switzerland, France, and Finland)

have set up long-term policy targets concerning the management and remediation of contaminated sites (Castelo-Grande et al., 2018; MTES, 2018; Pérez and Eugenio, 2018; Ramon and Lull, 2019). Legislation and issues involved in creating a contaminated site strategy for low- and middle-income countries were reviewed by Kovalick and Montgomery (2014, 2017). Most risk-oriented policies focus on the abandonment of those policies aimed at restoring soils to their original reference state, and consider the acceptable residual risks. Some national trigger values classifying soils as contaminated or requiring remediation have included the bioavailability concept explicitly (*e.g.*, United Kingdom, Belgium, and Switzerland) or implicitly (trigger values set according to the main influencing soil physicochemical properties, *e.g.* soil pH, granulometry, and OM content). Some phytomanagement options are aimed at removing the bioavailable contaminant fraction ("bioavailable contaminant stripping"), a target which significantly reduces the length of time required for remediation and addresses the pollutant linkages. Some European projects (*e.g.*, Rejuvenate, Greenland, and Hombre) developed decision support frameworks as a holistic approach to return marginal damaged/contaminated land to functional and economic use for producing biomass in line with the bioeconomy paradigm (Bardos et al., 2011; Cundy et al., 2013, 2015, 2016).

The Working Group on Contaminated Sites and Brownfields under the EIONET NRCs Soil set a *Land and Soil Indicator* (LSI003) for presenting the progress made in the management of contaminated sites by EU Member States and cooperating countries (Pérez and Sánchez, 2017). Relevant current barriers to SLM are not related to sustainability appraisal but the lack of regulatory requirements, client demand and economic considerations (Hou et al., 2014; Rizzo et al., 2016; Bardos et al., 2018). The lack of specific policy instruments and clear guidelines, *e.g.* on the reuse of contaminated soil, can impede sustainable remediation even in countries with well-developed policies and regulatory frameworks (Reinikainen et al., 2016). One important issue for risk-based land management is the perception of perpetual management of the hazard (source) when the chemical substances are not removed (Bagherifam et al., 2019). Consequently, questions of ongoing liability and transfer of information remain. This requires, from the start of the risk assessment process to 1) educate stakeholders, 2) allow communities to become contributors to scientific knowledge and 3) maintain collaboration with environmental authorities (Cundy et al., 2013). Prospects and threats regarding future land use (*e.g.*, options of site redevelopment and potential long-term liabilities), in line with economic and social uncertainties, influence the risk management strategy and the desired remediation level. Progress toward sustainable remediation is driven by three generic elements: 1) increased recognition of secondary environmental impacts of remediation, 2) stakeholder demand for more sustainable practices, and 3) institutional pressure that promotes sustainable practices (Hou et al., 2014).

DISCUSSION

Total cost of soil remediation in Europe is estimated up to 119 billion euro (Huysegoms et al., 2019). In addition, site

FIGURE 2 | Effects of AMF inoculation in plant development grown under TE-contaminated soils.
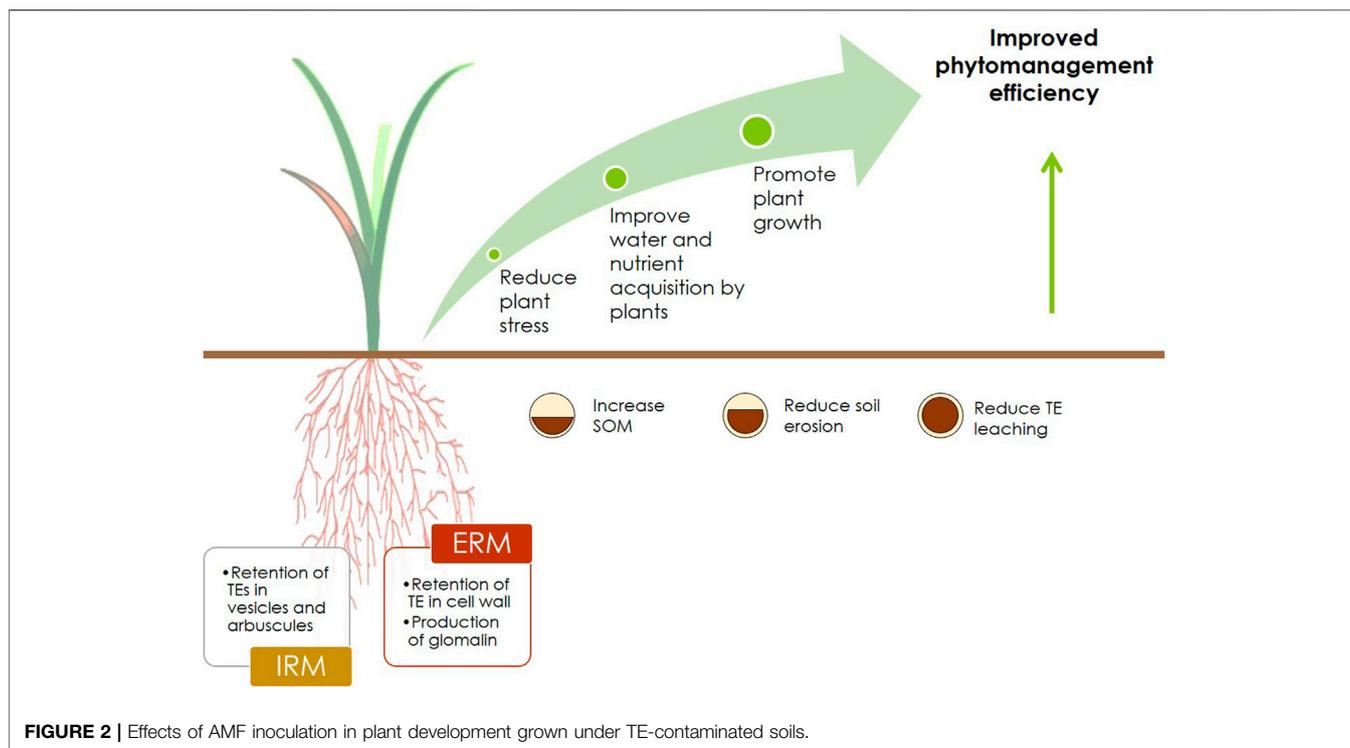
remediation is often affected by substantial cost overruns, due to misguided land use and wasteful delays in its reconversion (Barrieu et al., 2017). As outlined in this review, the commercial success of many phytotechnologies much relies on the subsequent processing of biomass for the production of valuable products. Long-term economic studies on the implementation and maintenance of phytomanagement strategies are highly needed to gather data and develop indicators on their economic sustainability, which could then be included in policy frameworks (Reinikainen et al., 2016). These indicators must include the economic benefits derived from the variety of ecosystem services provided by a vegetation cover, that are far from being easy to identify and quantify (Boerema et al., 2017), including avoiding pollutant dispersion and, consequently, reducing potential risks to human health (Burges et al., 2020). Currently, leading practitioners of restoration projects carry out investment analyses in order to estimate total financial costs, so that they can then choose the most convenient project (usually, by means of the application of conventional remediation technologies) from an economical point of view (Gerhardt et al., 2017). On the other hand, those focused only on contaminant remediation *per se* are usually more concerned with the values of specific environmental and technical indicators. A way to gather consensus is to firstly assess pollutant linkages and associated risks for conventional *vs.* nature-based remediation solutions, possibly integrating both of them in the management of polluted sites. Importantly, legislation on soil protection has to respect the principle of subsidiarity and proportionality, be in line with advanced research findings, and take into consideration energy expenditure and related issues such as climate change.

Life Cycle Assessment (LCA) has been proposed as an important tool to assess the sustainability of nature-based solutions (O'Connor et al., 2019; Visentin et al., 2019), and has provided valuable insights about positive and adverse impacts of land remediation. LCA studies clearly attested that the sustainable management of contaminated sites can provide positive environmental and socioeconomic outputs (Vigil et al., 2015; O'Connor et al., 2019), bringing meaningful benefits for a country's development. A central and up-to-date environmental benefit of soil rehabilitation encompasses climate change mitigation, through the increase of soil C sequestration (Paustian et al., 2019; Kasanke et al., 2021; Souki et al., 2021), which can help tackling the net-zero carbon targets in urban and sub-urban areas (Olsson et al., 2019; Bardos et al., 2020). Carbon neutrality is a multinational pledge that is gaining momentum and is supporting efforts toward including degraded land conversion actions in decarbonation strategies (Lask et al., 2021). Additionally, the worldwide scaling up of the demand for renewable energy can also help leveraging the recovery of degraded areas through phytomanagement, with the support and engagement of multiple stakeholders and investors. Finally, it would assist the production of energy for local use, in addition to contributing to the economic development of the region where phytomanagement takes place, through the creation of employment opportunities and the provision of a cheaper supply of bioenergy and biomass.

Several aspects still have to be addressed and improved for pushing forward the phytomanagement of contaminated soils. The technical skills for implementing phytotechnologies are

TABLE 5 | Effects of PGPB inoculation on growth and development of economically valuable crops grown under TE-contamination.

Host Plants	Bacterial inoculants	Source of bacterial inoculants	Growth-promoting traits	Type of inoculation	Contamination	Effect in inoculated host plants	References
Sunflower (*Helianthus annuus* L.)	*Psychrobacter* sp. SRS8	Rhizosphere of *A. serpyllifolium* colonizing serpentine soils	IAA, siderophore production, P solubilization	Immersion of roots in a bacterial suspension	Ni	- ↑ Plant growth (fresh/dry biomass) - ↑ Ni accumulation in roots and shoots - ↑ Nutrient accumulation (P, Fe) - ↑ Chlorophyll content - ↑ CAT, POD activities	Ma et al. (2011b)
	Micrococcus sp. MU1, *Klebsiella* sp. BAM1	Rhizosphere and roots of plants colonizing a Cd contaminated soil	IAA production, ACC-deaminase activity	Liquid inoculum sprayed on soil surface	Cd	- ↑ Plant growth (root length, plant dry biomass) - ↑ Cd in roots and leaves (*Micrococcus* sp. MU1) - ↑ Cd translocation from roots to shoots - ↑ Cd availability in soil	Prapagdee et al. (2013)
	Enterobacter ludwigii PSB 28	TE contaminated soil in an abandoned mine	P solubilization	Immersion of roots in a bacterial suspension	Co, Pb, Zn	- ↑ Plant growth (fresh/dry biomass) - ↑ TE accumulation in roots and shoots - ↑ TE translocation from roots to shoots - ↑ TE availability in soil	Arunakumara et al. (2014)
	Shewanella xiamenensis MH14	Tailings of an abandoned mine	*n.p.*	Immersion of roots in a bacterial suspension	Co, Pb, Cd	- ↑ Plant growth (fresh/dry biomass) - ↑ TE accumulation in roots and shoots - ↑ Cd translocation from roots to shoots - ↑ TE availability in soil	Walpola et al. (2014)
	Pseudomonas sp. CP5B21	Cr contaminated field in the vicinity of a tannery industry	IAA, siderophore, ammonia production, P solubilization	Seed coating	Cr (VI)	- ↑ Plant growth (root and shoot length, dry biomass) - ↑ Cr (VI) accumulation in roots and shoots - ↑ Nutrient uptake (N, P, Fe) - ↑ Chlorophyll content - ↑ Cr (VI) availability in soil - ↑ CAT, SOD, POD activities - ↓ MDA levels	Gupta et al. (2018)
	Pseudomonas gessardii BLP141, *P. fluorescens* A506, *P. fluorescens* LMG2189	*n.p.*	*n.p.*	Seed coating	Pb	- ↑ Plant growth and yield (root and shoot length, fresh/dry biomass	Saleem et al. (2018)

(Continued on following page)

TABLE 5 | (*Continued*) Effects of PGPB inoculation on growth and development of economically valuable crops grown under TE-contamination.

Host Plants	Bacterial inoculants	Source of bacterial inoculants	Growth-promoting traits	Type of inoculation	Contamination	Effect in inoculated host plants	References
						- ↑ Pb accumulation in roots - ↑ Chlorophyll and carotenoids content - ↑ Proline content - ↑ CAT, SOD, APX, GR activities - ↓ MDA levels	
Maize (*Zea mays L.*)	*Burkholderia* sp. GL12, *B. megaterium* JL35, *Sphingomonas* sp. YM22	Tissues of Cu-tolerant species *Elsholtzia splendens* and *Commelina communis*	IAA, siderophore production, ACC-deaminase activity	Liquid inoculum sprayed on soil surface	Cu	- ↑ Plant growth (dry biomass) - ↑ Cu accumulation in roots - ↑ Cu accumulation in shoots (strain JL35) - ↑ Cu availability in soil	Sheng et al. (2012)
	Ralstonia eutropha 1C2, *Chryseobacterium humi* ECP37	Industrial TE contamined soil	n.p.	Liquid inoculum sprayed on soil surface	Cd	- ↑ Plant growth (dry biomass) - ↑ Cd accumulation in roots - ↓ Cd accumulation in shoots - ↑ Cd availability in soil	Moreira et al. (2014)
	Rhodococcus erythropolis P30	Rhizosphere of *Festuca rubra* colonizing mine tailings	IAA production	.Seed bacterization .Soil inoculation: Liquid inoculum sprayed on soil surface .Dual soil inoculation .Seed bacterization + soil inoculation At two inocula sizes: 10^6 and 10^8 CFU ml^{-1}	Pb, Zn, Cd	- ↑ Plant growth (shoot dry biomass - soil inoculation - at 10^6) - ↑ Pb accumulation in shoots (soil; seed + soil inoculation - at 10^8) - ↑ Zn accumulation in shoots (seed inoculation - at 10^6) - ↑ Zn availability in soil (seed; seed + soil inoculation) - ↑ Cd availability in soil (seed; dual soil inoculation)	Álvarez-López et al. (2016)
	Serratia sp. ZTB15, ZTB24, ZTB28, ZTB29	Zn contaminated soil in a mining area	IAA, siderophore, ammonia production, phytase production, ACC-deaminase activity, P solubilization	Seed coating	Zn	- ↑ Plant growth (shoot and root length, root number) - ↓ Zn accumulation in roots and shoots - ↑ Chlorophyll content - ↑ CAT, SOD, POD, PPO, PAL activities	Jain et al. (2020)

TABLE 5 | (*Continued*) Effects of PGPB inoculation on growth and development of economically valuable crops grown under TE-contamination.

Host Plants	Bacterial inoculants	Source of bacterial inoculants	Growth-promoting traits	Type of inoculation	Contamination	Effect in inoculated host plants	References
Tobacco (*Nicotiana tabacum* L.)	*Rhodococcus erythropolis* P30, *Pseudomonas, constantinii* P29, *Bacillus subtilis* B32.34, *Streptomyces, costaricanus* RP92	- Rhizosphere of *F. rubra* colonizing mine tailings (P29, P30) - PAH/TE contaminated soil (B32.34) - Rhizoplane of *Cytisus striatus* colonizing a hexachlorocyclohexane contaminated soil (RP92)	IAA (P30, RP92, B32.34), siderophore (RP92, B32.34) production, P solubilization (P29), ACC-deaminase activity (RP92, B32.34)	Liquid inoculum sprayed on soil surface	Cu, Ni, Zn	- ↑ Plant growth (shoot dry biomass - strain P30; leaf number - strains P29, P30, RP2) - ↓ Ni accumulation in shoots (strain P30) - ↓ Zn accumulation in shoots (strains P29, P30) - ↑ Cu availability in soil (strain P30) - ↑ Ca availability in soil (strains P29, P30) - ↑ Mg availability in soil (strains P29, P30, B32.34) - ↓ P availability in soil (strains P29, P30)	Rosenkranz et al. (2018)
Vetiver (*Chrysopogon zizanioides* (L.) Roberty)	*Aeromonas* sp. VITJAN13	Rhizosphere of paddy fields	IAA, siderophore production, P solubilization	Liquid inoculum sprayed on soil surface	Cd	- ↑ Plant growth (root length, shoot height) - ↑ Cd accumulation in roots and shoots	Itusha et al. (2019)
Miscanthus (*Miscanthus* sp.)	*P. koreensis* AGB-1	Roots of *M. sinensis* colonizing mine-tailing soil	IAA production, P solubilization, N_2 fixation, ACC-deaminase activity	Liquid inoculum sprayed on soil surface	As, Cd, Cu, Pb, Zn	- ↑ Plant growth (height, number of culms and leaves, dry biomass) - ↑ TE accumulation in roots and shoots - ↑ TE availability in soil - ↑ Chlorophyll content - ↑ Protein content - ↑ SOD, CAT activities - ↓ MDA levels	Babu et al. (2015)
Willow (*Salix* spp.)	*Rahnella* Rh1, *Sphingobacterium* sp. Ro1, *Caulobacter* Ro2, *Curtobacterium sp.* TW1, *Pseudomonas sp.* TW2	Rhizosphere and tissues of *S. viminalis* and *S. alba* x *alba* clone	IAA, siderophore production, organic acid production, P solubilization	Immersion of roots in a bacterial suspension	Zn, Cd, Pb	- ↑ Plant growth (root, twig, leaves biomass - strains Rh1 and TW2) - ↑ Cd and Zn extraction by twigs (strain Rh1)	Janssen et al. (2015)
	Micrococcus sp. Con8.28, *Rhizobium radiobacter* MR28, *Massilia niastensis* P87, *Bacillus subtilis* B32.34	- Rhizosphere of *P. maximowiczii* x *P. trichocarpa* cv. ′Skado′ colonizing a PAH/TE contaminated site (Con8.28) - Roots of *Z. mays* colonizing a PAH/TE contaminated soil (MR28) - Rhizosphere of *F. rubra* colonizing mine tailings (P87)	IAA (Con8.28, B32.34, MR28, P87), siderophore (B32.34; MR28) production, P solubilization (MR28), ACC-deaminase activity (B32.34)	Liquid inoculum sprayed on soil surface	Cu, Ni, Zn	- ↑ Plant growth (root dry biomass - all strains; shoot dry biomass - strains MR28, P87, B32.34 - ↑ Ca, Mg accumulation (all strains) - ↑ K accumulation (strains MR28, B32.34)	Rosenkranz et al. (2018)

TABLE 5 | (*Continued*) Effects of PGPB inoculation on growth and development of economically valuable crops grown under TE-contamination.

Host Plants	Bacterial inoculants	Source of bacterial inoculants	Growth-promoting traits	Type of inoculation	Contamination	Effect in inoculated host plants	References
		- PAH/TE contaminated soil (B32.34)				- ↑ Zn availability in soil (strain P87) - ↓ Ni availability in soil (strains Con8.28, MR28, P87) - ↑ Ca availability in soil (all strains) - ↓ K, Mg availability in soil (all strains) - ↑ P availability in soil (strain P87)	
	Burkholderia sp. RX232, *Actinobacterium* sp. EX72	Rhizosphere and leaves of *S. caprea* colonizing a former Zn/Cd mine	Siderophore production, ACC-deaminase activity (strain RX232)	Liquid inoculum sprayed on soil surface	Cd, Zn	- ↑ Zn, Cd accumulation in leaves (strain EX72)	Kuffner et al. (2010)
Poplar (*Populus* sp.)	*Agrobacterium radiobacter* D14	Rhizosphere of *P. vittata* L.	IAA, siderophore production	Bacterial suspension mixed with soil	As	- ↑ Plant growth (height, root-collar diameter, root, stem, leaves dry biomass) - ↑ As accumulation in roots, stems and leaves - ↑ As translocation ratio - ↑ Chlorophyll content - ↑ Soluble sugar content in leaves - ↑ SOD and CAT activities in roots and leaves - ↓ POD activities in roots and leaves - ↓ MDA levels	Wang et al. (2011)
	Pseudomonas fluorescens BT4, commercial microbial consortia (Micosat F Fito®)	Compost and organic amendment (strain BT4)	*n.p.*	Bacterial suspension	Cd	- ↑ Plant growth (dry biomass - strain BT4, Micosat) - ↑ Cd accumulation in roots (strain BT4) and stems (Micosat) - ↑ Bioconcentration factor (Micosat)	Cocozza et al. (2014)
	Serratia marcescens	Natural wood peat	*n.p.*	Bacterial suspension mixed with substrate (two inoculum sizes: 1×10^7 and 2×10^7 CFU g soil^{-1})	Cd	- ↑ Plant growth (total dry weight; leaf area, perimeter, length, and width; stem height, and diameter) - ↓ Cd accumulation in leaf, stem and roots - ↓ Bioconcentration factors	Cocozza et al. (2015)

Abbreviations are as follows: n.p., not provided; CAT, catalase; SOD, superoxide dismutase; POD, peroxidase; APx, ascorbate peroxidase; GR, glutathione redutase; MDA, malondialdehyde; ↑ higher; ↓ lower.

TABLE 6 | Effects of AMF inoculation on growth and development of economically valuable crops grown under TE-contamination.

Host Plant	AMF species	Contamination	Effect in inoculated host plants	References
Sunflower (*Hellianthus annuus* L.)	*Glomus mosseae, G. intraradices*	Pb, Cd	- ↑ Plant growth (dry biomass, yield) - ↑ TE accumulation in roots and shoots	Adewole et al. (2010)
	F. mosseae	Ni	- ↑ Plant growth (dry biomass) - ↑ Fe, Zn, Mn accumulation in shoots - ↑ Ni accumulation in roots and shoots	Jarrah et al. (2019)
Maize (*Zea mays* L.)	*G. mosseae, Acaulospora laevis*	Cu, Cd	- ↑ Plant growth (height, root and shoot length, dry biomass) - ↑ Cu accumulation in roots - ↑ Cd accumulation in roots and shoots - ↑ Chlorophyll content - ↓ Proline content	Abdelmoneim et al. (2014)
	R. irregularis	Cu	- ↑ Plant growth (root fresh weight) - ↓ Cu accumulation in roots and shoots - ↑ Phytochelatins in shoots	Merlos et al. (2016)
	Claroideoglomus etunicatum	Cd, La	- ↑ Plant growth (dry biomass) - ↑ MDA levels - ↓ Cd accumulation in roots and shoots	Chang et al. (2018)
	F. mosseae	Cd	- ↑ Plant growth (root length, shoot height, dry biomass) - ↑ Chlorophyll content - ↑ Photosynthetic rate - ↑ Transpiration rate and stomatal conductance - ↓ Cd accumulation in shoots	He et al. (2020b)
Vetiver (*Chrysopogon zizanioides* (L.) Roberty)	*G. mosseae, G. intraradices*	Pb, Zn	- ↑ Plant growth (dry biomass) - ↓ TE accumulation in plants varied with soil TE concentration	Wong et al. (2007)
	Rhizophagus intraradices, G. versiforme	Pb	- ↑ Plant growth (dry biomass) - ↑ Pb accumulation in roots and shoots - ↑ P accumulation in shoots - ↑ Fe, Mn uptake of shoots (*R. irregularis*)	Bahraminia et al. (2016)
Miscanthus (*Miscanthus* sp.)	*G. margarita*	Pb	- ↑ Plant growth (root dry biomass) - ↑ Pb accumulation in root and shoots - ↑ Chlorophyll content - ↑ Fv/Fm total dry - ↑ IAA concentration in plant tissues - ↑ P, N uptake - ↓ H₂O₂ levels - ↓ POD activities	Sarkar et al. (2018)
Poplar (*Populus* sp.)	*R. irregularis*	Cd, Zn	- ↑ Plant growth (shoot dry biomass) - ↑ Transpiration rate - ↓ Cd accumulation in shoots	de Oliveira et al. (2020)

Abbreviations are as follows: POD, peroxidase; MDA, malondialdehyde; IAA, indole-acetic-acid; Fv/Fm, minimum fluorescence/maximum fluorescence ratio; ↑, high/higher; ↓, low/lower.

scattered and require an articulation of various disciplines. Priorities are to 1) guide the collection of case study data, and 2) identify and evaluate methods/tools to assess the recovery of soil functions and ecosystem services, without ignoring groundwater pollution during soil remediation. Relevant feasible phytomanagement solutions, focused on contaminant transfer and exposure pathways, must be assessed in the long-term under real field conditions, taking into account at least the site zoning, the conceptual remediation model, the targeted pollutant linkages and the future land use (before full deployment of solutions). Overall, such information will provide experience feedback on plant species (cash crops), plant communities, associated microbes, soil amendments and cultural management, in order to drive the trajectory toward a desired sustainable and resilient novel ecosystem. As many different case studies as possible, *e.g.,* rural and urban land,

brownfields, natural ecosystems including marshes, etc. must be evaluated. Methods currently used in Ecology and Ecotoxicology (including bio-monitoring techniques, guides to assess soil functions and ecosystem services, quantification of residual risks) must be adapted to assess those sites under and after phytomanagement. Besides, monitoring of control cards and archiving of samples is highly recommended. The recognition of secondary environmental impacts of remediation initiatives demands for a full assessment of traits over a range of conditions. Long-term economic studies that include revenues derived from biomass produced compared to the costs of inaction or remediation with alternative technologies, are a priority for the implementation and maintenance of phytomanagement. This calls for the multi-actor approach required under the circular economy paradigm where a chain of players from different

sectors must be connected. In fact, the success of phytomanagement greatly relies on the feed of the biomass produced into industrial processes for a range of applications (Puschenreiter et al., 2014; Evangelou et al., 2015; Cundy et al., 2016; Jiang et al., 2019). Nevertheless, the use of contaminated biomass for several purposes has been proven at the bench scale but its full-scale use has been yet less clearly demonstrated and is still constrained by a series of regulatory issues. Progress in phytomanagement acceptance across authorities, legislators and the industry sector depend, to a great extent, on the publication and outreach of successful case studies and the identification of reasons for failure. In this respect, scientists, consultants, authorities and landowners must work together to boost, and benefit from, the full potential of phytotechnologies, such as phytomanagement, for an efficient, economically viable and sustainable recovery of contaminated sites.

CONCLUSION

Managing contaminated land through phytomanagement holds great promise but it is still far from broad and large-scale implementation. Still, the application of nature-based solutions for the recovery of polluted sites is progressively increasing. Long-term field trials will shed light on the environmental, social and economic benefits, which will contribute to gain the trust of stakeholders and investors. The pursuit of soil restoration through a cost-effective and sustainable solution requires a shift in cultural and policies paradigms. Fortunately, halting and reversing land degradation, while ensuring restoration of ecosystem services, is nowadays strongly supported by multiple national and international organizations. In fact, in March 2019, the United Nations declared 2021–2030 the "Decade on Ecosystem Restoration". Phytomanagement strategies are intrinsically aligned with many of the UN's 2030 Sustainable Development Goals, but particularly with SDG #15, which aims to "protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification and halt and reverse land degradation and halt biodiversity loss." Furthermore, The European Parliament has just adopted a resolution for soil protection and the remediation of contaminated soils (European Parliament, 2021).

In this review, we summarized the benefits and drawbacks of phytomanaging metal (loid)-contaminated soils, discussed guidelines for selecting the most suitable crops, and described those used in phytomanagement (encompassing phytoremediation) studies with noteworthy benefits for soil functions. We have also reviewed different cropping patterns that could assist this practice, and addressed the use of bioinoculants, namely stress-tolerant PGPB and/or AMF that can fuel crop establishment and growth in TE-contaminated soils. Although holding great potential, there are still conflicts underlying the use of bioinoculants, which mainly come from unforeseen outcomes concerning the role they should fulfill. Nonetheless, we have identified positive outcomes brought by the bound of bioinoculants and crops. We have analyzed the legal and regulatory framework of soil management and discussed the constraints in the application of phytomanagement across borders, which mainly falls on a lack of common strategies and legislation. More field research is required for understanding the myriad of poorly understood links and mechanisms that still challenge the large-scale employment of phytomanagement. Finally, education and public awareness about soil health and climate change mitigation are of crucial importance to boost the demand for acute and appropriate actions from governments and societal organizations.

AUTHOR CONTRIBUTIONS

HM and SP conceptualized the review. HM, SP, PK, CG, and MM performed literature review and the critical analysis of the data. HM, SP, CG, and MM wrote sections of the manuscript. CG and PC revised the first draft. All authors contributed to manuscript revision, read, and approved the submitted version.

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