Chemosphere 168 (2017) 1230–1247

Contents lists available at ScienceDirect

Chemosphere

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

Review

Native herbaceous plant species with potential use in phytoremediation of heavy metals, spotlight on wetlands — A review

Mayerly Alexandra Oyuela Leguizamo*, Wilmar Darío Fernández Gómez, Martha Cecilia Gutiérrez Sarmiento

Faculty of Environmental and Natural Resources, Campus El Vivero, Natura Building, Universidad Distrital Francisco José de Caldas, Bogota, Colombia

HIGHLIGHTS

- Wetland degradation is linked to heavy metals contamination.
- Phytoremediation impacts the environment with the use of exotic species.
- Future research in phytoremediation must be focuses on native and endemic biodiversity.
- Discoveries of wild plants decontaminators, could conserving the nature's remnants in urban wetlands.

ARTICLE INFO

Article history:
Received 20 April 2016
Received in revised form
19 October 2016
Accepted 20 October 2016
Available online 4 November 2016

Handling Editor: T Cutright

Keywords:
Allochthonous plants
Autochthonous plants
Heavy metals
Herbaceous vegetation species
Phytotechnologies
Wetlands worldwide

ABSTRACT

Soil, air and water pollution caused by the mobility and solubility of heavy metals significantly damages the environment, human health, plants and animals. One common *in situ* method used for the decontamination of heavy metals is phytoremediation. This usually involves the use of exotic species. However, these species may exhibit invasive behavior, thereby, affect the environmental and ecological dynamics of the ecosystem into which they are introduced. This paper focuses on some native herbaceous plant species reported on the wetlands of Bogota, Colombia, with potential use in phytoremediation of heavy metals. To do that, the authors identified and searched a bibliography based on key words related to heavy metal decontamination. In addition, authors gathered and analyzed relevant information that allowed the comprehension of the phytoremediation process. This paper suggests the study of 41 native or endemic species regarding their behavior towards heavy metal contamination. From a survey of herbaceous plants reported in Bogota, native and endemic species that belong to predominant families in heavy metal accumulation processes were selected. Although found in Colombian's wetlands, these can also be found worldwide. Therefore, they are of great interest due to their global presence and their potential for use in phytoremediation. The current research about the development of phytoremediation focuses on the identification of new herbaceous species able to decontaminate substratum polluted with heavy metals to contribute with the investigation of the ecology and environment of the nature's remnants in urban wetland ecosystems.

Contents

1. Introduction 1231
2. Wetlands 1231
   2.1. Wetland importance 1231
   2.2. Wetland degradation 1232
   2.3. Heavy metals (HM) worldwide 1232

* Corresponding author. Diagonal 5 D Bis Number 46-10, 111611, Colombia.
E-mail addresses: mayerly.oyuela@gmail.com (M.A. Oyuela Leguizamo), wfernandez@udistrital.edu.co (W.D. Fernández Gómez), marthagutierrezs@gmail.com (M.C.G. Sarmiento).

http://dx.doi.org/10.1016/j.chemosphere.2016.10.075
0045-6535/

2.4. Allochthonous and autochthonous plants 1234
3. Phytoremediation of heavy metals (HM) 1234
3.1. Phytoremediation 1234
3.2. Phytoremediation approaches and technologies 1235
3.3. Advantages of phytoremediation 1236
3.4. Limitations of phytoremediation 1237
4. Methodology 1238
5. Analysis of research on heavy metals (HM) and native herbaceous vegetation species 1238
6. Analysis of research on heavy metals (HM) and herbaceous plants 1238
7. Results 1238
8. Discussion and conclusions 1242
Acknowledgments 1243
References 1244

1. Introduction

Wetlands are aquatic and semiaquatic ecosystems, subject to a permanent or periodic water logging with low depth; wetlands also have a strip of land that acts as a zone of environmental management and preservation. Wetlands are recognized for their ecologically significant contributions to biogeochemical cycles (Garg, 2015; Lewis, 1995) and as mega-diversity regions. Heavy metals (HM) are found in large quantities of wetlands around the globe, spread, among other factors, by anthropogenic interferences such as mining, industry, agriculture and construction activities (Cunningham et al., 1995; Switras, 1999). These metals eventually reach a point of toxicity—naturally and due to human activity—which causes them to exceed the limits of geochemical background limits. Despite the essential functions of many metals for living beings, others can be toxic in high concentrations (Henry, 2000). Therefore, the principal problem lies in the toxicity of HM elements and the bioaccumulative behavior that harms humans, plants and animals' health (Athar and Ahmad, 2002; Moalla et al., 1998; Paz et al., 2014) by and large, with deleterious effects on the environment like pollution, degradation of resources in quantity and quality, among others. See Sections 2.2, 2.3, 3 and 8 for more detailed information about all these topics.

In light of the aforementioned problems related to high metal concentrations, HM must be immobilized or physically removed. The literature widely reports wetlands, natural or constructed, in wastewater treatment, also known as sinks for metals and metalloids (Calijuri et al., 2011; Dunbabin and Bowmer, 1992; Kohler et al., 2004; Mays and Edwards, 2001; Yadav and Chandra, 2011). Phytoremediation, also named as phytocorrection and phytocleaning are promising technological approaches in the decontamination process (Chaney et al., 1997; Clemens, 2001; Cunningham et al., 1995; McGrath and Zhao, 2003; Pilon et al., 2000; Prasad, 2004; Rascio and Navari, 2011; Raskin, 1996; Raskin et al., 1997; Vassilev et al., 2004). See Section 3 for more information. Among the benefits described in Sections 2.1 and 3.3, about wetland importance and the advantages of phytoremediation, wetland water treatment is considered as a “socially acceptable” form of decontamination that does not have a negative impact on the landscape.

For phytoremediation and wetland restoration projects, it is necessary to determine the ecosystem-related and ecological functions of native or endemic species (Budelsky and Galatowitsch, 2004). Since, paradoxically, ecological restoration increases biological invasion. In many cases, this increase is direct via revegetation with exotic species. Other times, it is indirect, and caused by the creation of an artificial or altered environment in which exotic or invasive species can thrive (Castro et al., 2004). See Section 2.4.

This review focuses on native herbaceous plant species reported on wetlands, especially in terms of studies of plant behavior with the HM in growth substrate. Species were chosen based on global presence; that is, species taken from a local base of Bogota's plain wetlands, located approximately between 2540 and 2700 meters above sea level, in the urban zone (Alcaldía Mayor de Bogotá, 2006). The authors selected species present in other countries, for a greater geographical expansion in its study and application, and took into consideration that these species are fast growing, and easily propagated. Together, these characteristics are important for phytoremediation (Ghosh and Singh, 2005a) and enable heavy metal research. The present article discusses the importance and degradation of wetlands, the presence of HM worldwide, the implications of allochthonous and autochthonous vegetation in an ecosystem, and delves into the phytoremediation of HM. Finally, after a detailed summary of native herbaceous plants and the corresponding research, authors recommend vegetation species that belong to predominant families in heavy metal accumulation processes, that do not have a negative impact on the environment (because they are native or endemic). In other words, the plants recommended in this paper differ from those generally reviewed, i.e., exotic species that exhibit invasive behavior. In short, the aim of this paper is to go beyond the limited number of species previously investigated for phytoremediation, as well as the greatest limitations of the use of plants in decontamination processes (Ghosh and Singh, 2005b), described in Section 3.4, and contribute to the ecology and environment of the nature's remnants in urban wetland ecosystems.

2. Wetlands

2.1. Wetland importance

Wetlands ecosystems figure among the most productive on Earth thanks to their components (biotic and abiotic features), functions (which give rise to component interactions, nutrient cycle, surface and subterranean water interchange and surface and atmosphere interaction, among others) and properties (i.e., species diversity) (Barbier et al., 1997; Mitsch and Gosselink, 2015a). Functions, components and properties attributed to wetlands depend on their geographical distribution, environmental components (i.e., hydrospheric, geospheric) and interactions between them. All these goods or services are ecosystem services (Cortés and Estupiñán, 2016). The classification of ecosystem services is subject to analysis by a multiplicity of approaches, either at a purely anthropocentric or ecological level, but the implementation of any of the classifications depends on the ecosystem characteristics or the purpose of their application (Camacho and Ruiz, 2012). The Millennium Ecosystem Assessment divided ecosystem services into four types: supporting, provisioning, regulating and cultural (Millennium Ecosystem Assessment, 2005).

Wetlands ecosystems provide tangible and intangible ecosystem services. The intangible ecosystem services, such as supporting, with or without direct implications on human life, are the ecological roles in the hydrologic, biogeochemical cycles, maintaining biodiversity and biological pest control (Cortés and Estupiñán, 2016; Millennium Ecosystem Assessment, 2005). Likewise, regulating services, i.e. the regulation of climate, floods, disease, and water treatment. Finally, tangible and intangible cultural services, such as recreational, (i.e. pedagogical spaces for bird watching, sailing or scientific studies), aesthetic enjoyment, and spiritual fulfillment. In sum, they are socially and culturally important. An example: sphagnophile soils found on wetlands conserve vestiges of towns and roads with historical and archaeological significance (Barbier et al., 1997), in turn boosting regional tourism (Burton and Tiner, 2009). Tangible services of wetlands, a group from provisioning services, which are countable and consume ecosystems products, like food, medicines, fibers and water. In accordance with the aforementioned, wetlands are even referred to as "nature's kidneys" (Garg, 2015), given that they temporarily store water and reduce flooding damage, refill subterranean water supplies (i.e. wetlands connected to groundwater systems or aquifers). Wetlands also purify water, maintain or transform nutrients, sediments and toxic materials circulating in water, help control erosion and sediment transport (i.e. wetlands that occur along the shoreline of lakes, the banks of rivers and streams) and improve air quality (Lewis, 1995; Millennium Ecosystem Assessment, 2005).

As far as environmental benefits are concerned, wetlands are viewed as "biological supermarkets" due to the complex food web and biological diversity they support as a wildlife habitat (Mitsch and Gosselink, 2015a). Furthermore, they sustain and nurture human life through a set of scenarios and processes present in the ecosystems (Daily, 2000). Likewise, it has been suggested that wetlands will play a key role in achieving the goals set out in the UN's Millennium Development Goals (MDG) and the Sustainable Development Goals (SDGs), among others (Barbier et al., 1997; Ten Brink et al., 2013). Soil is one of the most important components in wetlands. In accordance with Keesstra et al. (2016), that mention the soil as a decisive component in an ecosystem management, when taking into account the relationship between the soil and one or more of the SDGs: food security; human health; land management, including land restoration; water security; climate change; and biodiversity preservation. Likewise, Keesstra et al. (2016) discuss how soils can contribute to the development of the SDGs.

2.2. Wetland degradation

Drivers of change, direct or indirect, may affect the ecosystem degradation. Indirect drivers of change include demographics, economics (e.g. globalization, trade, market, policy framework), society and politics (e.g. government, institutional and legal frameworks), science and technology, culture and religion (e.g. beliefs, consumption choices) (Abínzano, 1998). Changes in these drivers can impact direct drivers of change, such as changes in local land use, species introduction or removal, technology adaptation and use, external inputs (e.g. fertilizer use, pest control, and irrigation), harvest and resource consumption, climate change, natural, physical, and biological drivers (e.g. evolution, volcanoes). These factors may lead to changes of ecosystems, as a result in the services offered, including provisioning (e.g. food, water, fiber, and fuel), regulation (e.g. climate regulation, water, and disease), culture (e.g. spiritual, aesthetic, recreation, and education) and support (e.g. primary production, and soil formation). In so doing, human well-being and biodiversity are diminished and spatio-temporal disturbances are brought about (Lewis, 1995; Millennium Ecosystem Assessment, 2005). Moreover, the loss of wetlands reduces an ecosystem's ability to filter and decompose waste (Lewis, 1995; Millennium Ecosystem Assessment, 2005).

From a market-based perspective—with the concomitant emphasis on economic criteria such as environmental goods and services, natural resources—one of the main causes of degradation and excessive conversion of wetland resources is related to the frequency in which wetland ecological contributions are not considered in economic terms. Measuring and comparing the numerous benefits provided by wetlands commercially, then, facilitates taking wetlands into account as part of sociopolitical decision-making processes. Expressing these ecosystems as providers of values in perpetuity (Mitsch and Gosselink, 2000). Additionally, this commercial consideration of wetlands establishes the foundation for rational use, management and preservation of wetland resources (Barbier et al., 1997). This concurs with "biodiversity prospecting," that offers a several opportunities to save as much as possible of the variety of ecosystems, focus on a commercial viewpoint (Prasad, 2003). From a different perspective, the holistic approach to resource management eschews economic value in favor of an ethical and cultural conception (Grayman et al., 2012). This approach goes against viewing ecosystems through a market-based perspective, where the focal point is its use and usufruct as an exploitable good or service, resulting in the inevitably of quantitative and qualitative deterioration of ecosystems (Lozano, 2011).

Colombia has 30.781.149 ha and more than eighty-eight types of wetlands between marine-coastal, inland and artificial (Flóres et al., 2015; Ricaurte et al., 2015). In Colombia, the market-based view has taken hold, but that has not prevented the massive underappreciation of the economic value of natural resources. And this underappreciation is even less understandable when accounting for the existence of clear-cut Methodologies for Valuing Natural and Environmental Goods and Services (MAVDT, 2003). Therefore, a large part of the economy has been building around unsustainable extraction activities, such as mining, fishing, cattle, agriculture, etc., without taking steps towards realizing the biodiversity as a sustainable potential of the country's economy. This underappreciation becomes more significant if readers keep in mind that Colombia has the second most biological diversity in the world, product of evolution, Tertiary and Quaternary speciation, concentrated in the Amazon and the Andes Region (MMA, 2002).

In Bogota, uninformed decisions regarding land use, urbanization and haphazard expansion have negatively impacted wetlands, of 50,000 ha, only eight hundred (perhaps less) remain. The result is fragmented ecosystems, obliterated connectivity and modified hydric systems. In sum, Bogota has a serious wastewater dumping problem, unsuitable waste regulation, difficult access to better income opportunities, depredation of local flora and fauna and adaptation of land for farming, among other issues (Alcaldía Mayor de Bogotá, 2006). Mitsch and Gosselink (2015b) indicate the extent of the world's wetlands are about 5%–8% of the land surface of Earth. The documentation of loss of wetlands in various locations in the world estimates over (27%) Asia, (6%) South America, (2%) Africa, (90%) New Zealand, (60%) China and (60–80%) Europe (Mitsch and Gosselink, 2015b).

2.3. Heavy metals (HM) worldwide

From a chemical perspective, HM are transition metals with an atomic mass greater than 0,002 Kg and a specific weight greater than 5 N/m^3. Biologically speaking, "heavy" describes a series of metals, in some cases metalloids, that even in low concentrations can be toxic for plants and animals (Rascio and Navari, 2011). The origin of heavy metal contamination consists of activities that contribute to soil, air and water contamination. Common examples include industrial activities (dumping and inadequate waste disposal), mining for waste processing (Pfeifer et al., 2000; Tanhan et al., 2007), chemical industries, metal processing industries (Athar and Ahmad, 2002), traffic jams, construction materials and agriculture (inadequate agricultural practices), among others (Barba and Edith, 2002; Diez, 2008). Around the world, there is a strong presence of chemical elements in sediments in different types of natural wetlands, considered as sink for metals, in the anoxic zone, containing very high concentrations of this HM in reduced state. Therefore, wetlands may be sources as well as sinks for contaminants, as metals, persistent materials (Das and Maiti, 2008). Schaller et al. (2013) refer to many articles about the presence of metals/metalloids, such as Ni, Zn, U, Pb, Cu and As, in sediments of different types of wetlands around the world, in China, Canada, Germany, Zambia, USA, Spain, UK, India, Paraguay, Poland, Vietnam, Brazil, Malaysia, Egypt, Czech Republic, Taiwan, Pakistan, Italy, Australia and Belgium. Some examples of these sediments are paddy fields, wetlands receiving waste waters, river sediments, marsh sediments, river sediments, free water surface, wetlands near copper smelter, wetland effected by mine waters, urban lakes, lake and wetland sediments, anthropogenic lake sediments, among others. In Bogota-Colombia's natural wetlands, the metals present include Al, Cu, Zn, Cn, (found in all wetlands), Cd, Cr, Ni and Pb. The presence of these HM can be attributed to flawed connections in the industrial sector and a flawed urban sewage system (Alcaldía Mayor de Bogotá D.C, 2010).

Ultramafic rocks describe, among others, piroxenite, dunite and igneous (such as peridotites) or metamorphics (such as serpentinites) (Sánchez, 2010), that are less than 45% silica (SiO_2) and have high concentrations of Mg and Fe. Often, these soils present, in addition, high levels of Cr, Co and concentrations of Ni, raised Mg/Ca quotient and low levels of P, K and Ca (Ghaderian et al., 2007; Robinson et al., 1997). Consequently, soils that evolve on these types of ultramafic rocks are slightly acidic, have low nutrient contents (K, N, P), exhibit more erosion, contain high concentrations of HM (Fe, Co, Cr, Hg, Ni) and have a low Ca/Mg ratio. These conditions diminish the vegetation's productivity and represent a serious disadvantage for conditions for life of all organisms present (Brady et al., 2005; Garnier et al., 2006; Maleri et al., 2007).

Hyperaccumulation, per the definition provided by Jaffré et al. (1976), describes plants that grow on serpentine soils, species able to concentrate up to 1000 mg/kg of Ni. The high concentration of HM in the vegetative organs of most plants reaches toxicity for Ni around 10–15 mg/kg. Consequently, hyperaccumulators can withstand up to 100 times higher metal concentrations than typical plants without reported accumulation processes (Boyd et al., 2008). Ni has been shown to reach its highest concentration in a plant between 100 and 1000 mg/kg of dry weight (accumulators) or more than 1000 mg/kg (hyperaccumulators) (Rascio and Navari, 2011). This confirms results of studies on serpentine substrates in Central America, Greater Antilles and South America, where most hyperaccumulators are focused on nickel(Reeves et al., 1999). Baker et al. (2000) indicates global distribution of nickel hyperaccumulation, geographic areas and number of hyperaccumulators: New Caledonia (50), Australia (5), South Europe/Asia Minor (90), South West of Asia (11), Cuba (128), Dominican Republic (1), U.S. (Pacific N.W. and California) (5), Zimbabwe (5), South Africa (Transvaal) (4), Japan (Hokkaido) (1), Brazil (Goiás) (11), Canada (Newfoundland) (4). Plants that can grow and develop in soils with high concentrations of HM are part of a specialized flora (Reeves et al., 1999). Those plants colonize soils characterized as lightly acidic, from soils in serpentinized or ultramafic areas, rich in Ni, and calamine (a mineral with high levels of Zn and Cd), whether naturally occurring or the product of anthropic- or mining-related contamination.

Heavy metal hyperaccumulators species occur in metal-rich soils, temperate and tropical zones (South America, North America, Europe and New Caledonia) (Baker and Brooks, 1989). These plants are chosen based on their high tolerance. Serpentine tolerance can be explained as part of three established physiological and evolutionary mechanisms: tolerance to a low Ca/Mg ratio, avoidance of Mg toxicity or high Mg requirement. The vast majority of known hyperaccumulator species belong to vegetation communities, characteristic of soils in serpentinized areas. Not all accumulative species concentrate metal equally. Small to large concentrations are tolerated in different plants. The former are considered "accumulators," while the latter are considered "hyperaccumulators" (Jaffré et al., 1976). See Section 3.2.

Metalliferous soils have high concentration of metals elements, of natural origin, as well as sites polluted with HM due to anthropogenic activities. These soils are geographically distributed in New Caledonia (Baker and Brooks, 1989), China (Wei et al., 2009), Saudi Arabia, Sudan, Papua New Guinea's (Bertram et al., 2011), The Democratic Republic of Congo (50% of the world's cobalt reserves), Chile (30% of the world's copper reserves), Ukraine and South Africa (world's manganese reserves) (USGS, 2010). Some others are located in Switzerland, Eastern France, Northern Italy (Pfeifer et al., 2000) and in Tanzania's North Mara Gold Mine (Mataba et al., 2016). Serpentine, ultramafic, and calamine soils are sources characterized by high concentrations of HM of natural origin (Maleri et al., 2007). Some authors refer areas with these characteristics in Southern Europe, Asia Minor, Portugal, West of Iraq, Turkey, Iran, northwestern North America, Zimbabwean, Cuba, New Caledonia, Brazil (Baker and Brooks, 1989; Ghaderian et al., 2007; Pollard et al., 2014). Also in Australia, Canada, Russia, Congo and Zambia, (USGS, 2010), Puerto Rico, Brazil, Costa Rica, California, the Philippines, Indonesia (Reeves et al., 2007; Van der Ent et al., 2013b), Albania (Shallari et al., 1998). Ultramafic rocks, places like Cuba and New Caledonia, where serpentine-endemic hyperaccumulators can be concentrated in one locality (Baker et al., 2000). Phytomining, the use of plants in mining, was the result of the discovery of HM, accompanied by considering their use in commercial processes, as well as the increase of HM in plant growth substrate (Chaney et al., 1997; Prasad, 2004; Sheoran et al., 2009; Vassilev et al., 2004).

In worldwide is evaluated actual and potential metal environmental risks, in order to decide the necessity/urgency of rehabilitation, Mench et al. (2000) describes some cases as follows: in many parts of Europe and North America, natural vegetation completely disappears due to HM contamination. In France, 2000 of the most polluted sites were rehabilitated. In northeast Belgium, the surface soil of more than 280 km2 contains the highest background values levels of HM. In the U.K., soils were contaminated by the industrial legacy from base metal mining, metal refining and smelting. In India, there was a contamination, product of industrial effluent and sewage sludge on agricultural land (Athar and Ahmad, 2002); also, in this country, there are more than 3000 tanning industries (Arora et al., 2005). Rail junction areas in Poland are contaminated with numerous compounds and chemical substances (Malawska and Wiołkomirski, 2001). Sultan Marsh, one of the most important wetlands in Turkey, Middle East and Europe, is mainly polluted with Pb, Cd and Cr (Aksoy et al., 2005). Distribution and phase association of Cd, Cr, Cu, Fe, Mn, Ni, Pb, Zn in Lake Nasser (Aswan, Egypt), were investigated (Moalla et al., 1998). In Spain, the potential use of spontaneous vegetation (*Festuca rubra* and *Juncus* sp.) was evaluated for phytoremediation and/or phytostabilization in the former serpentinite quarry of Penas Albas (Moeche, Galicia, NW Spain). The activity on this quarry left behind a large amount of waste material scattered over the surrounding area (Lago et al., 2015).

2.4. Allochthonous and autochthonous plants

Allochthonous plants are not native to a phytogeographic region; in contrast, autochthonous plants are native to a territory (Elorza et al., 2001; Vigo, 1976). The terminology for the former has been the object of various revisions (Kasperek, 2008; Pyšek, 1995; Schroeder, 1968), in which, allochthonous plants have also been called non-native plants or exotic plants (Elorza et al., 2001). The term even includes plants that are cultivated (intentionally introduced by humans to satisfy certain demands; they have specific care requirements), non-cultivated (arrived with cultivated species without knowledge of their reproduction or if they constitute a population), sub-spontaneous plants (also known as escaped cultivated plants that are found near cultivated areas) (Vigo, 1976). Some others revisions incorporate naturalized plants (allochthonous plants that establish themselves in the environment, without human assistance; these plants do not, over time, become native members of the local plant community) (Ojasti, 2001). Established plants (those exotic species that do not exhibit invasive behavior, but can reproduce and produce a viable population) (McNeely et al., 2001). Other revisions include adventitious (which are established spontaneously in natural or artificial habitats, although they are not able to effectively prosper and are at risk of disappearing due to changes in the factors that favored their introduction) (Vigo, 1976) and invasive plants (Pyšek, 1995). The work of the Swiss botanist Albert Thellung (1881–1928), a pioneer in invasion research, contributed to the understanding of this field (Kowarik and Pyšek, 2012). Invasive species are introduced by humans in areas out of their natural distribution (Elorza et al., 2001); in some cases, they are used for biological control or in research (Ojasti, 2001; Villaronga, 2003). Biological control only represent 6.8% of cases of exotic species introduced to continental waters (Welcomme, 1988).

To clarify, biological invasion is a process in which plants disperse and establish themselves, mainly affecting the ecosystem and local species in ways that are generally incalculable and irreversible, namely exotic species can impact native species abundance (Tilman and Lehman, 2001). Moreover, they generate massive economic losses; vast sums are spent on controlling these invasive species. For example, the South African government spends 40 million dollars annually to control three invasive plant species (Kowarik, 2008; Villaronga, 2003). On top of that, these species present sanitary, i.e. vectors for disease (Villaronga, 2003), and ecological issues, i.e. the invasion of these species is second biggest factor in the species extinction worldwide (Lowe et al., 2000), followed by habitat destruction. Damage at an ecological level is due to the competition between invasive and autochthonous species, acting as plagues and pathogens, or disperse allergenic and infectious agents in native plants, which may affect biodiversity and, in turn, *in situ* conservation of endemic vegetation (Elorza et al., 2001; UICN, 2000). These issues unequivocally demonstrate the global problem presented by invasive species. This problem requires international action and cooperation to avoid the most dire threats and further reduction of biological diversity (Clavero and García, 2005; Lodge, 1993).

3. Phytoremediation of heavy metals (HM)

3.1. Phytoremediation

Heavy metal decontamination relies on chemical, physical and biological techniques. These methods fall into one of two categories: *ex situ* and *in situ*. In the former, the contaminated substrate is removed, treated and returned; conventional *ex situ* techniques include excavation (followed by burial of contaminated substrate at a disposal site), detoxification and/or destruction of contaminants by physical or chemical processes. In addition, these methods may subject the contaminants to stabilization, solidification, immobilization, incineration or destruction (Macnair et al., 1999; Sadowsky, 1999). The immobilization of inorganic contaminants can be used to recover HM from contaminated soils (Bhargava et al., 2012), by way of increasing pH to decrease the solubility of HM such as Cd, Cu, Ni, Zn in the soil. The majority of conventional technologies for remediation carry a large price tag and run the risk of further damaging the affected area. In their deployment, then, the problem is not resolved but rather, deferred. As a result, ex situ treatments do not represent a viable option for addressing heavy metal decontamination (Arshad et al., 2008; Macnair et al., 1999; Sadowsky, 1999); in fact, they affect biological properties in the soils where they are used (Arshad et al., 2008). *In situ* methods, on the other hand, do not entail contaminated soil excavation. Instead, they employ remediation technologies that destroy or transform contaminants in the substrate (Ghosh and Singh, 2005b; Prasad and Freitas, 2003; Sadowsky, 1999; Susarla et al., 2002). One of these *in situ* methods is the use of plants for bioremediation, known as phytoremediation (Sadowsky, 1999). There are many advantages of these methods, compared to *ex situ* methods. Key benefits include lower costs and reduced impact on the ecosystem (Tanhan et al., 2007).

In 1983, metal-accumulating plants were first used to remove HM and other compounds. Although using metal-accumulating plants for soil decontamination is an innovative idea, it must be understood as a variation of a technique already implemented for over 300 years in wastewater decontamination (Henry, 2000). The generic term "phytoremediation" consists of a Greek prefix, "*phyto*," which means plant, and a Latin root, "*remedium*", which means a cure or remedy (Prasad, 2004); naturally, then, phytoremediation "cures" the soil by means of plants. This technology can be applied to organic and inorganic contaminants present in soil (solid substrate), water (liquid substrate), air (Prasad, 2003), sludge, sediments and groundwater (Prasad, 2004; Ramamurthy and Memarian, 2012). The use of physico-chemical approaches for soil recovery leave the ground unusable for the plant growth because this decontamination process eliminates the biological activities of the soil's "edofauna" (Ali et al., 2013).

Phytoremediation is a form of bioremediation that detoxifies by means of biological processes. This has been utilized since the eighteenth century (Barceló and Poschenrieder, 2003). It specifically consists of using plants to remove contaminants from the environment and render them harmless; it is also known as phytocleaning or phytocorrection (Chaney et al., 1997; Cunningham et al., 1995; McGrath and Zhao, 2003; Pilon et al., 2000; Raskin, 1996; Sarma, 2011). Phytoremediation entails the use of plants to sequester, bioaccumulate and transform hazardous substances in the environment in order to control contamination (Rodríguez, 2003).

One example of this technology is the study of aquatic macrophytes to absorb HM (Celis et al., 2005; Robinson et al., 2006). In phytoremediation, *in situ* or *ex situ* plants reduce the concentration of various compounds via their biochemical processes (Delgadillo et al., 2011). Furthermore, this technique employs associated micro-biotics, soil improvement and agronomic techniques to remove or retain contaminants or neutralize their effects (Cunningham et al., 1995). Phytoremediation is currently used for the decontamination of soils that display increased levels of HM. These levels exceed the soil's naturally occurring chemical elements, which is known as natural or geochemical background or geochemical baselines; this geochemical background is the average local content of chemical elements in rock soil, natural waters, surface atmosphere and plants, generally in air, water, soil and sediments. Different degrees of metal accumulation have been reported, from slightly higher than the geochemical background to extreme cases in which the metal exceeds 2% of the plant's dry material (Kidd et al., 2007). In sum, phytoremediation represents a promising solution to the decontamination of a number of contaminants across a variety of sites.

3.2. Phytoremediation approaches and technologies

Phytoremediation or phytocorrection encompasses a group of techniques and mechanisms premised on the use of vegetation species and their related microorganisms to remove, uptake, immobilize or transform soil or water contaminants (Barceló and Poschenrieder, 2003; Cunningham et al., 1995; Ghosh and Singh, 2005b; Pilon et al., 2000; Prasad, 2003). Contaminants may be immobilized, stabilized or degraded in the rhizosphere, sequestered or degraded within the plant or volatilized (Cunningham et al., 1995; Horne, 2000).

Within the field of phytoremediation of soils contaminated by HM, there are numerous sub-fields: phytostabilization or phytoimmobilization, phytovolatilization, phytodegradation, phytoextraction (principally soil, sediments and sludge's applications), rhizofiltration, rhizodegradation, hydraulic barriers, hydraulic control, vegetative caps, constructed wetlands (principally water applications), and phytorestauration (Adams et al., 2000; Barceló and Poschenrieder, 2003; Prasad, 2004, 2003). Vassilev et al. (2004) summarize information concerning development, achievements, and research needs of metal phytoremediation concept, technological, and economical aspects, including description of phytoremediation subfields, and focuses on phytoextraction and phytovolatilization.

The following paragraphs describe some of the phytoremediation technologies.

Phytostabilization: (soil, sediment and sludge's media) is used to reduce or eliminated the mobility of toxic elements from the contaminated soil to the environment, thus, they are stabilized in the substrate or roots. These subfield transform soil metals to less toxic forms, without removing the metal from soil (Adams et al., 2000; Chaney et al., 1997; Cunningham and Berti, 2000; Prasad, 2004).

Phytovolatilization: (groundwater, soil, sediment and sludge's media) is the extraction from media and transpiration of a contaminant or a modified form, by plants (Adams et al., 2000; Prasad, 2004). Terry and Bañuelos (2000) described investigations on phytovolatilization of heavy metals like As, Hg, and Se.

Phytodegradation: (soil, sediments, sludge's media) refers to the destruction and breakdown of the contaminant (Adams et al., 2000).

Rhizofiltration: (groundwater, surface water media), also known as phytotransformation, is the use of plants' roots to remove contaminants from flowing water (Adams et al., 2000; Chaney et al., 1997).

Hydraulic control: (groundwater, surface water media), is the use of plant uptake and consumption, to remove, contain or control the migration of contaminants (Adams et al., 2000).

Phytoextraction: (soil, sediments and sludge's media), also known as phytoaccumulation, uses the plants' ability to absorb and remove contaminants from the soil and “uptake” them into their leaves and stalks. Next, the plant parts storing contaminants are removed and destroyed or recycled; in either scenario, the metal is extracted from the soil (Cunningham et al., 1995; Vassilev et al., 2004). The first step in applying this method is the selection of plant species best suited for: a) the removal of the metals to be addressed and b) site characteristics. Once the plant is fully grown, it is cut and incinerated before finally being transported, in the form of ashes, to a disposal site (Delgadillo et al., 2011).

It is worth pointing out that some plants that survive, grow and reproduce naturally in metalliferous soils are restricted to that type of soils (obligate metallophytes), whereas others do all that in normal soils or occasionally in metalliferous soils (facultative hyperaccumulators) (Pollard et al., 2014) See Section 2.3. Plants growing on metalliferous soils are classified as 1) Excluders, in which concentrations in the sprouts are tolerated, until reaching a critical value. Likewise, they prevent the absorption and/or translocation of HM in the cells in the roots towards the leaves, and maintain constant low concentrations over a long period in the aerial biomass, regardless of metal concentration in the soil (Ghosh and Singh, 2005b; Lasat, 2002; Srivastava and Bhargava, 2015; Van der Ent et al., 2013a). Excluders can be used for soil stabilization and to avoid greater propagation of the contamination through erosive processes (Lasat, 2002). 2) Accumulators allow the plant to absorb and uptake metals—without toxic effects—in their aerial biomass. Ranges for accumulation run from low to high concentrations in the soil. These plants, upon preventing metals from entering their roots, bioaccumulate high concentrations of metals. 3) Indicators, an intermediate plant between excluders and accumulators, have internal concentrations that reflect external concentrations found in the growth substrate (Ghosh and Singh, 2005b).

The majority of these species, that tolerate toxic concentrations of HM, exhibit the behavior of “excluders,” which employ tolerance and even hypertolerance strategies to help prevent the metal from entering the plant's aerial parts. These species retain and detoxify most of the HM in the root tissues with minimized translocation to the leaves and stalks, which are sensitive to the effects of phytotoxicity. Some plants reach higher exclusion values; this is called hyperexclusion. Hyperexclusion does not present metal translocation to the plant's aerial parts and entails minimal impact (Proctor, 1999). Yet, there are hypertolerant species that behave quite differently: some have uptake and distribution of the heavy metal throughout the entire plant. These are known as “hyperaccumulators” (Rascio and Navari, 2011).

To develop phytoextraction mechanisms, there are numerous plants endowed with the natural ability to tolerate some toxic compounds, for they are perfectly adapted to their habitat's specific environmental conditions. Hyperaccumulators concentrate more than 1000 mg/kg of As, Co, Cr, Cu, Ni, Pb, Sb and Se; more than 10,000 mg/kg of Mn or Zn; more than 100 mg/kg of Cd in their aerial biomass (Ghosh and Singh, 2005b; Jaffré et al., 1976; Wei et al., 2008), until exceeding concentrations of 2% of their dry material (Kidd et al., 2007).

Bioaccumulation refers to the mechanism for absorbing chemical compounds performed by an organism within the biotic (food resource) or abiotic (environment) medium. A quantitative form of expressing bioconcentration is via the bioconcentration factor (BCF), which is the ratio of the compound's concentration in the environment or in the growth substrate to that found in the living element. In the case of heavy metal phytoremediation, this factor is the ratio of the metal in the plant biomass, or tissue (mg/Kg dry weight) to the initial metal concentration in the soil or solution (Kg/mL) where the plant grows—this ratio is a determining factor when it comes to identifying a plant's metal removal efficiency. Vegetation species are considered to exhibit removal potential when they have a ratio greater than 1 (Sun et al., 2009a).

To ensure the plant's roots uptake metals in the soil, the metals from the soil-bound metal must first be solubilized; this can be achieved if the plant secretes metal-chelating molecules or metallic reducers. Once solubilized, the metal ions can enter the roots via extracellular or intracellular pathways, where the latter generally require the presence of an ion channel or a metal transport protein in the root cell's plasma membrane. It has been suggested that some of these channels may be non-specific, meaning that various metals may be used. After metal ions reach the root, they can be stored in the root's vacuoles, often in chelated form, or they can be transported to the plant's aerial parts (Prasad, 2003). The aforementioned process outlines how plants in growth conditions in native soils concentrate HM in their aerial organs without undergoing phytotoxicity (Baker and Brooks, 1989; Sarma, 2011). Hyperaccumulation is a phenomenon that cannot be defined at a laboratory level; it is necessary to study it in a natural environment (Van der Ent et al., 2013a).

Some researchers have hypothesized that plant hyperaccumulation, understood as an evolutionary process, can be described in five categories: i) a tolerance mechanism; ii) an unnoticed absorbance; iii) an allelopathic interference mechanism; iv) a drought-resistance mechanism; v) a defense against herbivores/pathogens (Boyd and Martens, 1998). Of these five hypotheses, the fifth one has garnered the most attention (Boyd, 2007; Pollard and Baker, 1997; Poschenrieder et al., 2006), while the first four have been scantly researched. Therefore, they cannot be discarded outright, though hypothesis iii (allelopathic interference) seems to merit rejection based on results from a study on the Ni hyperaccumulator *Alyssum murale* (Zhang, Angle et al., 2007).

Research into hyperaccumulators identified more than 500 hyperaccumulator species (Van der Ent et al., 2013a), as shown on Table 1, that belong to a total of 45 families. The families *Asteraceae, Brassicaceae, Caryophyllaceae, Cyperaceae, Cunouniaceae, Fabaceae, Flacourtiaceae, Lamiaceae, Poaceae, Violaceae and Euphobiaceae* accounted for the majority of hyperaccumulator species and, subsequently, were the predominant families in metal accumulation processes (Baker and Brooks, 1989; Prasad and Freitas, 2003). From hyperaccumulators, 79% are plant species found in Central and South America, mainly distributed throughout Cuba, 18% in Puerto Rico and Hispaniola, with values frequently between 1000 and 10000 mg/kg. Families that stand out include: *Euphobiaceae*, *Asteraceae*, *Buxaceae*, *Acanthaceae*, *Ochnaceae*, *Clusiaceae*, *Tiliaceae*, *Turmeraceae*, *Boraginaceae*, *Scrophulariaceae* and *Violaceae* (Marrero et al., 2012).

3.3. Advantages of phytoremediation

It is safe to say that phytoremediation is the best alternative for the removal, decontamination and primary isolation of toxic substances in soils, for it does not destroy soil structure or fertility (Ghosh and Singh, 2005b). This method also boasts positive cost-benefit and engineering-economy relations as an environmental friendly technology (Raskin et al., 1997; Bennicelli et al., 2004). Horne (2000) specified the historical background of wetlands and traditional remediation techniques, similarities and differences between conventional bioremediation, phytoremediation, and wetlands phytoremediation. Baker et al. (2000) reported successful cases of metal removal by phytoremediation wetlands.

The following are a few of the advantages of some phytoremediation technologies:

Phytostabilization: it is not necessary soil removal, ecosystems restoration is enhanced by the revegetation, does not required the disposal of hazardous materials, reduces bioavailability for entry in food chains, prevents erosion and improves visual amenity in a derelict site (Adams et al., 2000; Cunningham and Berti, 2000; Prasad, 2004).

Phytovolatilization: pollutants could be transformed into less-toxic forms; the contaminants forms released into the atmosphere might be subject to a natural degradation process (Adams et al., 2000; Prasad, 2004).

Phytodegradation: potentially could occur in soils where biodegradation cannot. This phytoremediation technology has the advantage that enzymes produced by a plant can occur in an environment free of microorganisms. Also, plants are able to grow in sterile soils, and in those with toxic levels for microorganisms (Adams et al., 2000).

Rhizofiltration: can be used in terrestrial or aquatic plants. Terrestrial plants, through a platform, can accumulate more contaminants than aquatic species. Rhizofiltration is applicable to *in situ* and *ex situ* systems, the latter can be placed anywhere, it is not necessary to be at the original location of the contamination (Adams et al., 2000).

Hydraulic control: avoids migration of leachate towards groundwater or receiving waters (Prasad, 2004). It is not necessary to install an engineered system. The roots are in contact with a much greater volume of soil than a pumping well (Adams et al., 2000).

Phytoextraction: This technology may be used to improve livestock nutrition (Robinson et al., 2015), that is how plant biomass can be a resource of the extracted contaminant (Prasad, 2004). Species that hyperaccumulate metals have a massive biotechnological potential (Hassan and Aarts, 2011; Lone et al., 2008; Mengoni et al., 2010). Plants capable of accumulating HM are grown on contaminated sites, and the aerial biomass rich in metals is collected when mature. As a result, part of the soil contaminant is removed. The success of phytoextraction as an environmental “purifier” hinges on factors such as availability of the metal for absorption, and the plant's capacity to absorb and uptake metals in its aerial parts. For cost-related purposes, the harvested biomass is generally incinerated or composted, and, less frequently, recycled for reuse (Prasad, 2003). Phytoextraction is one of the most

Table 1
Number of metal hyperaccumulator plants.

Description	References
145 hyperaccumulators of Ni, distributed in 22 families	(Baker and Brooks, 1989)
400 angiosperm hyperaccumulator species	(Baker and Brooks, 1989; Baker et al., 2000; Prasad and Freitas, 2003)
415 species from 45 botanical families	(Boyd et al., 2008)
317 species hyperaccumulate Ni, (28) Co, (37) Cu, (14) Pb, (1) Cd, (11) Zn and (9) Mn	(Baker et al., 2000)
320 species hyperaccumulate Ni, (30) Co, (34) Cu, (20) Se, (14) Pb, (1) Cd, (11) Zn and (10) Mn	(Ghosh and Singh, 2005b)
450 species of angiosperms with accumulative behavior of: As, Cd, Co, Cu, Mn, Ni, Pb, Sb, Se, Tl, Zn	(Rascio and Navari, 2011)
320 species hyperaccumulate Ni, (34) Cu, (34) Co, (20) Se, (18) Zn, (14) Pb, (9) Mn and (4) Cd	(Bhargava et al., 2012)
More than 450 species of known hyperaccumulators hyperaccumulate Ni, generally from ultramafic or serpentine soils	(Chiarucci and Baker, 2007)
500 species that hyperaccumulate at least one heavy metal. (450) Ni, (32) Cu, (30) Co, (20) Se, (14) Pb, (12) Zn, (12) Mn, (5) As, (2) Cd.	(Van der Ent et al., 2013a)

promising approaches in terms of commercialization. Patents for commercial phytoextraction were issued in Japan in 1980 (Cunningham et al., 1995).

3.4. Limitations of phytoremediation

Phytoremediation also comes with its own obstacles: this process is limited by the depth reached by the roots. It is restricted to surface area soil available for root uptake, surface and subterranean water. Moreover, phytoremediation is a slow method and one affected by the contaminant's bioavailability, which often means it ought to be employed in conjunction with other approaches so as to optimize performance (Monferrán and Wunderlin, 2013). As an example of this combination, there is the potential use of biochar and phytoremediation technologies on heavy metal polluted soils, with the goal that biochar can reduce the bioavailability and leachability of heavy metals in the soil (Paz et al., 2014). In spite of phytoremediation is an environmental friendly technology, it has its own impact, i.e., the decrease of resources, plus the risk for human health, as a result of exposure conditions, and viability of businesses (Cundy et al., 2013).

Here, attention shifts toward the reported costs associated with heavy metal decontamination. In Cunningham et al. (1995), it is reported that, on average, *in situ* remediation cost between 10 and 100 dollars for each cubic meter (m^3) of soil for volatile contaminants or water soluble contaminants; this increased to 60–300 dollars per cubic meter (m^3) for those deposited in landfills or undergoing low-temperature treatments. This figure jumped to a range of 200–700 dollars per cubic meter (m^3) for contaminants that require disposal in special dumps or high-temperature treatments. For the sake of comparison, contaminated soil incineration costs roughly 100 dollars per cubic meter (m^3). Some materials require specialized treatments that are in the neighborhood of thousands of dollars (1000–3000) per m^3 of soil. Thus, conventional treatments can surpass 1.6 million dollars, while phytoremediation can be up to 1000 times cheaper. Therefore, it comes as no surprise that in the United States, 16–29 million dollars were spent on phytoremediation from 1998 to 2000, whereas this number spiked to 214–370 million in 2005 (Barceló and Poschenrieder, 2003). To reiterate, phytoremediation is much more cost-effective than *ex situ* methods; Terry and Bañuelos (2000) have more information on specific cases. Adams et al. (2000) indicate phytoextraction, rhizofiltration, phytostabilization, hydraulic control and vegetative cover costs. For more information, see Sections 3.1 and 3.2. In spite of the benefit mentioned above, in Sections 3.1, 3.2 and 3.3, about the low-cost of phytoremediation technology, Robinson et al. (2007) mentioned that it does not include the opportunity cost of not using the land for a long time, as it happens in urban environments.

The following are few of the limitations of some phytoremediation technologies:

Phytostabilization: It will be irrelevant if its focal point is the prevention of erosion and the improvement of visual amenity of a contaminated site. The land value is reduced due to the presence of toxic metals in the plants on sites to be used as a resource for biofuels or timber (Cunningham and Berti, 2000). Another limitation is that vegetation and soil require long-term permanence in order to prevent discharge of the contaminants and leaching. Plants require extensive fertilization or the use of amendments. Phytostabilization might be considered as an interim measure. Soil and plant characteristics must be monitored to prevent the increase of metal solubility and leaching (Adams et al., 2000).

Phytovolatilization: There is evidence of low levels in plant tissues of the contaminant forms, which might have been released into the atmosphere or could accumulate in vegetation and be passed on in latter products as fruits (Adams et al., 2000; Prasad, 2004).

Phytodegradation: contaminants destruction could be difficult to confirm, thus the presence or identity of metabolites may form degradation or intermediate products (Adams et al., 2000).

Rhizofiltration: needs continuous adjustment of pH influent; interactions of species in the influent have to be understood; an engineered system is required to control influent concentration and flow rate; are required periodic harvesting and plant disposal (Adams et al., 2000).

Phytoextraction success using hyperaccumulators has been hindered, in many cases, by the limited biomass of these species. Species with this limitation include that most heavily studied, *T. caerulescens,* which is probably feasible in moderately contaminated soils (McGrath and Zhao, 2003). However, there are exceptions: hyperaccumulators with large biomasses include the Ni hyperaccumulator *Berkheya coddii* (Robinson et al., 1997). Therefore, there doesn't seem to be any intrinsic reason for low biomass in hyperaccumulators, i.e. there is no reason for all hyperaccumulator plants and/or plants capable of tolerating high metal concentrations to be competitively inferior or slow growth (Macnair et al., 1999). In any case, more biomass production by hyperaccumulator plants could significantly minimize the time needed for phytoextraction (McGrath and Zhao, 2003). Robinson et al. (2015) explain the relation between soil conditions and the bioaccumulation coefficient, in which a small minority of plants with roots forage contaminant hotspots, would remove greater rate of metals. It is also mentioned that HM uptake is in function of the soil concentration, therefore, uptake and the metal concentration in soil decrease at the same time. That would represent a significantly efficiency into the field and laboratory experiments.

Hyperaccumulation is indeed a rare phenomenon, and documented cases are naturally low, given that the phenomenon occurs in less than 0.2% of the global vascular plant inventory (Baker et al., 2000; Tanhan et al., 2007; Van der Ent et al., 2013b). This particularity, coupled with the potential of these plants in commercial and scientific terms, provides a strong foundation for further study (Pollard et al., 2014). The limitations of using hyperaccumulator plants for phytoextraction stem primarily from the scarcity of potential species and the high concentrations of dangerous metals possibly toxic to plants, e.g. HM (As, Cd, Hg, Pb or Se) which are not essential for plants. That is to say, the plant in its normal growth and metabolism does not use these metals, and their presence alters physiological processes at a cellular and molecular level by virtue of inactivating enzymes and blocking functional groups of molecules that are highly metabolically significant for the plant. In so doing, damage is caused in the cellular membrane—the deterioration of cellular macromolecules—and, finally, in the DNA (Rascio and Navari, 2011). Likewise, these HM are highly toxic due to their reactivity with S and N atoms present in the amino acids DNA proteins (Clemens, 2001). Another limitation is related to weather: climactic conditions, drought and flooding can interfere with or inhibit plant growth, reduce remediation efficiency and/or prolong treatment duration (Robinson et al., 2015; Vanek et al., 2010). Hyperaccumulator plant biomass must be picked and removed, followed by metal recovery or proper disposal (Prasad, 2004).

The success of phytoextraction and the commercial use in phytomining, see Section 2.3, and 3.3, has some implications that oppose this hypothesis, as an example, Robinson et al. (2015) analyzed phytoextraction effectiveness in a time-frame <25 years, and they also determined that there is not a commercial use for phytomining of valuable metals. Otherwise, phytoextraction needs specific concentrations of HM in both substrate and plant for an effective decontamination; this represents a hypothetical idea. In addition, it indicates that phytomining may have a grater ecological footprint than conventional mining. Nevertheless, it is crucial to demonstrate phytoextraction convincing basic mass-balance calculations, through new plants and soil investigation and in a full-scale file operation.

4. Methodology

The authors identified and selected the following group of words. “Wetlands, Heavy Metals, phytoremediation, phytoremediation technologies, native species, exotic species, herbaceous plants, environmental, biodiversity.” These words were searched individually and in groups in different database all over the world. The search returned about 3568 documents; of these, 304 were chosen based on relevance to the present meta-review's objectives. The correlation between keywords and the number of articles selected was as follows: Wetlands (10), HM (108), phytoremediation (112), hyperaccumulation (97), native species (12), exotic species (21), herbaceous plants (110), environmental (15), biodiversity (6) and plants (22). From a list of Bogota's wetland plants, provided by the Botanical Garden of Bogotá “José Celestino Mutis” and found in the city's Environmental Management Plans, the following information was searched with criteria of species and family: status, presence of invader character and global distribution. From this list, native, endemic, and non-invader plants were selected. Most of these plants also belong to predominant families in HM accumulation processes. Information about HM studies of all of these plants was researched at a family, gender or specie level. The result was 41 species, shown in Section 5.

5. Analysis of research on heavy metals (HM) and native herbaceous vegetation species

Below is a list of plants reported in the information of the characterization of the vegetation of Bogota's wetlands provided by the “Botanical Garden of Bogotá José Celestino Mutis”. Bogota's wetlands are home to more than 90 herbaceous plants species (Vargas et al., 2012), which are also distributed globally. Table 2 includes species status (native and/or endemic) and the “Global Distribution”, based on information from standard regional flora and online searches of herbarium records from the Missouri Botanical Garden, Calflora Database, Global Invasive Species Database, Catalogue of Plants and Lichens of Colombia, Catalog of Bogota's Invasive Wetland Plants, Navarra's Floristic Catalog, Catalogue of the Vascular Plants of Venezuela, Illustrated Catalog of Cundinamarca's Plants, Catalogue of the Vascular Plants of the Department of Antioquia, and Catalogue of the Southern Cone's Vascular Plants.

Of the herbaceous species aforementioned, emphasis is placed on species with no human intervention, just a natural process, known as species endemics (exclusives and unique of a specific geographic area) and natives (can found in a wide geographic range in the word) (Calvachi, 2002), that contribute positively to the biodiversity of ecosystems and represent evolution and residence in an area across millennia. Furthermore, these species are constitutive elements that help regulate the ecosystem, maintain equilibrium and are capable of adjusting to the biogeographic conditions of their growth habitat (Ojasti, 2001). In addition, these species are important insofar as HM are concerned: see Table 2 for a summary of the literature review related to HM and native and/or endemic species selected.

Of the 41 species described, five are endemic and 36 are native (not one presents invasive behavior). The species come from the following 18 families: Apiaceae, Asteraceae, Brassicaceae, Cyperaceae, Equisetaceae, Hypericaceae, Juncaceae, Lamiaceae, Marsilaceae, Oenotheraceae, Plantaginaceae, Plumbaginaceae, Poaceae, Polygonaceae, Potamogetonaceae, Ranunculaceae, Scrophulariaceae/Calceolariaceae, Solanaceae. Studies directly related to HM have been done for only 19 plants.

Most studies have looked into the toxic effects of metals in plants. Hence, the need for studies on plant tolerance of metals becomes even greater. Only 11 species studied included information on family (even fewer included discussion of genus); (5) reported hyperaccumulation studies, (3) accumulation and (3) metal tolerance (see Information on Publications about Metals column). Studies on hyperaccumulation for most native herbaceous vegetation species listed are lacking, as are studies focusing on the phytoremediation of HM.

6. Analysis of research on heavy metals (HM) and herbaceous plants

In contrast to the results discussed in Section 5, there are a myriad of studies on the accumulation of HM in non − native plants, as Table 3 shows.

Most species studied in terms of HM —i.e. phytoremediation processes, among others—are naturalized (*Marrubium vulgare, Chenopodium album, Chenopodium polyspermum, Hypochaeris radicata, Malva sylvestris, Portulaca oleracea, Silybum marianum, Taraxacum officinale, Trifolium pratense, Trifolium pratense, Cyperus difformis*), cultivated (*Armeria marítima, Pelargonium* sp., *Allium schoenoprasum*), adventitious (*Sonchus oleraceus, Bidens pilosa),* naturalized and adventitious *(Sorghum bicolor, Brachiaria decumbens*), exotic with documented invasive behavior *(Cotula coronopifolia)* according to the Catalog of Invasive Exotic Species, and of invasive nature according to the Global Invasive Species Database (*Cynodon dactylon, Bidens pilosa, Sonchus oleraceus, Hypericum perforatum, Hypochaeris radicata*, *Taraxacum officinale*, *Trifolium repens*). Some of the aquatic plants, e.g. *Eichhornia crassipes* (Farago and Parsons, 1994), *Lemna minor* (Vardanyan and Ingole, 2006; Wang et al., 2002) and *Azolla pinnata* (Arora et al., 2005), *Myriophylhum aquaticum*, *Ludwigina palustris*, and *Mentha aquatic* (Kamal et al., 2004), have been the subject of research into rhizofiltration, phytodegradation, phytoextraction and their ability to remove heavy metals from contaminated water. Similarly, many aquatic species have been used in the bioremoval of HM: *Fillifuloides azolla, A. pinnata, Typha orientalies* and *Salvinia molesta*, among others (Ghosh and Singh, 2005b). In summary, studies on hyperaccumulation, accumulation and heavy metal tolerance for species listed in Fig. 1 have centered on some naturalized species and species with invasive behavior.

7. Results

Of the wide range of extant native or endemic herbaceous species, Table 2 contains the following: *Baccharis latifolia, Bromus catharticus, Calceolaria bogotensis, Carex buchananii, Carex lanuginosa, Carex lurida, Cestrum buxifolium, Cyperus bipartitus, Cyperus rufus, Eleocharis dombeyana, Eleocharis montana, Eleocharis palustris, Equisetum bogotense, Fuirena incompleta, Gratiola bogotensis, Hydrocotyle umbellata, Hypericum humboldtiana, Hypericum humboldtii, Juncus densiflorus, Juncus effusus, Juncus microcephalus, Juncus ramboi colombianus, Juncus ramboii Barros* subsp. *Colombianus Balslev, Kyllinga brevifolia, Ludwigia peploides, Ludwigia peruviana, Marsilea mollis, Polygonum hydropiperoides, Polygonum segetum, Potamogeton paramoanus, Ranunculus flagelliformis, Senecio carbonelli, Physalis peruviana, Hyptis capitata, Nicotiana tabacum, Arabidopsis thaliana, Leersia hexandra, Bidens triplinervia Kunth, Lepidium bipinnatifidum, Plantago orbignyana and Potamogeton pectinatu*. These species are found worldwide, and are commonly found in wetlands. Table 2 displays the global distribution of each species listed above; these species can be found in Colombia, New

Table 2
Examples of endemic and native herbaceous vegetation species studied in terms of heavy metals.

Species	Species Status		Endangered species	Global distribution	Information on Publications about Metals	Countries of cited publications	Reference
	Native	Endemic					
Family: Apiaceae							
Hydrocotyle umbellata	x			Colombia, present throughout the American continent	Apiaceae Family Pb, Zn, Cu, Cd, As	India	(Sharma and Agrawal, 2005)
Family: Asteraceae							
Baccharis latifolia	x			Colombia, Northern Andes	As, Hyperaccumulator of Pb	Brazil, USA, Mexico, Spain	(Durán, 2010; Haque et al., 2008; Menezes et al., 2015; Santos et al., 2012)
Bidens triplinervia Kunth	x				Hyperaccumulator of Cu, Fe, Accumulator of Mn, Zn and in greater proportion of Pb	Spain, Peru	(Bech et al., 2012; Durán, 2010)
Senecio carbonelli		x	x	Colombia-Bogota Endemic Plant	Senecio Genus Ni, Zn, Cu, Mn, Al	Cuba, Canada, Africa, Poland	(Boyd et al., 2008; Reeves et al., 1999)
Family: Brassicaceae							
Lepidium bipinnatifidum	x			Venezuela, Colombia, Ecuador, Peru, Bolivia, Brazil and Argentina	Accumulator of Pb	Spain	(Durán, 2010)
Family: Cyperaceae							
Carex buchananii	x			Colombia and New Zealand	Cd, Zn, Ni	France, Spain, Ireland	(Ladislas et al., 2014; Matthews et al., 2005; Walker et al., 2004)
Carex lanuginosa	x			Colombia, and reported in the American continent	Cd, Zn, Ni, Al, Co, Cr, Ni, Pb, Cu		
Carex lurida	x		x				
Cyperus bipartitus	x			Colombia, North, Central and South America	Mn, Zn, Cd, Ni, Pb, Cu, Cr	Malaysia, Egypt, Tanzania, USA	(Akinbile et al., 2012; Ewais, 1997; Mganga et al., 2011)
Cyperus rufus	x			Colombia, Argentina, Chile			
Eleocharis dombeyana	x			Colombia, Costa Rica, Central and South America	As	Mexico, Brazil, Argentina	(Litter et al., 2012)
Eleocharis montana	x			Colombia, the Antilles, South Eastern United States to Chile and Argentina			
Eleocharis palustris	x			Colombia, America, Europe, Asia and Africa			
Fuirena incompleta	x		x	Mexico to Argentina, Colombia and Uruguay	Apiaceae Family Pb, Zn, Cu, Cd, As	India	(Das and Maiti, 2008)
Kyllinga brevifolia	x			Colombia, Tropical America. Paraguay, India, Malaysia, the Philippines and China	U, Th, Sr, Ba, Ni and Pb	China	(Li et al., 2011)
Family: Equisetaceae							
Equisetum bogotense	x			Colombia, Costa Rica and South America	As	Mexico, Brazil, Argentina	(Litter et al., 2012)
Family: Hypericaceae							
Hypericum humboldtiana		x	x	Colombia-Bogota Endemic Plant	*Hypericum perforatum* was tolerant to Cr, presents effects to Ni	Italy, Canada	(Murch et al., 2003; Tirillini et al., 2006)
Hypericum humboldtii		x	x	Colombia Endemic Plant			
Family: Juncaceae							
Juncus densiflorus	x			Colombia, Venezuela, Brazil, Argentina, Uruguay and Paraguay	Juncaceae Family Cu, Pb, Cd, studies on plant tolerance	Austria, Australia, Scotland	(Archer and Caldwell, 2004; Johnston and Proctor, 1977; Wenzel and Jockwer, 1999)
Juncus effusus	x			Colombia, common to large number of countries	Cd, Zn, Ni, Al, Co, Cr, Ni, Pb, Cu	France, Poland, China	
Juncus microcephalus	x		x	Colombia, Central Mexico to Bolivia and South West Brazil	Juncaceae Family Cu, genus juncus Pb, Cd, studies on plant tolerance	Austria, Australia, Scotland	
Juncus ramboi colombianus		x	x	Colombian Endemic Plant	Juncaceae Family with Cu, Pb, Cd, studies on plant tolerance		
Juncus ramboii Barros subsp. *Colombianus Balslev*	x			Colombia, Southeastern Brazil	Juncaceae Family with Cu, Pb, Cd, studies on plant tolerance		(Ladislas et al., 2014; Samecka and Kempers, 2001; Yanqun et al., 2004)
Family: Lamiaceae							
Hyptis capitata	x			Colombia, neotropics; introduced in tropical Asia and Pacific	Accumulator of Cd and Cu	Australia	(Nedelkoska and Doran, 2000)

(*continued on next page*)

Table 2 (*continued*)

Species	Species Status: Native	Species Status: Endemic	Endangered species	Global distribution	Information on Publications about Metals	Countries of cited publications	Reference
Family: Marsilaceae							
Marsilea mollis	x			North America, Colombia to Argentina	Cu	Canada	(Kamal et al., 2004)
Family: Oenotheraceae							
Ludwigia peploides	x			Colombia, South America, Greater Antilles and East Asia	Cu, Pb, Cr, Zn, Cd, Ni and Hg	India, Tanzania	(Das and Maiti, 2008; Kamal et al., 2004; Mganga et al., 2011)
Ludwigia peruviana	x			Colombia, South America; introduced in South Asia and Australia			
Family: Plantaginaceae							
Gratiola bogotensis	x		x	Colombia	Plantaginaceae Family Pb, Zn, Cu, Cd, As	Spain	(Del Río et al., 2002)
Plantago orbignyana	x			Southern Colombia to Bolivia	Accumulator of Pb y Zn		(Durán, 2010)
Family: Plumbaginaceae							
Arabidopsis thaliana	x	x		Native to Europe, Asia, Africa, present in all five continents, poor in South America, Asia and Canada	Hg	Germany	(Battke et al., 2008)
Family: Poaceae							
Bromus catharticus	x			Colombia, Guatemala to South America	Cu, Zn	USA	(O'Dell et al., 2007)
Leersia hexandra	x			Colombia, the Antilles, Canada to Argentina and Uruguay	Hyperaccumulator of Cr		(Zhang, Liu et al., 2007)
Family: Polygonaceae							
Polygonum hydropiperoides	x			Colombia, the Antilles, Canada to South America	Cu, Pb	USA	(Kamal et al., 2004)
Polygonum segetum	x			Mexico to Colombia, and the Antilles	Genus Polygonum accumulator of Cd	Japan	(Shinmachi et al., 2003)
Family: Potamogetonaceae							
Potamogeton paramoanus	x		x	Colombia, Venezuela to Bolivia	Potamogeton Cd, Pb, Cr, Ni, Zn y Cu	Turkey	(Demirezen and Aksoy, 2004)
Potamogeton pectinatus	x			Colombia, common to a large number of countries	Accumulator of Cd, Pb, Cr, Ni, Zn y Cu		
Family: Ranunculaceae							
Ranunculus flagelliformis	x		x	Colombia, Venezuela to Argentina	Genus Ranunculus Cr	Turkey	(Aksoy et al., 2005)
Family: Scrophulariaceae/Calceolariaceae							
Calceolaria bogotensis	x		x	Colombian Native Endemic Plant	Cu	USA	(Silk et al., 2006)
Family: Solanaceae							
Cestrum buxifolium	x			Colombia, Venezuela to Peru	Solanaceae Family Cr, Hg, Pb, Zn, Cu, Cd, As	Poland, Spain	(Bennicelli et al., 2004; Del Río et al., 2002)
Nicotiana tabacum	x			Colombia and South America. Native cultivated plant	Accumulator of Cd and Cu. Tolerance to Hg	Australia, Poland	(Bennicelli et al., 2004; Nedelkoska and Doran, 2000)
Physalis peruviana	x			Colombia, American continent. Native cultivated plant	Cr, Mn, Fe, Co, Ni, Cu, Zn, Cd, Hg, Pb	Kenya	(Maobe et al., 2012)

Table 3
Examples of herbaceous vegetation species studied in terms of heavy metals.

Species	Information on Publications about Metals	Countries of cited publications	Reference
Family: Asteraceae			
Conyza dioscorides	Zn, Cu, Pb	Egypt	(Abou et al., 2007)
Bidens pilosa	Hyperaccumulator of Cd. Excluders of As. Appropriate adaptation and a good level of tolerance with Pb, Cr, and Ni, characterized by the greater absorption of Cr	China	(Sánchez, 2010; Sun et al., 2009a, 2009b)
Foeniculum bulgare	Studies on Pb, Zn, Cu, Cd, As,	Spain	(Del Río et al., 2002)
Hypochaeris radicata	Zn	Portugal	(Moreira et al., 2011)
Senecio azulensis	Hyperaccumulator of Ni	Cuba	(Reeves et al., 1999)
Senecio biseriatus			
Senecio brasiliensis	Zn		
Senecio coronatus	Hyperaccumulator of Ni and Zn; accumulator of Cu and Mn	Africa, Poland	(Boyd et al., 2008; Mesjasz et al., 1994)
Senecio ekmanii	Hyperaccumulator of Ni	Cuba	(Reeves et al., 1999)
Senecio plumbeus			
Senecio rivalis			
Senecio subsquarrosus			
Silybum marianum	Studies on Pb, Zn, Cu, Cd, As	Spain	(Del Río et al., 2002)
Sonchus oleraceus	Hyperaccumulator of Pb	Spain, China	(Durán, 2010; Xiong, 1997)
Taraxacum officinale	Pb, Cd, Cu, Zn, Hg, Fe, Co, Cr, Mo	Case Studies, Poland	(Malawska and Wiołkomirski, 2001)
Family: Brassicaceae			
Armeria maritima	Pb and Zn	Poland	(Olko et al., 2008)
Family: Chenopodiaceae			
Chenopodium ambrosioides	Studies on effects of Cd, Ni and Pb	Egypt	(Ewais, 1997)
Chenopodium album	Tolerance to Pb, Cu, Zn, Mn	Spain, China	(Hu et al., 2012; Walker et al., 2004)
Family: Cistaceae			
Cistus salvifolius	Hyperaccumulator of Al	Spain	(Durán, 2010)
Family: Convolvulaceae			
Chenopodium amaranticolor	U	India	(Eapen et al., 2003)
Family: Cyperaceae			
Schonoeplectous americanus	Tolerance to As	Mexico	(Valles and Alarcón, 2014)
Cyperus difformis	Studies on effects of Cd, Ni and Pb	Egypt	(Ewais, 1997)
Eleocharis machrostachya	Tolerance to As. Efficient rhizofiltrator of As	Mexico, Brazil, Argentina	(Litter et al., 2012; Valles and Alarcón, 2014)
Family: Equisetaceae			
Equisetum arvense	Zn and some possible accumulators of Pb and Cu	USA, Alaska	(Cannon et al., 1968)
Equisetum kansanum			
Equisetum variegatum			
Family: Fabaceae			
Lupinus albus	Zn, Mn, Co, Cd	Switzerland	(Page et al., 2006)
Trifolium pratense	Cd, Zn	USA, Slovenia, China	(Grčman et al., 2001)
Trifolium repens		Belgium, Lithuania	(Lambrechts et al., 2014; Stravinskienė and Račaitė, 2014)
Family: Geraniaceae			
Pelargonium sp.	Pb	France	(Arshad et al., 2008)
Family: Hypericaceae			
Hypericum perforatum	Tolerance to Cr	Italy	(Tirillini et al., 2006)
Family: Juncaceae			
Juncus squarrosus	Pb, Zn, Ca, Ni, Cr, Co, Fe, Mg	Scotland	(Johnston and Proctor, 1977)
Juncus usitatus	Tolerance to Cd, Pb	Australia	(Archer and Caldwell, 2004)
Family: Liliaceae			
Allium schoenoprasum	Cd	Israel	(Barazani et al., 2004)
Family: Malvaceae			
Malva nicaeensis	Studies on Pb, Zn, Cu, Cd, As	Spain	(Del Río et al., 2002)
Malva sylvestris			(Pastor and Hernández, 2002)
Family: Poaceae			
Cynodon dactylon	Zn, Cu, Pb	Egypt	(Abou et al., 2007)
Brachiaria decumbens	Cd, Zn, Pb	Portugal	(Santos et al., 2007)
Digitaria sanguinolis	Studies on effects of Cd, Ni and Pb	Egypt	(Ewais, 1997)
Sorghum bicolor	Cd, Co, Cu, Zn	Italy	(Marchiol et al., 2007)
Family: Polygonaceae			
Polygonium thunbergii	Studies on accumulation of Cd	Japan	(Shinmachi et al., 2003)
Family: Portulaceae			
Portulaca oleracea	Studies on toxicity for Cd, Cu, Zn, Hg, Pb, Se, Hg	India	(Thangavel and Subburam, 1998; Thangavel et al., 1999)
Family: Ranunculaceae			
Ranunculus sphaerosphermus	Cr	Turkey	(Aksoy et al., 2005)
Family: Solanaceae			
Datura stramonium L.	Studies on Pb, Zn, Cu, Cd, As	Spain	(Del Río et al., 2002)
Solanum elaeagnifolium	Tolerance to Cr	Poland	(Bennicelli et al., 2004)
Solanum nigrum L.	Studies on Pb, Zn, Cu, Cd, As,	Spain	(Del Río et al., 2002)
Family: Typhaceae			
Typha angustifolia	Cd, Pb, Cr, Ni, Zn, Cu	Turkey	(Demirezen and Aksoy, 2004)

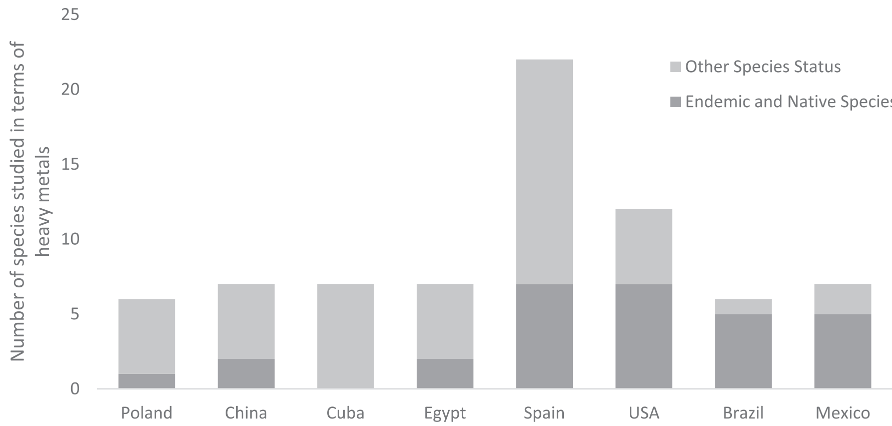

Fig. 1. Relative frequency of native herbaceous species and other species (naturalized, introduced, invasive, cultivated and adventitious) that were studied in terms of their relation to HM. Eight representative countries have been chosen, as recorded in Tables 2 and 3. The classification (species status) of each species is based on information from the data sources mentioned in Section 5.

Zealand, Chile, Costa Rica, the Antilles, Paraguay, India, Malaysia, the Philippines, China, Uruguay, Venezuela, Brazil, Australia, Guatemala, Canada, Peru; in other words, they are found across Asia, Europe, North, Central and South America. Most of those countries are exposed to metalliferous, serpentine (ultramafic) or calamine soils, and in some of them, there are wetlands with HM contamination problems. See Section 2.3 for more information.

These 41 species named in the previous paragraph reported non-invasive behavior. Besides, all of these species have been studied in relation to HM, some directly overlapped in terms of the species studied, while others indirectly overlapped with species from the same family or genus (see Tables 2 and 3, Sections 5 and 6) in countries such as Austria, Australia, Scotland, Brazil, USA, Mexico, Canada, China, Cuba, Poland, France, Spain, Ireland, India, Tanzania, Malaysia, Italy, Argentina, Turkey, Egypt, Switzerland, Portugal, Germany, Israel, Peru, Slovenia, Belgium, Lithuania, United States (specifically Alaska) and on the African Continent. Additionally, the majority of the studies compiled only record heavy metal concentrations; most do not analyze plant processes.

Of the 41 plants mentioned in this article, noteworthy vegetation species include *Hypericum humboldtii, Senecio carbonelli, Hypericum humboldtiana, Juncus ramboi colombianus, Calceolaria bogotensis* (endemic herbaceous species); *Juncus microcephalus, Gratiola bogotensis, Carex lurida, Fuirena incompleta, Potamogeton paramoanus, Arabidopsis thaliana* and *Ranunculus flagelliformis* (native species). The importance of these endemic species is intertwined with their endangered status.

As can be seen in Table 2, of the predominant families in HM accumulation processes, see Section 3.2, there are three: *Asteraceae*, *Cyperaceae* and *Poaceae*. The following species are part of these three families: *Baccharis latifolia, Senecio carbonelli, Carex buchananii, Carex lanuginosa, Carex lurida, Cyperus bipartitus, Cyperus rufus, Eleocharis leocharis dombeyana, Eleocharis montana, Eleocharis palustris, Fuirena incompleta, Kyllinga brevifolia* and *Bromus catharticus*.

8. Discussion and conclusions

In Section 2.2, the problem of wetland degradation is outlined, explaining that HM contamination is one of the most relevant concerns. In Section 2.3, the presence of HM worldwide is discussed. In light of this grave situation, the use of technologies to minimize the presence of HM —and concomitant toxicity depending on concentration of metal in the substrate—becomes imperative. Therefore, special attention is paid to the source of heavy metal contamination, that is, the activities that contribute to soil, air and water contamination. See Section 2.3 for common examples of these sources. As previously mentioned, these activities affect both biotic and abiotic components, which in turn affect the ecosystem structure and functionality, eventually leading to the loss of landscape. In an effort to mitigate these deleterious effects, heavy metal contamination has been addressed with vegetation species in bioremediation processes, often mentioned as phytoremediation; see Section 3 for an in-depth discussion of phytoremediation.

Given the previously reported potential of species found on wetlands, it is crucial to contribute to the preservation of native species faced with an endangerment status, as well as to strengthen the sustainable uses of phytoremediation as a cleaning alternative for environmental contaminants. To this end, phytoremediation has gained popularity among scientists and professionals: phytoremediation improves the environment as part of ecological recovery and restoration processes (Bhargava et al., 2012). In addition, as concluded by Yang et al. (2005), for a greater success in phytoremediation technology, a multidisciplinary work of different professionals is essential. Likewise, it is necessary to study the systems involved in the environment, since the soil system is one of the decisive components in the process of understanding the behavior of an ecosystem. As mentioned by Smith et al. (2015), the soil's importance lies on the fact that it hosts the largest diversity of organisms on land. Specifically, in the case of HM, and based on the wide study on this substrate compared to others, it is concluded that the use of the knowledge already acquired can improve the provision of ecosystem services. According to the latter, and as mentioned by Brevik et al. (2015), after demonstrating the examples involved in the study of soil science, it was concluded that a holistic investigation of this component requires an interdisciplinary study of the atmosphere, biosphere, hydrosphere, and lithosphere, linking as central convergence axis the biodiversity (mechanisms, organisms and soil properties), biogeochemical cycling, hydrology, human health, social sciences and soil threats; important topics to be incorporated in a future article. In the same way, it is necessary to involve sustainable practices in the high impact activities on the soil or growth substrate, which will lead to restoring and conserving biodiversity (Decock et al., 2015).

Based on the lack of current knowledge on phytoremediation, the fact that most thorough studies on this technology delve into the mechanisms underlying hyperaccumulation, the advantages and limitations of the phytoremediation process, set out in Section 3, indicate that further research is warranted. There is a pressing need to focus on the mechanism behind "phyto" technologies, which can be studied based on the already existing knowledge, such as hyperaccumulation, one of the most studied. In consequence, it will allow researchers to move beyond the obstacles created by the lack of specific experiments and experimental plants, as Schaller et al. (2013) indicate, as a limitation in the recovery potential of wetlands. In addition, as Proctor (1999) argues, the phytobiography of regions with high concentrations of HM requires further investigation. All of this, in order to deal with phytoremediation limits, and use what nature brings, instead of developing antagonist technologies.

As we can see in Section 2.4 future studies on native or endemic herbaceous vegetation species could help avoid the use of exotic and invasive species that have transformed biogeographical barriers. The spread of the exotic and invasive species, whether accidental or deliberate, into new habitats, constitutes an environmental problem. The introduction of these exotic species is one of the main issues facing existing biological communities on wetlands, often resulting in the displacement or extinction of the zone's characteristic species, which may lead to native biota loss, mentioned by Yang et al. (2005). Likewise, reduced biodiversity in various ecosystems all over the world, as noted by Tilman and Lehman (2001), poses one of the great ecological challenges to humankind. Exotic species with invasive behavior cause perturbations in vegetation and animal species and affect the environment. Such changes may result in a native species becoming invasive (Montes et al., 2007). Likewise, in accordance with Rodríguez (2009), invasive species disrupt native mutualist edaphic networks, and thereby limit an ecosystem's natural capacity for resilience and recovery.

Also, biological invasions alter the structure and function of the ecosystem into which they are introduced (Pimentel et al., 2001). The use of invasive species presents challenges to spreading control; wetlands are especially vulnerable to invasion by new species due to their position as ecotones or interfaces between terrestrial and aquatic environments, exposing both environments to invasion, as Montes et al. (2007) indicated; see Section 2.1, about wetland importance. To prevent this undesirable outcome, in agreement with Díaz et al. (2012), the recovery of wetlands is sought via the use of native species and management for control and eradication of invasive vegetation.

The study of plants with endangered status, see Sections 5 and 7, would allow scientists to create propagation processes and thus, increase the number of individuals. An in-depth study would broaden understanding of these species, possibly grant them an economic valuation that could facilitate decisions fostering protection and development and rational use, as is described by Barbier et al. (1997). Seeing these ecosystems from a purely economic perspective is not mutually exclusive with resource management from an ethical and cultural conception of water, give an account by Grayman et al. (2012). The unavoidable commercial perspective denominated "biodiversity prospecting" defined by Prasad (2003), would lead to the discovery of wild plants that could decontaminate polluted environments of the world, on target of conserving nature.

Studies on phytoremediation may also enhance knowledge of waste management, which reinforces the need for further studies, pointed out by Boyd et al. (2008) and Pollard et al. (2014). These studies should be carried out in native sources of propagules, i.e., Colombia, Venezuela, Chile, Costa Rica, Paraguay, Guatemala, Costa Rica and Cuba; see Table 2. Looking at those countries with studies on native, endemic or other species and HM, Fig. 1—at least those analyzed in the present article—three things become evident. One, the majority of reported studies are over non-native species. Second, Spain and the US are countries with more native and endemic studies than others.Third, Spain is the most reported country with non-native and non-endemic heavy metal studies.

The uses of native herbaceous species for phytoremediation process must be researched in depth, supported by the plants' adaptation to the climate of the contaminated site, toxicity level and properties of substrate, among other factors described by Xue et al. (2014). As Conesa et al. (2009) mentioned, the herbaceous species fast adaptation to these conditions, compared to other plants, is allowed by the production of various genotypes in a shorter time because of their short life cycles. In light of the aforementioned issues, the authors of this paper recommend investigating the behavior between HM and the 41 species mentioned in Table 2, described in Sections 5 and 7. The global distribution of the plants recommended, combined with the worldwide presence of soils with high concentrations of HM, naturally occurring, or the product of anthropic related contamination, set the stage for the continued investigation of phytoremediation. Future research should seek to identify the mechanism by which these plants transport the metal from the growth substrate (see Section 3 for more information) in order to establish possible uses in the phytoremediation of HM. The aforementioned can be accomplished by identifying the mechanisms and techniques employed by plants themselves, as well as the plants' morphological characteristics, height, coverage, biomass and seed production, growth capacity, bio-concentration factors, concentrations and bioavailability of metals (in the plant and the substrate) and other determining factors and characteristics, according to Ghosh and Singh (2005a). Kardanpour et al. (2015) mentioned the heterogeneity characterization as the first, and in some cases, the most important step in soil contaminant studies.

Phytoremediation studies provide information on physiological, photochemical, ecological and environmental aspects (such as life cycles, morphological changes, chemical, biological structure and composition of growth substrate, the move towards environmental friendly technologies, etc.), biogeochemical cycles and possible effects on the trophic chain. Plants could be used as accumulation indicators, as Zechmeister et al. (2003) revealed and investigated for indication of water quality. This kind of investigations brings the opportunity to establish ecological risks through the food chain in ecosystems contaminated with HM.

It is essential to engage in further research to increase our understanding of the vegetation species found especially on wetlands. Supported by investigations, which show that many wetland plants can colonize heavy metal-polluted areas, and are found to deal with a wide range of soil metals (Pb, Zn, Cu and Cd), and demonstrated their possible use in phytoremediation, as it is mentioned by Deng et al. (2004). Investigations such as those developed by Conesa et al. (2009), Galfati et al. (2011) and García et al. (2011), mentioned the experience of research on native vegetation with potential use in phytoremediation, specifically with herbaceous plants in Xue et al. (2014), wetland plants in Mateos (2013) and Wang et al. (2002). The importance of biodiversity is gradually being considered for the decontamination of ecosystems affected by metals, as it is mentioned by Prasad (2003), is gradually being considered for the decontamination of ecosystems affected by metals. An overriding research into native and endemic biodiversity in cities around the world is critical, for being nature's remnants in urban areas.

Acknowledgments

The authors would like to recognize the invaluable information provided by Bogota's Botanical Garden, Botanical Garden of Bogota José Celestino Mutis, as well as the support of the Distrital

University of Bogota Francisco José de Caldas.

References

Abínzano, R., 1998. Globalization, Regions y Borders.

Abou, R., Ghanem, N., Ghanem, K., Al-Kolaibe, A., 2007. Phytoremediation potential of crop and wild plants for multi-metal contaminated soils. RJABS 3, 370–376.

Adams, N., Carroll, D., Madalinski, K., Rock, S., 2000. Introduction to Phytoremediation. United States Environmental Protection Agengy, Office of Research and Development, Washington D.C. USA.

Akinbile, C., Yusoff, M., Ahmad, A., 2012. Landfill leachate treatment using subsurface flow constructed wetland by Cyperus haspan. Waste Manag. 32, 1387–1393. http://dx.doi.org/10.1016/j.wasman.2012.03.002.

Aksoy, A., Demirezen, D., Duman, F., 2005. Bioaccumulation, detection and analyses of heavy metal pollution in Sultan marsh and its environment. Water Air Soil Poll. 164, 241–255. http://dx.doi.org/10.1007/s11270-005-3538-x.

Alcaldía Mayor de Bogotá, 2006. Política de humedales del Distrito Capital. Departamento Administrativo del Medio Ambiente. DAMA, Bogotá D.C., Colombia.

Alcaldía Mayor de Bogotá D.C, 2010. Informe de la IX Fase del programa de seguimiento y monitoreo de efluentes y afluentes al recurso hídrico de Bogotá. Convenio Interadministrativo 020/2008, suscrito entre la Secretaria Distrital de Ambiente (SDA) y la Empresa de Acueducto y Alcantarillado de Bogotá ESP (EAAB-ESP), del 2010, Bogotá D.C., Colombia.

Ali, H., Khan, E., Sajad, M., 2013. Phytoremediation of heavy metals—concepts and applications. Chemosphere 91, 869–881. http://dx.doi.org/10.1016/j.chemosphere.2013.01.075.

Archer, M., Caldwell, R., 2004. Response of Six australian plant species to heavy metal contamination at an abandoned mine site. Water Air Soil Poll. 157, 257–267. http://dx.doi.org/10.1023/B:WATE.0000038900.66771.bf.

Arora, A., Saxena, S., Sharma, D., 2005. Tolerance and phytoaccumulation of Chromium by three Azolla species. World J. Microb. Biotechnol. 22, 97–100. http://dx.doi.org/10.1007/s11274-005-9000-9.

Arshad, M., Silvestre, J., Pinelli, E., Kallerhoff, J., Kaemmerer, M., Tarigo, A., Shahid, M., Guiresse, M., Pradere, P., Dumat, C., 2008. A field study of lead phytoextraction by various scented Pelargonium cultivars. Chemosphere 71, 2187–2192. http://dx.doi.org/10.1016/j.chemosphere.2008.02.013.

Athar, R., Ahmad, M., 2002. Heavy metal toxicity: effect on plant growth and metal uptake by wheat, and on free living azotobacter. Water Air Soil Poll. 138, 165–180. http://dx.doi.org/10.1023/A:1015594815016.

Baker, A., Brooks, R., 1989. Terrestrial higher plants which hyperaccumulate metallic elements - a review of their distribution, ecology and phytochemistry. Biorecovery 1, 81–126.

Baker, A., McGrath, S., Reeves, R., Smith, J., 2000. Metal hyperaccumulator plants: a review of the ecology and physiology of a biological resource for phytoremediation of metal-polluted soils. In: Terry, N., Bañuelos, G. (Eds.), Phytoremediation of Contaminated Soils and Waters. CRC Press LLC, Boca Raton, FL, USA, pp. 95–117 doi:10.1.1.467.809.

Barazani, O., Dudai, N., Khadka, U., Golan-Goldhirsh, A., 2004. Cadmium accumulation in Allium schoenoprasum L. grown in an aqueous medium. Chemosphere 57, 1213–1218. http://dx.doi.org/10.1016/j.chemosphere.2004.08.037.

Barba, L., Edith, L., 2002. Conceptos básicos de la contaminación del agua y parámetros de medición. Universidad del Valle.

Barbier, E., Acreman, M., Knowler, D., 1997. Valoración económica de los humedales Guía para decisores y planificadores, Communications. Oficina de la Convención de Ramsar, Gland, Suiza doi:2940073252.

Barceló, J., Poschenrieder, C., 2003. Phytoremediation;: Principles and Perspectives, Contributions to Science. Barcelona.

Battke, F., Ernst, D., Fleischmann, F., Halbach, S., 2008. Phytoreduction and volatilization of mercury by ascorbate in Arabidopsis thaliana, European beech and Norway spruce. Appl. Geochem 23, 494–502. http://dx.doi.org/10.1016/j.apgeochem.2007.12.023.

Bech, J., Duran, P., Roca, N., Poma, W., Sánchez, I., Roca-Pérez, L., Boluda, R., Barceló, J., Poschenrieder, C., 2012. Accumulation of Pb and Zn in Bidens triplinervia and Senecio sp. spontaneous species from mine spoils in Peru and their potential use in phytoremediation. J. Geochem. Explor. 123, 109–113. http://dx.doi.org/10.1016/j.gexplo.2012.06.021.

Bennicelli, R., Stępniewska, Z., Banach, A., Szajnocha, K., Ostrowski, J., 2004. The ability of Azolla caroliniana to remove heavy metals (Hg(II), Cr(III), Cr(VI)) from municipal waste water. Chemosphere 55, 141–146. http://dx.doi.org/10.1016/j.chemosphere.2003.11.015.

Bertram, C., Krätschell, A., O'Brien, K., Brückmann, W., Proelss, A., Rehdanz, K., 2011. Metalliferous sediments in the Atlantis II Deep—Assessing the geological and economic resource potential and legal constraints. Resour. Policy 36, 315–329. http://dx.doi.org/10.1016/j.resourpol.2011.09.001.

Bhargava, A., Carmona, F., Bhargava, M., Srivastava, S., 2012. Approaches for enhanced phytoextraction of heavy metals. J. Environ. Manag. 105, 103–120. http://dx.doi.org/10.1016/j.jenvman.2012.04.002.

Boyd, R., Davis, M., Balkwill, K., 2008. Elemental patterns in Ni hyperaccumulating and non-hyperaccumulating ultramafic soil populations of Senecio coronatus. S. Afr. J. Bot. 74, 158–162. http://dx.doi.org/10.1016/j.sajb.2007.08.013.

Boyd, R., Martens, S., 1998. The significance of metal hyperaccumulation for biotic interactions. Chemoecology 8, 1–7 doi:0937–7409:98:010001–07.

Boyd, R.S., 2007. The defense hypothesis of elemental hyperaccumulation: status, challenges and new directions. Plant Soil 293, 153–176. http://dx.doi.org/10.1007/s11104-007-9240-6.

Brady, K., Kruckeberg, A., Bradshaw, J., 2005. Evolutionary ecology of plant adaptation to serpentine soils. Annu. Rev. Ecol. Evol. Syst. 243–266.

Brevik, E.C., Cerdà, A., Mataix-Solera, J., Pereg, L., Quinton, J.N., Six, J., Van Oost, K., 2015. The interdisciplinary nature of SOIL. SOIL 1, 117–129. http://dx.doi.org/10.5194/soil-1-117-2015.

Budelsky, R., Galatowitsch, S., 2004. Establishment of Carex stricta Lam. seedlings in experimental wetlands with implications for restoration. Plant Ecol. 175, 91–105. http://dx.doi.org/10.1023/B:VEGE.0000048102.72101.50.

Burton, T., Tiner, R., 2009. Ecology of Wetlands, Encyclopedia of Inland Waters. Elsevier. http://dx.doi.org/10.1016/B978-012370626-3.00056-9.

Calijuri, M., da Fonseca Santiago, A., Moreira Neto, R., de Castro Carvalho, I., 2011. Evaluation of the ability of a natural wetland to remove heavy metals generated by runways and other paved areas from an airport complex in Brazil. Water Air Soil Poll. 219, 319–327. http://dx.doi.org/10.1007/s11270-010-0709-1.

Calvachi, B., 2002. Una Mirada Regional. La Biodiversidad Bogotana. Rev. La Tadeo, pp. 89–98.

Camacho, V., Ruiz, A., 2012. Marco conceptual y clasificación de los servicios ecosistémicos. Rev. Biol. Ciencias 1, 15.

Cannon, H., Shacklette, H., Bastron, H., 1968. Metal absorption by Equisetum (horsetail). U.S. Geol. Surv. Bull. (U.S.) (United States) 1278-A.

Castro, P., Valladares, F., Alonso, A., 2004. La creciente amenaza de las invasiones biológicas. Ecosistemas. http://dx.doi.org/10.7818/re.2014.13-3.00.

Celis, H., Junod, M., Sandoval, E., 2005. Recientes aplicaciones de la depuracion de aguas residuales con plantas acuáticas. Rev. Theor. Cienc. Arte Humanidades 14, 17–25.

Chaney, R., Malik, M., Li, Y., Brown, S., Brewer, E., Angle, J., Baker, A., 1997. Phytoremediation of soil metals. Curr. Opin. Biotechnol. 8, 279–284. http://dx.doi.org/10.1016/S0958-1669(97)80004-3.

Chiarucci, A., Baker, A., 2007. Advances in the ecology of serpentine soils. Plant Soil 293, 1–2. http://dx.doi.org/10.1007/s11104-007-9268-7.

Clavero, M., García, E., 2005. Invasive species are a leading cause of animal extinctions. Trends Ecol. Evol. 20, 110. http://dx.doi.org/10.1016/j.tree.2005.01.003.

Clemens, S., 2001. Molecular mechanisms of plant metal tolerance and homeostasis. Planta 212, 475–486. http://dx.doi.org/10.1007/s004250000458.

Conesa, H.M., Moradi, A.B., Robinson, B.H., Kühne, G., Lehmann, E., Schulin, R., 2009. Response of native grasses and Cicer arietinum to soil polluted with mining wastes: implications for the management of land adjacent to mine sites. Environ. Exp. Bot. 65, 198–204. http://dx.doi.org/10.1016/j.envexpbot.2008.09.004.

Cortés, J., Estupiñán, L. (Eds.), 2016. Las huellas del agua. Propuesta metodológica para identificar y comprender el límite de los humedales de Colombia, first ed. Bogotá D.C., Colombia.

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

Cunningham, S., Berti, W., 2000. Phytoextraction and phytostabilization: technical, economic and regulatory considerations of the soil-lead issue. In: Terry, N., Buñuelos, G. (Eds.), Phytoremediation of Contaminated Soils and Waters. CRC Press LLC, Boca Raton, FL, USA, pp. 363–380 doi:10.1.1.467.809.

Cunningham, S., Berti, W., Huang, J., 1995. Phytoremediation of contaminated soils. Trends Biotechnol. 13, 393–397. http://dx.doi.org/10.1016/S0167-7799(00)88987-8.

Daily, G., 2000. Management objectives for the protection of ecosystem services. Environ. Sci. Policy 3, 333–339. http://dx.doi.org/10.1016/S1462-9011(00)00102-7.

Das, M., Maiti, S., 2008. Comparison between availability of heavy metals in dry and wetland tailing of an abandoned copper tailing pond. Environ. Monit. Assess. 137, 343–350. http://dx.doi.org/10.1007/s10661-007-9769-0.

Decock, C., Lee, J., Necpalova, M., Pereira, E.I.P., Tendall, D.M., Six, J., 2015. Mitigating N_2O emissions from soil: from patching leaks to transformative action. SOIL 1, 687–694. http://dx.doi.org/10.5194/soil-1-687-2015.

Del Río, M., Font, R., Almela, C., Vélez, D., Montoro, R., De Haro Bailón, A., 2002. Heavy metals and arsenic uptake by wild vegetation in the Guadiamar river area after the toxic spill of the Aznalcóllar mine. J. Biotechnol. 98, 125–137. http://dx.doi.org/10.1016/S0168-1656(02)00091-3.

Delgadillo, A., González, C., Prieto, F., Villagómez, J., Acevedo, O., 2011. Fitorremediación: una alternativa para eliminar la contaminación. Trop. Subtrop. Agroecosyst. 14, 597–612.

Demirezen, D., Aksoy, A., 2004. Accumulation of heavy metals in Typha angustifolia (L.) and Potamogeton pectinatus (L.) living in Sultan Marsh (Kayseri, Turkey). Chemosphere 56, 685–696. http://dx.doi.org/10.1016/j.chemosphere.2004.04.011.

Deng, H., Ye, Z., Wong, M., 2004. Accumulation of lead, zinc, copper and cadmium by 12 wetland plant species thriving in metal-contaminated sites in China. Environ. Pollut. 132, 29–40. http://dx.doi.org/10.1016/j.envpol.2004.03.030.

Díaz, A., Díaz, J., Vargas, O., 2012. Catálogo de plantas invasoras de los humedales de Bogotá. Grupo de Restauración Ecológica de la Universidad Nacional de Colombia y Secretaría Distrital de Ambiente, Bogotá D.C., Colombia.

Diez, J., 2008. Fitocorrección de suelos contaminados con metales pesados: Evaluación de plantas tolerantes y optimización del proceso mediante prácticas agronómicas. Universidad de Santiago de Compostela.

Dunbabin, J., Bowmer, K., 1992. Potential use of constructed wetlands for treatment of industrial wastewaters containing metals. Sci. Total Environ. 111, 151–168. http://dx.doi.org/10.1016/0048-9697(92)90353-T.
Durán, A., 2010. Transferencia de metales de suelo a planta en áreas mineras: Ejemplos de los Andes peruanos y de la Cordillera Prelitoral Catalana. Universidad de Barcelona.
Eapen, S., Suseelan, K., Tivarekar, S., Kotwal, S., Mitra, R., 2003. Potential for rhizofiltration of uranium using hairy root cultures of Brassica juncea and Chenopodium amaranticolor. Environ. Res. 91, 127–133. http://dx.doi.org/10.1016/S0013-9351(02)00018-X.
Elorza, M., Dana, E., Sobrino, E., 2001. Aproximación al listado de plantas alóctonas invasoras reales y potenciales en España. Lazaroa.
Ewais, E., 1997. Effects of cadmium, nickel and lead on growth, chlorophyll content and proteins of weeds. Biol. Plant 39, 403–410. http://dx.doi.org/10.1023/A:1001084327343.
Farago, M., Parsons, P., 1994. The effects of various platinum metal species on the water plant Eichhornia crassipes (MART.). Chem. Speciat. Bioavailab. 6, 1–12. http://dx.doi.org/10.1080/09542299.1994.11083218.
Flóres, C., Estupiñan, L., Rojas, S., Aponte, C., Quiñones, M., Vilardy, S., Jaramillo, U., 2015. Colombia y su naturaleza anfibia. El entramado anfibio. In: Jaramillo, U., Cortes, J., Flóres, C. (Eds.), Colombia Anfibia. Un País de Humedales. Instituto de Investigación de Recursos Biológicos Alexander von Humboldt, Bogotá D.C., p. 140
Galfati, I., Bilal, E., Sassi, A., Abdallah, H., Zaïer, A., 2011. Accumulation of heavy metals in native plants growing near the phosphate treatment industry, Tunisia. Carpath. J. Earth. Env. 6, 85–100.
García, S., García, D., Quijano, M., Bonilla, M., 2011. Arsenic and heavy metal uptake and accumulation in native plant species from soils polluted by mining activities. Water Air Soil Poll. 223, 559–572. http://dx.doi.org/10.1007/s11270-011-0882-x.
Garg, J., 2015. Wetland assessment, monitoring and management in India using geospatial techniques. J. Environ. Manag. 148, 112–123. http://dx.doi.org/10.1016/j.jenvman.2013.12.018.
Garnier, J., Quantin, C., Martins, E., Becquer, T., 2006. Solid speciation and availability of chromium in ultramafic soils from Niquel??ndia. Braz. J. Geochem. Explor 88, 206–209. http://dx.doi.org/10.1016/j.gexplo.2005.08.040.
Ghaderian, S., Mohtadi, A., Rahiminejad, M., Baker, A., 2007. Nickel and other metal uptake and accumulation by species of Alyssum (Brassicaceae) from the ultramafics of Iran. Environ. Pollut. 145, 293–298. http://dx.doi.org/10.1016/j.envpol.2006.03.016.
Ghosh, M., Singh, S., 2005a. A comparative study of cadmium phytoextraction by accumulator and weed species. Environ. Pollut. 133, 365–371. http://dx.doi.org/10.1016/j.envpol.2004.05.015.
Ghosh, M., Singh, S., 2005b. A review on phytoremediation of heavy metals and utilization of it's byproducts. Appl. Ecol. Env. Res. 3, 1–18.
Grayman, W., Loucks, W., Saito, L., 2012. Visions of Green Technologies in 2050 for Municipal Resource Management, in: toward a Sustainable Water Future. American Society of Civil Engineers, Environmental and Water Resources Institute (U.S.), p. 386. http://dx.doi.org/10.1061/9780784412077.
Grčman, H., Velikonja-Bolta, Š., Vodnik, D., Kos, B., Leštan, D., 2001. EDTA enhanced heavy metal phytoextraction: metal accumulation, leaching and toxicity. Plant Soil 235, 105–114. http://dx.doi.org/10.1023/A:1011857303823.
Haque, N., Peralta-Videa, J., Jones, G., Gill, T., Gardea-Torresdey, J., 2008. Screening the phytoremediation potential of desert broom (Baccharis sarothroides Gray) growing on mine tailings in Arizona, USA. Environ. Pollut. 153, 362–368. http://dx.doi.org/10.1016/j.envpol.2007.08.024.
Hassan, Z., Aarts, M., 2011. Opportunities and feasibilities for biotechnological improvement of Zn, Cd or Ni tolerance and accumulation in plants. Environ. Exp. Bot. 72, 53–63. http://dx.doi.org/10.1016/j.envexpbot.2010.04.003.
Henry, J., 2000. Overview of the Phytoremediation of Lead and Mercury. EPA, Washington, D.C.
Horne, A., 2000. Phytoremediation by constructed wetlands. In: Terry, N., Bañuelos, G. (Eds.), Phytoremediation of Contaminated Soils and Waters. CRC Press LLC, Boca Raton, FL, USA, pp. 25–51 doi:10.1.1.467.809.
Hu, R., Sun, K., Su, X., Pan, Y., Zhang, Y., Wang, X., 2012. Physiological responses and tolerance mechanisms to Pb in two xerophils: Salsola passerina Bunge and Chenopodium album L. J. Hazard. Mater 205–206, 131–138. http://dx.doi.org/10.1016/j.jhazmat.2011.12.051.
Jaffré, T., Brooks, R., Lee, J., Reeves, R., 1976. Sebertia acuminata: a hyperaccumulator of nickel from New Caledonia. Sci. (80-.) 193, 579–580.
Johnston, W., Proctor, J., 1977. A comparative study of metal levels in plants from two contrasting lead-mine sites. Plant Soil 46, 251–257. http://dx.doi.org/10.1007/BF00693131.
Kamal, M., Ghaly, A., Mahmoud, N., CoteCôté, R., 2004. Phytoaccumulation of heavy metals by aquatic plants. Environ. Int. 29, 1029–1039. http://dx.doi.org/10.1016/S0160-4120(03)00091-6.
Kardanpour, Z., Jacobsen, O.S., Esbensen, K.H., 2015. Local versus field scale soil heterogeneity characterization – a challenge for representative sampling in pollution studies. SOIL 1, 695–705. http://dx.doi.org/10.5194/soil-1-695-2015.
Kasperek, G., 2008. Eine Bibliographie zur Klassifikation von Anthropochoren. Braunschweiger Geobot. Arb.
Keesstra, S.D., Bouma, J., Wallinga, J., Tittonell, P., Smith, P., Cerdà, A., Montanarella, L., Quinton, J.N., Pachepsky, Y., van der Putten, W.H., Bardgett, R.D., Moolenaar, S., Mol, G., Jansen, B., Fresco, L.O., 2016. The significance of soils and soil science towards realization of the United Nations Sustainable Development Goals. SOIL 2, 111–128. http://dx.doi.org/10.5194/soil-2-111-2016.
Kidd, P., Becerra, C., Garcia, M., Monterroso, C., 2007. Aplicación de plantas hiperacumuladoras de níquel en la fitoextracción natural: el género Alyssum L. Ecosistemas 16, 1–18.
Kohler, E., Poole, V., Reicher, Z., Turco, R., 2004. Nutrient, metal, and pesticide removal during storm and nonstorm events by a constructed wetland on an urban golf course. Ecol. Eng. 23, 285–298. http://dx.doi.org/10.1016/j.ecoleng.2004.11.002.
Kowarik, I., 2008. On the role of alien species in urban flora and vegetation. In: Marzluff, J.M., Shulenberger, E., Endlicher, W., Alberti, M., Bradley, G., Ryan, C., Simon, U., ZumBrunnen, C. (Eds.), Plant Invasions: General Aspects and Special Problems. Springer US, Boston, MA, pp. 321–338. http://dx.doi.org/10.1007/978-0-387-73412-5.
Kowarik, I., Pyšek, P., 2012. The first steps towards unifying concepts in invasion ecology were made one hundred years ago: revisiting the work of the Swiss botanist Albert Thellung. Divers. Distrib. 18, 1243–1252. http://dx.doi.org/10.1111/ddi.12009.
Ladislas, S., Gérente, C., Chazarenc, F., Brisson, J., Andrès, Y., 2014. Floating treatment wetlands for heavy metal removal in highway stormwater ponds. Ecol. Eng. 80, 85–91. http://dx.doi.org/10.1016/j.ecoleng.2014.09.115.
Lago, M., Arenas, D., Rodríguez, A., Andrade, M., Vega, F., 2015. Cobalt, chromium and nickel contents in soils and plants from a serpentinite quarry. Solid Earth 6, 323–335. http://dx.doi.org/10.5194/se-6-323-2015.
Lambrechts, T., Lequeue, G., Lobet, G., Godin, B., Bielders, C., Lutts, S., 2014. Comparative analysis of Cd and Zn impacts on root distribution and morphology of Lolium perenne and Trifolium repens: implications for phytostabilization. Plant Soil 376, 229–244. http://dx.doi.org/10.1007/s11104-013-1975-7.
Lasat, M., 2002. Phytoextraction of toxic metals. J. Environ. Qual. 31, 109–120.
Lewis, W., 1995. Wetlands: Characteristics and Boundaries. National Academies Press, Washington, D.C. doi:10.17226/4766.
Li, G., Hu, N., Ding, D., Zheng, J., Liu, Y., Wang, Y., Nie, X., 2011. Screening of plant species for phytoremediation of uranium, thorium, barium, nickel, strontium and lead contaminated soils from a uranium mill tailings repository in South China. Bull. Environ. Contam. Toxicol. 86, 646–652. http://dx.doi.org/10.1007/s00128-011-0291-2.
Litter, M., Alarcón, M., Arenas, M., Armienta, M., Avilés, M., Cáceres, R., Cipriani, H., Cornejo, L., Dias, L., Cirelli, A., Farfán, E., Garrido, S., Lorenzo, L., Morgada, M., Olmos, M., Pérez, A., 2012. Small-scale and household methods to remove arsenic from water for drinking purposes in Latin America. Sci. Total Environ. 429, 107–122. http://dx.doi.org/10.1016/j.scitotenv.2011.05.004.
Lodge, D., 1993. Biological invasions: Lessons for ecology. Trends Ecol. Evol. 8, 133–137. http://dx.doi.org/10.1016/0169-5347(93)90025-K.
Lone, M., He, Z., Stoffella, P., Yang, X., 2008. Phytoremediation of heavy metal polluted soils and water: progresses and perspectives. J. Zhejiang Univ-SC B 9, 210–220. http://dx.doi.org/10.1631/jzus.B0710633.
Lowe, S., Browne, M., Boudjelas, S., De Poorter, M., 2000. 100 of the World's Eorst Invasive Alien Species: a Selection from the Global Invasice Species Database. The Invasive Species Specialist Group (ISSG) a Specialist Group of the Species Survival Commission (SSC) of the World Conservation Union (IUCN), New Zealand.
Lozano, W., 2011. Elementos de Recuperación de Ríos Urbanos. Caso de estudio: Río Torca (Tramo 1), Bogotá D.C. (Bogotá, D. C.: La Secretaría, Bogotá D.C., Colombia).
Macnair, M., Tilstone, G., Smith, S., 1999. The Genetics of metal tolerance and accumulation in higher plants. In: Terry, N., Bañuelos, G. (Eds.), Phytoremediation of Contaminated Soil and Water. CRC Press LLC, Boca Raton, FL, USA, pp. 241–256 doi:10.1.1.467.809.
Malawska, M., Wiołkomirski, B., 2001. An analysis of soil and plant (Taraxacum Officinale) contamination with heavy metals and polycyclic aromatic hydrocarbons (PAHs) in the area of the railway junction iława Główna. Pol. Water Air Soil Poll. 127, 339–349. http://dx.doi.org/10.1023/A:1005236016074.
Maleri, R., Reinecke, S., Mesjasz, J., Reinecke, A., 2007. Growth and reproduction of earthworms in ultramafic soils. Arch. Environ. Contam. Toxicol. 52, 363–370. http://dx.doi.org/10.1007/s00244-005-0132-6.
Maobe, M., Gatebe, E., Gitu, L., Rotich, H., 2012. Profile of heavy metals in selected medicinal plants used for the treatment of diabetes, malaria and pneumonia in Kisii Region, Southwest Kenya. Glob. J. Pharmacol. 6, 245–251. http://dx.doi.org/10.5829/idosi.gjp.2012.6.3.65128.
Marchiol, L., Fellet, G., Perosa, D., Zerbi, G., 2007. Removal of trace metals by Sorghum bicolor and Helianthus annuus in a site polluted by industrial wastes: a field experience. Plant Physiol. Biochem. 45, 379–387. http://dx.doi.org/10.1016/j.plaphy.2007.03.018.
Marrero, J., Amores, I., Coto, O., 2012. Fitorremediación, una tecnología que involucra a plantas y microorganismos en el saneamiento ambiental, ICIDCA. Sobre los Derivados de la Caña de Azúcar. La Habana, Cuba. ISSN: 0138–6204.
Mataba, G., Verhaert, V., Blust, R., Bervoets, L., 2016. Distribution of trace elements in the aquatic ecosystem of the Thigithe river and the fish Labeo victorianus in Tanzania and possible risks for human consumption. Sci. Total Environ. 547, 48–59. http://dx.doi.org/10.1016/j.scitotenv.2015.12.123.
Mateos, E., 2013. Interpopulation responses to metal pollution: metal tolerance in wetland plants. In: Gupta, D., Corpas, F., Palma, J. (Eds.), Heavy Metal Stress in Plants. Springer-Verlag Berlin Heidelberg, Berlin, Heidelberg, pp. 149–162. http://dx.doi.org/10.1007/978-3-642-38469-1.

Matthews, D., Moran, B., Otte, M., 2005. Screening the wetland plant species Alisma plantago-aquatica, Carex rostrata and Phalaris arundinacea for innate tolerance to zinc and comparison with Eriophorum angustifolium and Festuca rubra Merlin. Environ. Pollut. 134, 343–351. http://dx.doi.org/10.1016/j.envpol.2004.07.023.
MAVDT, 2003. Metodologías para la Valoración Económica de Bienes, Servicios Ambientales y Recursos Naturales. Bogotá D.C., Colombia, Colombia.
Mays, P., Edwards, G., 2001. Comparison of heavy metal accumulation in a natural wetland and constructed wetlands receiving acid mine drainage. Ecol. Eng. 16, 487–500. http://dx.doi.org/10.1016/S0925-8574(00)00112-9.
McGrath, S., Zhao, F., 2003. Phytoextraction of metals and metalloids from contaminated soils. Curr. Opin. Biotech. 14, 277–282. http://dx.doi.org/10.1016/S0958-1669(03)00060-0.
McNeely, J.A., Mooney, H.A., Neville, L.E., Schei, P., Waage, J.K. (Eds.), 2001. Global Strategy on Invasive Alien Species. IUCN, Gland, Switzerland and Cambridge, UK in Collaboration with the Global Invasive Species Programme 2001 IUCN on Behalf of the Global Invasive Species Programme, Switzerland, and Cambridge, UK.
Mench, M., Vangronsveld, J., Clijsters, H., Lepp, N., Edwards, R., 2000. In situ metal immobilization and phytostabilization of contaminated soils. In: Terry, N., Bañuelos, G. (Eds.), Phytoremediation of Contaminated Soils and Waters. CRC Press LLC, Boca Raton, FL, USA, pp. 328–363 doi:10.1.1.467.809.
Menezes, A., Da Silva, J., Rossato, R., Santos, M., Decker, N., Da Silva, F., Cruz, C., Dihl, R., Lehmann, M., Ferraz, A., 2015. Genotoxic and biochemical changes in Baccharis trimera induced by coal contamination. Ecotoxicol. Environ. Saf. 114, 9–16. http://dx.doi.org/10.1016/j.ecoenv.2015.01.001.
Mengoni, A., Schat, H., Vangronsveld, J., 2010. Plants as extreme environments? Ni-resistant bacteria and Ni-hyperaccumulators of serpentine flora. Plant Soil 331, 5–16. http://dx.doi.org/10.1007/s11104-009-0242-4.
Mesjasz, J., Balkwill, K., Przybyłowicz, W., Annegarn, H., 1994. Proton microprobe and X-ray fluorescence investigations of nickel distribution in serpentine flora from South Africa. Nucl. Instrum. Methods Phys. Res. Sect. B 89, 208–212. http://dx.doi.org/10.1016/0168-583X(94)95174-8.
Mganga, N., Manoko, M., Rulangaranga, Z., 2011. Classification of plants according to their heavy metal content around North Mara Gold Mine, Tanzania: implication for phytoremediation. Tanz. J. Sci. 37, 109–119.
Millennium Ecosystem Assessment, 2005. Ecosystems and Human Well-being: Synthesis, Ecosystems. Island Press, Washington D.C. http://dx.doi.org/10.1196/annals.1439.003.
Mitsch, W., Gosselink, J., 2015a. Wetlands: human use and science. In: Wetlands. John Wiley & Sons, Inc., Hoboken, New Jersey, pp. 3–26.
Mitsch, W., Gosselink, J., 2015b. Wetlands of the world. In: Wetlands. John Wiley & Sons, Inc., Hoboken, New Jersey, pp. 45–110.
Mitsch, W., Gosselink, J., 2000. The value of wetlands: importance of scale and landscape setting. Ecol. Econ. 35, 25–33. http://dx.doi.org/10.1016/S0921-8009(00)00165-8.
MMA, 2002. Política Nacional para humedales interiores de Colombia, first ed. Ministerio del Medio Ambiente, Bogotá D.C., Colombia, Colombia. Estrategias para su conservación y uso sostenible.
Moalla, S., Awadallah, R., Rashed, M., Soltan, M., 1998. Distribution and chemical fractionation of some heavy metals in bottom sediments of Lake Nasser. Hydrobiologia 364, 31–40. http://dx.doi.org/10.1023/A:1003112008122.
Monferrán, M., Wunderlin, D., 2013. Biochemistry of metals/metalloids toward remediation process magdalena. In: Gupta, D., Corpas, F., Palma, J. (Eds.), Heavy Metal Stress in Plants. Springer-Verlag Berlin Heidelberg, Berlin, Heidelberg, pp. 43–72. http://dx.doi.org/10.1007/978-3-642-38469-1.
Montes, C., Rendón, M., Varela, L., Cappa, M.J., 2007. Manual de restauración de humedales mediterráneos. Sevilla.
Moreira, H., Marques, A., Rangel, A., Castro, P., 2011. Heavy metal accumulation in plant species indigenous to a contaminated portuguese site: prospects for phytoremediation. Water Air Soil Poll. 221, 377–389. http://dx.doi.org/10.1007/s11270-011-0797-6.
Murch, S., Haq, K., Rupasinghe, H., Saxena, P., 2003. Nickel contamination affects growth and secondary metabolite composition of St. John's wort (Hypericum perforatum L.). Environ. Exp. Bot. 49, 251–257. http://dx.doi.org/10.1016/S0098-8472(02)00090-4.
Nedelkoska, T., Doran, P., 2000. Characteristics of heavy metal uptake by plant specie with potential for phytoremediation and phytomining. Min. Eng. 13, 549–561. http://dx.doi.org/10.1016/S0892-6875(00)00035-2.
O'Dell, R., Silk, W., Green, P., Claassen, V., 2007. Compost amendment of Cu-Zn minespoil reduces toxic bioavailable heavy metal concentrations and promotes establishment and biomass production of Bromus carinatus (Hook and Arn.). Environ. Pollut. 148, 115–124. http://dx.doi.org/10.1016/j.envpol.2006.10.037.
Ojasti, J., 2001. Estudio sobre el estado actual de las especies exóticas, Proyecto Estrategia Regional de Biodiversidad para los Países del Trópico Andino. Caracas-Venezuela.
Olko, A., Abratowska, A., Zyłkowska, J., Wierzbicka, M., Tukiendorf, A., 2008. Armeria maritima from a calamine heap-Initial studies on physiologic-metabolic adaptations to metal-enriched soil. Ecotoxicol. Environ. Saf. 69, 209–218. http://dx.doi.org/10.1016/j.ecoenv.2007.01.010.
Page, V., Bayon, R., Feller, U., 2006. Partitioning of zinc, cadmium, manganese and cobalt in wheat (Triticum aestivum) and lupin (Lupinus albus) and further release into the soil. Environ. Exp. Bot. 58, 269–278. http://dx.doi.org/10.1016/j.envexpbot.2005.09.005.
Pastor, J., Hernández, A., 2002. Estudio de suelos de vertederos sellados y de sus especies vegetales espontáneas para la fitorrestauración de suelos degradados y contaminados del centro de España. In Anales de biología. Toxicology 24, 145–153.
Paz, J., Lu, H., Fu, S., Méndez, A., Gascó, G., 2014. Use of phytoremediation and biochar to remediate heavy metal polluted soils: a review. Solid Earth 5, 65–75. http://dx.doi.org/10.5194/se-5-65-2014.
Pfeifer, H., Derron, M., Rey, D., Schlegel, C., Atteia, O., Piazza, R., Dubois, J.-P., Mandia, Y., 2000. Natural trace element input to the soil-sediment-water-plant system: examples of background and contaminated situations in Switzerland, Eastern France and Northern Italy. In: Markert, B., Friese, K. (Eds.), Trace Elements - Their Distribution and Effects in the Environment, Trace Metals in the Environment. Elsevier, pp. 33–86. http://dx.doi.org/10.1016/S0927-5215(00)80005-3.
Pilon, E., Zayed, A., DeSouza, M., Lin, Z., Terry, N., 2000. Remediation of selenium-polluted soils and waters by phytovolatilization. In: Terry, N., Bañuelos, G. (Eds.), Phytoremediation of Contaminated Soils and Waters. CRC Press LLC, Boca Raton, FL, USA, pp. 72–94 doi:10.1.1.467.809.
Pimentel, D., McNair, S., Janecka, J., Wightman, J., Simmonds, C., O'Connell, C., Wong, E., Russel, L., Zern, J., Aquino, T., Tsomondo, T., 2001. Economic and environmental threats of alien plant, animal, and microbe invasions. Agric. Ecosyst. Environ. 84, 1–20. http://dx.doi.org/10.1016/S0167-8809(00)00178-X.
Pollard, A., Baker, A., 1997. Deterrence of herbivory by zinc hyperaccumulation in Thlaspi caerulescens (Brassicaceae). New Phytol. 135, 655–658. http://dx.doi.org/10.1046/j.1469-8137.1997.00689.
Pollard, A., Reeves, R., Baker, A., 2014. Facultative hyperaccumulation of heavy metals and metalloids. Plant Sci. 217–218, 8–17. http://dx.doi.org/10.1016/j.plantsci.2013.11.011.
Poschenrieder, C., Tolrà, R., Barceló, J., 2006. Can metals defend plants against biotic stress? Trends Plant Sci. 11, 288–295. http://dx.doi.org/10.1016/j.tplants.2006.04.007.
Prasad, M., 2004. Phytoremediation of metals in the environment for sustainable development. Proc. Indian Natn. Sci. Acad. B70 70, 71–98.
Prasad, M., 2003. Phytoremediation of metal-polluted ecosystems: hype for commercialization. Russ. J. Plant. Physl+ 50, 686–701. http://dx.doi.org/10.1023/A:1025604627496.
Prasad, M., Freitas, H., 2003. Metal hyperaccumulation in plants: biodiversity prospecting for phytoremediation technology. Electron. J. Biotechnol. 6, 285–321. http://dx.doi.org/10.4067/S0717-34582003000300012.
Proctor, J., 1999. Toxins, nutrient shortages and droughts: the serpentine challenge. Trends Ecol. Evol. 14, 334–335. http://dx.doi.org/10.1016/S0169-5347(99)01698-5.
Pyšek, P., 1995. On the terminology used in plant invasion studies. In: Pyšek, P., Prach, K., Rejmánek, M., Wade, M. (Eds.), Plant Lnvasions - General Aspects and Special Problems. Academic Publishing, Amsterdam. The Netherlands, pp. 71–81.
Ramamurthy, A., Memarian, R., 2012. Phytoremediation of mixed soil contaminants. Water Air Soil Poll. 223, 511–518. http://dx.doi.org/10.1007/s11270-011-0878-6.
Rascio, N., Navari, F., 2011. Heavy metal hyperaccumulating plants: how and why do they do it? And what makes them so interesting? Plant Sci. 180, 169–181. http://dx.doi.org/10.1016/j.plantsci.2010.08.016.
Raskin, I., 1996. Plant genetic engineering may help with environmental cleanup. Proc. Natl. Acad. Sci. 93, 3164–3166.
Raskin, I., Smith, R., Salt, D., 1997. Phytoremediation of metals: using plants to remove pollutants from the environment. Curr. Opin. Biotech. 8, 221–226. http://dx.doi.org/10.1016/S0958-1669(97)80106-1.
Reeves, R., Baker, A., Borhidi, A., Berazain, R., 1999. Nickel hyperaccumulation in the serpentine flora of Cuba. Ann. Bot. 83, 29–38 doi:0305-7364?99?010029?10$30.00?0.
Reeves, R., Baker, A., Romero, R., 2007. The ultramafic flora of the Santa Elena peninsula, Costa Rica: a biogeochemical reconnaissance. J. Geochem. Explor. 93, 153–159. http://dx.doi.org/10.1016/j.gexplo.2007.04.002.
Ricaurte, L., Patiño, J., Arias, J., Acevedo, O., Restrepo, D., Jaramillo, U., Flórez, C., Estupiñán, L., 2015. La pluralidad del agua, tipos de humedales de Colombia - Sistema de clasificación de humedales. In: Jaramillo, U., Cortes, J., Flóres, C. (Eds.), Colombia Anfibia. Un País de Humedales. Instituto de Investigación de Recursos Biológicos Alexander von Humboldt, Bogotá D.C, p. 140.
Robinson, B., Anderson, C., Dickinson, N., 2015. Phytoextraction: where's the action? J. Geochem. Explor 151, 34–40. http://dx.doi.org/10.1016/j.gexplo.2015.01.001.
Robinson, B., Chiarucci, A., Brooks, R., Petit, D., Kirkman, J., Gregg, P., De Dominicis, V., 1997. The nickel hyperaccumulator plant Alyssum bertolonii as a potential agent for phytoremediation and phytomining of nickel. J. Geochem. Explor 59, 75–86. http://dx.doi.org/10.1016/S0375-6742(97)00010-1.
Robinson, B., Kim, N., Marchetti, M., Moni, C., Schroeter, L., Van Den Dijssel, C., Milne, G., Clothier, B., 2006. Arsenic hyperaccumulation by aquatic macrophytes in the taupo volcanic zone, New Zealand. Environ. Exp. Bot. 58, 206–215. http://dx.doi.org/10.1016/j.envexpbot.2005.08.004.
Robinson, B.H., Green, S.R., Chancerel, B., Mills, T.M., Clothier, B.E., 2007. Poplar for the phytomanagement of boron contaminated sites. Environ. Pollut. 150, 225–233. http://dx.doi.org/10.1016/j.envpol.2007.01.017.
Rodríguez, D., 2003. El papel de los microorganismos en la biodegradación de compuestos tóxicos. Ecosistemas 12. http://dx.doi.org/10.7818/re.2014.12-2.00.
Rodríguez, S., 2009. Organismos del suelo: la dimensión invisible de las invasiones por plantas no nativas. Ecosistemas. http://dx.doi.org/10.7818/re.2014.18-2.00.
Sadowsky, M.J., 1999. Phytoremediation: past promises and future practises. In:

Proceedings of the 8th International Symposium on Microbial Ecology. Halifax, Canada.
Samecka, A., Kempers, A., 2001. Concentrations of heavy metals and plant nutrients in water, sediments and aquatic macrophytes of anthropogenic lakes (former open cut brown coal mines) differing in stage of acidification. Sci. Total Environ. 281, 87–98. http://dx.doi.org/10.1016/S0048-9697(01)00838-5.
Sánchez, M., 2010. Contaminacion por metales pesados en el botadero de basuras de Moravia en Medellin: transferencia a flora y fauna y evaluacion del potential fitorremediador de especies nativas e introducidas. Pontificia Universidad Javeriana.
Santos, J., Castro, A., Huezo, J., Torres, L., 2012. Arsenic and heavy metals in native plants at tailings impoundments in Queretaro, Mexico. Phys. Chem. Earth 37–39, 10–17. http://dx.doi.org/10.1016/j.pce.2011.12.002.
Santos, S., Cabanas, J., Pereira, J., 2007. Abundance and diversity of soil arthropods in olive grove ecosystem (Portugal): effect of pitfall trap type. Eur. J. Soil. Biol. 43, 77–83. http://dx.doi.org/10.1016/j.ejsobi.2006.10.001.
Sarma, H., 2011. Metal hyperaccumulation in plants: a review focusing on phytoremediation technology. J. Environ. Sci. Technol. 4, 118–138. http://dx.doi.org/10.3923/jest.2011.118.138.
Schaller, J., Vymazal, J., Brackhage, C., 2013. Retention of resources (metals, metalloids and rare earth elements) by autochthonously/allochthonously dominated wetlands: a review. Ecol. Eng. 53, 106–114. http://dx.doi.org/10.1016/j.ecoleng.2012.12.021.
Schroeder, F., 1968. Zur klassifizierung der Anthropochoren. Veg. Acta Geobot. 16, 225–238. http://dx.doi.org/10.1007/BF00257018.
Shallari, S., Schwartz, C., Hasko, A., Morel, J., 1998. Heavy metals in soils and plants of serpentine and industrial sites of Albania. Sci. Total Environ. 209, 133–142. http://dx.doi.org/10.1016/S0048-9697(98)80104-6.
Sharma, R., Agrawal, M., 2005. Biological effects of heavy metals: an overview. J. Environ. Biol. 26, 301–313. ISSN: 0254-8704.
Sheoran, V., Sheoran, A., Poonia, P., 2009. Phytomining: a review. Min. Eng. 22, 1007–1019. http://dx.doi.org/10.1016/j.mineng.2009.04.001.
Shinmachi, F., Kumanda, Y., Noguchi, A., Hasegawa, I., 2003. Translocation and accumulation of cadmium in cadmium-tolerant Polygonum thunbergii. Soil. Sci. Plant Nutr. 49, 355–361. http://dx.doi.org/10.1080/00380768.2003.10410020.
Silk, W., Bambic, D., O'Dell, R., Green, P., 2006. Seasonal and spatial patterns of metals at a restored copper mine site II. Copper in riparian soils and Bromus carinatus shoots. Environ. Pollut. 144, 783–789. http://dx.doi.org/10.1016/j.envpol.2006.02.018.
Smith, P., Cotrufo, M.F., Rumpel, C., Paustian, K., Kuikman, P.J., Elliott, J.A., McDowell, R., Griffiths, R.I., Asakawa, S., Bustamante, M., House, J.I., Sobocká, J., Harper, R., Pan, G., West, P.C., Gerber, J.S., Clark, J.M., Adhya, T., Scholes, R.J., Scholes, M.C., 2015. Biogeochemical cycles and biodiversity as key drivers of ecosystem services provided by soils. SOIL 1, 665–685. http://dx.doi.org/10.5194/soil-1-665-2015.
Srivastava, S., Bhargava, A., 2015. Genetic diversity and heavy metal stress in plants. In: Ahuja, M., Jain, S. (Eds.), Genetic Diversity and Erosion in Plants, Sustainable Development and Biodiversity. Springer International Publishing, Cham, pp. 189–225. http://dx.doi.org/10.1007/978-3-319-25637-5.
Stravinskienė, V., Račaitė, M., 2014. Impact of cadmium and zinc on the growth of white clover (Trifolium repens L.) shoots and roots. Pol. J. Environ. Stud. 23, 1355–1359.
Sun, Y., Zhou, Q., Liu, W., An, J., Xu, Z., Wang, L., 2009a. Joint effects of arsenic and cadmium on plant growth and metal bioaccumulation: a potential Cd-hyperaccumulator and As-excluder Bidens pilosa L. J. Hazard. Mater. 165, 1023–1028. http://dx.doi.org/10.1016/j.jhazmat.2008.10.097.
Sun, Y., Zhou, Q., Wang, L., Liu, W., 2009b. Cadmium tolerance and accumulation characteristics of Bidens pilosa L. as a potential Cd-hyperaccumulator. J. Hazard. Mater. 161, 808–814. http://dx.doi.org/10.1016/j.jhazmat.2008.04.030.
Susarla, S., Medina, V., McCutcheon, S., 2002. Phytoremediation: an ecological solution to organic chemical contamination. Ecol. Eng. 18, 647–658. http://dx.doi.org/10.1016/S0925-8574(02)00026-5.
Switras, S., 1999. The potential of phytoremediation techniques for selenium removal. J. Environ. Qual. 5, 1–7.
Tanhan, P., Kruatrachue, M., Pokethitiyook, P., Chaiyarat, R., 2007. Uptake and accumulation of cadmium, lead and zinc by Siam weed [Chromolaena odorata (L.) King & Robinson]. Chemosphere 68, 323–329. http://dx.doi.org/10.1016/j.chemosphere.2006.12.064.
Ten Brink, P., Russi, D., Farmer, A., Badura, T., Coates, D., Förster, J., Kumar, R., Davidson, N., 2013. La Economía de los Ecosistemas y la Biodiversidad relativa al agua y los humedales. Resumen ejecutivo.
Terry, N., Bañuelos, G. (Eds.), 2000. Phytoremediation of Contaminated Soils and Waters. CRC Press LLC, Boca Raton, FL, USA doi:10.1.1.467.809.
Thangavel, P., Shahira, A., Subburam, V., 1999. Interactive effects of selenium and mercury on the restoration potential of leaves of the medicinal plant. Portulaca oleracea Linn. Sci. Total Environ. 243–244, 1–8. http://dx.doi.org/10.1016/S0048-9697(98)00181-8.
Thangavel, P., Subburam, V., 1998. Effect of trace metals on the restoration potential of leaves of the medicinal plant. Portulaca oleracea Linn. Biol. Trace Elem. Res. 61, 313–321. http://dx.doi.org/10.1007/BF02789091.
Tilman, D., Lehman, C., 2001. Human-caused environmental change: impacts on plant diversity and evolution. PNAS 98, 5433–5440. http://dx.doi.org/10.1073/pnas.091093198.
Tirillini, B., Ricci, A., Pintore, G., Chessa, M., Sighinolfi, S., 2006. Induction of hypericins in Hypericum perforatum in response to chromium. Fitoterapia 77, 164–170. http://dx.doi.org/10.1016/j.fitote.2006.01.011.
UICN, 2000. Guías para la preservación de pérdidas de diversidad biológica ocasionadas por especies exóticas invasoras.
USGS, 2010. Mineral Commodity Summaries 2010. U.S. Geological Survey, Washington D.C. USA.
Valles, M., Alarcón, M., 2014. Retención de arsénico en humedales construidos con Eleocharis macrostachya y Schoenoplectus americanus. Rev. Int. Contam. Ambient. 30, 143–148.
Van der Ent, A., Baker, A., Reeves, R., Pollard, A., Schat, H., 2013a. Hyperaccumulators of metal and metalloid trace elements: facts and fiction. Plant Soil 362, 319–334. http://dx.doi.org/10.1007/s11104-012-1287-3.
Van der Ent, A., Baker, A., Van Balgooy, M., Tjoa, A., 2013b. Ultramafic nickel laterites in Indonesia (Sulawesi, Halmahera): mining, nickel hyperaccumulators and opportunities for phytomining. J. Geochem. Explor 128, 72–79. http://dx.doi.org/10.1016/j.gexplo.2013.01.009.
Vanek, T., Podlipna, R., Soudek, P., 2010. General factors influencing application of phytotchnology techniques. In: Kulakow, P.A., Pidlisnyuk, V.V. (Eds.), Application of Phytotechnologies for Cleanup of Industrial, Agricultural, and Wastewater Contamination, NATO Science for Peace and Security Series C: Environmental Security. Springer, Netherlands, pp. 1–13. http://dx.doi.org/10.1007/978-90-481-3592-9. Dordrecht.
Vardanyan, L., Ingole, B., 2006. Studies on heavy metal accumulation in aquatic macrophytes from Sevan (Armenia) and Carambolim (India) lake systems. Environ. Int. 32, 208–218. http://dx.doi.org/10.1016/j.envint.2005.08.013.
Vargas, O., Martínez, M., Díaz, A., 2012. Protocolo de Propagación de Plantas Hidrófilas y Manejo de Viveros para la Rehabilitación Ecológica de los Parques Ecológicos Distritales de Humedal. Grupo de Restauración Ecológica de la Universidad Nacional de Colombia y Secretaría Distrital de Ambiente, Bogotá D.C., Colombia.
Vassilev, A., Schwitzguebel, J., Thewys, T., Van der Lelie, D., Vangronsveld, J., 2004. The use of plants for remediation of metal-contaminated soils. ScientificWorldJournal 4, 9–34. http://dx.doi.org/10.1100/tsw.2004.2.
Vigo, J., 1976. Sobre algunas plantas aloctonas. Collect. Bot. 10, 351–364.
Villaronga, B., 2003. Una amenaza para la biodiversidad: especies exóticas invasoras. Ambient. Rev. Minist. Medio Ambient. 23, 58–66.
Walker, D., Clemente, R., Bernal, M., 2004. Contrasting effects of manure and compost on soil pH, heavy metal availability and growth of Chenopodium album L. in a soil contaminated by pyritic mine waste. Chemosphere 57, 215–224. http://dx.doi.org/10.1016/j.chemosphere.2004.05.020.
Wang, Q., Cui, Y., Dong, Y., 2002. Phytoremediation of polluted waters potentials and prospects of wetland plants. Acta. Biotechnol. 22, 199–208. http://dx.doi.org/10.1002/1521-3846(200205)22:1/2<199::AID-ABIO199>3.0.CO;2-T.
Wei, C., Wang, C., Yang, L., 2009. Characterizing spatial distribution and sources of heavy metals in the soils from mining-smelting activities in Shuikoushan, Hunan Province, China. J. Environ. Sci. 21, 1230–1236. http://dx.doi.org/10.1016/S1001-0742(08)62409-2.
Wei, S., Teixeira da Silva, J., Zhou, Q., 2008. Agro-improving method of phytoextracting heavy metal contaminated soil. J. Hazard. Mater. 150, 662–668. http://dx.doi.org/10.1016/j.jhazmat.2007.05.014.
Welcomme, R., 1988. International Introductions of Inland Aquatic Species.
Wenzel, W., Jockwer, F., 1999. Accumulation of heavy metals in plants grown on mineralised soils of the Austrian Alps. Environ. Pollut. 104, 145–155. http://dx.doi.org/10.1016/S0269-7491(98)00139-0.
Xiong, Z., 1997. Bioaccumulation and physiological effects of excess lead in a roadside pioneer species Sonchus oleraceus L. Environ. Pollut. 97, 275–279. http://dx.doi.org/10.1016/S0269-7491(97)00086-9.
Xue, L., Liu, J., Shi, S., Wei, Y., Chang, E., Gao, M., Chen, L., Jiang, Z., 2014. Uptake of heavy metals by native herbaceous plants in an antimony mine (Hunan, China). Clean. Soil Air Water 42, 81–87. http://dx.doi.org/10.1002/clen.201200490.
Yadav, S., Chandra, R., 2011. Heavy metals accumulation and ecophysiological effect on Typha angustifolia L. and Cyperus esculentus L. growing in distillery and tannery effluent polluted natural wetland site, Unnao, India. Environ. Earth Sci. 62, 1235–1243. http://dx.doi.org/10.1007/s12665-010-0611-6.
Yang, X., Feng, Y., He, Z., Stoffella, P., 2005. Molecular mechanisms of heavy metal hyperaccumulation and phytoremediation. J. Trace. Elem. Med. Biol. 18, 339–353. http://dx.doi.org/10.1016/j.jtemb.2005.02.007.
Yanqun, Z., Yuan, L., Schvartz, C., Langlade, L., Fan, L., 2004. Accumulation of Pb, Cd, Cu and Zn in plants and hyperaccumulator choice in Lanping lead-zinc mine area, China. Environ. Int. 30, 567–576. http://dx.doi.org/10.1016/j.envint.2003.10.012.
Zechmeister, H., Grodzińska, K., Szarek, G., 2003. Bioindicators & biomonitors - principles, concepts and applications. In: Markert, B.A., Breur, A.M., Zechmeister, H.G (Eds.), Bioindicators and Biomonitors, Trace Metals and Other Contaminants in the Environment. Elsevier, pp. 329–375. http://dx.doi.org/10.1016/S0927-5215(03)80140-6.
Zhang, L., Angle, J., Chaney, R., 2007. Do high-nickel leaves shed by the nickel hyperaccumulator Alyssum murale inhibit seed germination of competing plants? New Phytol. 173, 509–516. http://dx.doi.org/10.1111/j.1469-8137.2006.01952.x.
Zhang, X., Liu, J., Huang, H., Chen, J., Zhu, Y., Wang, D., 2007. Chromium accumulation by the hyperaccumulator plant Leersia hexandra Swartz. Chemosphere 67, 1138–1143. http://dx.doi.org/10.1016/j.chemosphere.2006.11.014.