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Keep and promote biodiversity at polluted sites under phytomanagement

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Abstract

The phytomanagement concept combines a sustainable reduction of pollutant linkages at risk-assessed contaminated sites with the generation of both valuable biomass for the (bio)economy and ecosystem services. One of the potential benefits of phytomanagement is the possibility to increase biodiversity in polluted sites. However, the unique biodiversity present in some polluted sites can be severely impacted by the implementation of phytomanagement practices, even resulting in the local extinction of endemic ecotypes or species of great conservation value. Here we highlight the importance of promoting measures to minimize the potential adverse impact of phytomanagement on biodiversity at polluted sites, as well as recommend practices to increase biodiversity at phytomanaged sites without compromising its effectiveness in terms of reduction of pollutant linkages and the generation of valuable biomass and ecosystem services.

Keywords: contaminated soil; metal; metallophytes; phytoremediation; trace elements.

1. Introduction

The notion of phytomanagement is based on the combination of (i) a sustainable reduction of pollutant linkages at degraded sites with (ii) the generation of valuable products and essential ecosystem services. In other words, its main purpose is to grow profitable plants to minimize pollutant-induced environmental risks while maximizing economic and/or ecological revenues. It is often claimed that one of the potential benefits of phytomanagement is the possibility to enhance biodiversity in the degraded site under recovery. Pertinently, it must be strongly emphasized that some polluted sites, most relevantly mining sites, can harbour a unique biodiversity that must be carefully preserved. In any event, protecting biodiversity is of the utmost importance as human well-being depends upon biodiversity in many different ways (Naeem et al., 2016). In consequence, under the current scenario of global change and biodiversity loss, it is crucial to use as many tools as possible to preserve the fabric of life and the natural capital on which our survival and well-being depend. Biodiversity is known to be critical for the supply of ecosystem services and, then, it is not surprising that much research effort has been directed at understanding how biodiversity impacts ecosystem functioning and resilience, and concomitantly the sustainable provision of goods and ecosystems services. This aspect has special relevance within the phytomanagement framework since, as described above, the main purpose of phytomanagement is to grow profitable plants in order to minimize pollutant-induced environmental risks while maximizing economic and/or ecological revenues in terms of products and ecosystem services. However, when implementing actions to promote such biodiversity in phytomanaged sites, in most cases, the only initiative is to enhance the number of different plant species grown for phytomanagement purposes. We must overcome such incomplete approach by widening our understanding of how the different taxonomic groups can be positively or negatively affected by phytomanagement practices. In addition, the unique biodiversity present in some polluted sites can be negatively affected by the implementation of phytomanagement practices. In this review paper, the importance of promoting (i) measures to minimize the potential adverse impact of phytomanagement on biodiversity; and (ii) practices to increase biodiversity at phytomanaged sites, is highlighted.

2. Phytomanagement: a sustainable gentle remediation option

As a result of a wide variety of anthropogenic activities and accidental spills, many soils are currently polluted with a myriad of potentially toxic compounds, such as trace elements (TEs), mineral oils, polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), pesticides, etc. Unfortunately, the remediation of polluted soils is often a very expensive, environmentally-disruptive activity, especially at large sites and/or in those soils simultaneously polluted with several contaminants inducing adverse effects on biological receptors (Agnello et al. 2016).

Opportunely, in the last decades, various Gentle Remediation Options (GROs) have been developed as more cost-effective, environmentally-friendly and aesthetically-pleasing technologies for the remediation of large areas with polluted soils from mild up to medium levels of contamination (Vangronsveld et al. 2009; Kidd et al. 2015; Mench et al. 2018). Among them, phytoremediation and phytomanagement have shown their great potential, on the long term, for the sustainable remediation of polluted sites due to their capacity to combine an effective mitigation of pollutant-induced risks with the provision of valuable plant biomass and ecosystem services (Mench et al. 2018).

The term *phytoremediation* refers to a set of sustainable phytotechnologies focused on the use of plant species to remediate polluted sites, mainly those affected by the presence of TEs via the phytoextraction or phytostabilization options, which aim at (i) decreasing the available soil TEs, through plant uptake and accumulation in the harvestable plant parts, or (ii) reducing the labile ("bioavailable") TE pool usually by combining the growth of TE-excluding plants with the application of soil amendments (Garbisu and Alkorta 2001; Alkorta et al. 2004a,b). However, the commercial application of phytoextraction has been seriously hampered by its intrinsic limitations, *e.g.* the long time required to effectively extract TEs from medium and highly polluted soils, root depth, lack of plants that can accumulate more than one or two TEs, decrease of metal(loid) market prices, etc. In turn, one constraint for the application of phytostabilization is that many risk-assessment regulations for soil remediation are still based on total soil TEs, not on their bioavailable concentrations or site-specific risk assessment. Paradoxically, the harmful effects of TEs on soil biota and, hence, soil health, are related to the sensitivity/tolerance of living organism populations and the bioavailable pool rather than total metal(loid) concentrations (Kumpiene et al. 2009, 2017), the bioavailable fraction being subject to uptake by soil organisms, leaching, and transfer to other environmental media (Madejón et al. 2006).

In any event, for effective clean-up of TE-polluted soils, the combination of different approaches, *e.g.* phytoextraction with hyperaccumulators, chelate-assisted phytoextraction, phytostabilization, microbe-assisted phytoremediation (bioaugmentation), traditional breeding and/or genetic engineering of phytoremediation plants, etc., appears necessary to increase the applicability of phytoremediation in the future (Yang et al. 2020a).

To overcome the phytoremediation limitations, the concept of *phytomanagement* evolved (Fig. 1), combining a sustainable reduction in pollutant linkages with the generation of plant biomass (mainly non-food crops) and ecosystem services (Mench et al. 2009; Robinson et al. 2009; Fässler et al. 2010; Robinson and McIvor 2013; Cundy et al. 2016; Burges et al. 2018). The phytomanagement objective is to grow profitable plants to control the bioavailable pool of soil pollutants (*e.g.*, TEs), thereby minimizing environmental risks, while maximizing economic and ecological revenues (Vangronsveld et al. 2009). In this way, phytomanagement (commonly based on the interactions among plants, microorganisms and soil amendments) is often considered a “holding strategy” for vacant sites until their remediation is undertaken according to future land use (Mench et al. 2018). Most importantly, compared to other remediation technologies, the requirements of phytomanagement for chemicals and energy are much lower, as well as the total cost, making it a viable strategy for the remediation of large polluted areas (Thewys et al. 2010; Kuppens et al. 2015; Giagnoni et al. 2020).

The production of valuable plant biomass (for timber, bioenergy, biofortified products, ecomaterials, etc.) is considered essential for the commercial success of phytomanagement (Conesa et al. 2012). Energy crops, *e.g. Miscanthus* spp., *Ricinus communis* L. and *Brassica napus* L., can be grown for biofuel production (Burges et al. 2018). Other plant species can be grown for the production of biochar, raw materials (oil, paper, chemicals, essential oils, etc.), medicinal purposes, etc. (Pandey et al. 2016). Likewise, the growth of fast growing trees opens the possibility to phytoextract some metals in excess (*e.g.*, Cd, Zn, Ni, U, Cs, and Sr) while producing biomass for bioenergy and products (*e.g.*, timber, resin, adhesives, etc.). Other phytomanagement options are aimed at removing the bioavailable metal(loid) fraction, the so-called “bioavailable contaminant stripping (BCS)”, while providing ecosystem services and feedstocks for biomass-processing (bio)technologies (Herzig et al. 2014).

Since the 2010s, an increasing attention is developing on the capacity of phytomanagement options to provide a wide variety of co-benefits and ecosystem services, such as primary production, control of soil erosion, water runoff/drainage management, carbon sequestration, amenity and recreation, aesthetic value, habitat for animals and microorganisms, biodiversity, etc. (Evangelou et al. 2015; Kidd et al. 2015; Cundy et al. 2016; Simek et al. 2017; Touceda-González et al. 2017a; Xue et al. 2018). Strictly speaking, biodiversity *per se* is not an ecosystem service (Haines-Young and Potschin 2010); rather, biodiversity supports the flow of vital ecosystem services that we depend upon (in other words, biodiversity forms the biological infrastructure that supports the provision of ecosystem services). Indeed, the ecosystem services (and, concomitantly, human well-being) depend essentially on the structures and processes generated by living organisms and their interactions with, and processing of, abiotic materials (Haines-Young and Potschin 2010; IPBES 2019). Although biodiversity has intrinsic value by itself (and, then, it should be preserved in its own right), its utilitarian value has increasingly become the central focus of the debates on the need to preserve our natural capital (Chang et al. 2007; Haines and Potskin 2010).

The phytomanagement capacity to promote biodiversity in polluted sites is of key importance, as we are currently at a crucial juncture in human history, with biodiversity being lost at an accelerating pace due to an increasingly affluent human population, climate change, uncontrolled development and habitat destruction (Sandifer et al. 2015; IPBES 2019). Taking into consideration the links between biodiversity, ecosystem functioning, ecosystem services and human well-being (Cardinale et al. 2006; Naeem et al. 2016), now more than ever, the importance of promoting biodiversity must be emphasized.

3. Unique biodiversity at polluted sites

One of the recurrently mentioned potential benefits of phytomanagement is the possibility to increase biodiversity in the polluted land in question. However, the unique biodiversity often present in many polluted sites (in particular, long-term abandoned mine sites) can be severely impacted by the implementation of phytomanagement practices, even resulting in the local extinction of endemic ecotypes or species of great conservation value. Here we highlight the importance of promoting measures to minimize the potential adverse impact of phytomanagement on biodiversity at polluted sites. After all, some polluted sites, most relevantly mining sites, harbour a unique biodiversity that must be painstakingly preserved. Actually derelict soils can provide an interesting biodiversity for a variety of uses (Vincent et al. 2018).

In particular, there is a need to conserve metallophytes (*i.e.*, unique plant species that have evolved to survive on soils with high TE levels), which are nowadays increasingly under threat of extinction from mining activity (Whiting et al. 2004; Batty 2005; Baker et al. 2010; Paul et al. 2018). Indeed, regrettably, metalliferous ecosystems are presently threatened at a global scale by the growth of mining activities with concomitant extinction risks for metallophyte diversity (Whiting et al. 2004; Séleck et al. 2013).

Metallophytes are the consequence of powerful selective pressures over long evolutionary times as a result of the presence of high total soil TEs (Ginocchio and Baker 2004). The intensity and duration of the sustained evolutionary exposure to these high TE levels direct the degree of specialization of the TE resistance trait. Thus, some populations of plant species can evolve TE resistance within a few years, for example around metal smelters, if the selection pressure is high enough (Barrutia et al. 2011a). Populations of pseudometallophytes present a greater capacity to withstand phytotoxicity induced by TE excess, as compared with other populations of the same plant species from non-polluted sites (Whiting et al. 2004; Barrutia et al. 2011a). But, as the duration of TE exposure increases, the mechanisms that allow survival and growth in the presence of high TE levels become gradually more specialized, resulting in true metallophytes or eumetallophytes that have developed evolutionary mechanisms to live and thrive on metalliferous soils. As a matter of fact, true metallophytes have often diverged genetically and morphologically to form new taxa endemic to their native metalliferous soils (Barrutia et al. 2011a). Regrettably, their restricted geographic range is, partly, responsible for the current high rates of population decline or, what is worse, irreversible extinction.

Plants growing in TE-polluted sites can be classified as (1) *excluders:* these plants limit TE uptake and translocation and, then, maintain low TE concentrations in their aerial tissues; (2) *indicators:* these plants accumulate TEs in their harvestable parts at concentrations similar to those present in the polluted soil; and (3) *accumulators/hyperaccumulators:* these plants increase TE uptake, translocation and accumulation in their aboveground biomass reaching levels that far exceed those present in the polluted soil (van der Ent et al. 2013, 2015a; Malik et al. 2017; Massoura et al. 2014; Reeves et al. 2017a). Particularly, hyperaccumulators are exceptional plants that accumulate metal(loid)s in their tissues to levels that can be hundreds or thousands of times greater than common ranges in other plant species (van der Ent et al. 2013), and whose ecology is an active field of research, focusing on anti-herbivore defences, allelopathy and biotic interactions (Reeves et al. 2017b). Logically, it is important to protect not only TE hyperaccumulators as nature´s oddities but also TE excluders, indicators and accumulators.

Apart from their intrinsic value as remarkable rare species, metallophytes are suitable candidates for the revegetation of mining sites, as well as for the implementation of phytoremediation (phytoextraction/phytomining, phytostabilization) and phytomanagement initiatives (van der Ent et al. 2015b; Rosenkranz et al. 2019; Corzo Remigio et al. 2020). Thus, TE excluders and hyperaccumulators have been extensively used for phytostabilization and phytoextraction purposes, respectively (Hernández-Allica et al. 2006; Epelde et al. 2008, 2009, 2010; Barrutia et al. 2009; Pardo et al. 2014; Garaiyurrebaso et al. 2017). There is nowadays an increasing interest in the use of native plant species and populations for the revegetation of TE polluted sites, as opposed to non-native, introduced species (Parraga-Aguado et al. 2014; Chen et al. 2019). In this respect, a sturdy commitment to conservation of metallophyte biodiversity is self-evident (Whiting et al. 2002).

Then, before starting any phytomanagement initiative, it is imperative to study the native vegetation of the polluted site in search of potential candidates (*e.g.*, metallophytes) for conservation purposes. If such candidates are identified, then, an area of the site (preferably, the area where the most interesting plant species have been identified) must be left unmanaged for conservation purposes (and, if needed, protection barriers must be installed).

In addition to protecting the natural environment of valued and treasured plant species (*i.e.*, *in situ* conservation in biotope ‘‘islands’’), efforts must also be directed at conserving them *ex situ*, that is to say, in germplasm banks, seed gardens, arboreta, botanic gardens, etc. (Whiting et al. 2004), ideally maintaining the required degree of contaminant exposure (otherwise, contaminant-sensitive, non-adapted individuals might again become dominant in the plant population after a few cultivations). Additionally, when designing a strategy to preserve the valuable native vegetation at the polluted site, attention should also be paid to plant assemblages (*e.g.*, metalliferous distinctive plant communities).

Apart from the presence of unique metallophytes in TE-polluted sites, these degraded environments can also harbour a valuable microbial diversity that can likewise be used for phytoremediation and/or phytomanagement initiatives. For instance, TE resistant plant growth-promoting rhizobacteria and endophytes, as well as TE-resistant mycorrhiza, can be isolated from TE-polluted sites and, subsequently, be used to improve plant survival, growth and performance under the harsh conditions usually present in many TE-polluted sites, particularly mining sites (Weyens et al. 2011; Ma et al. 2015; Burges et al. 2016, 2017; Harrison and Griffin 2020). For instance, rhizobacterial inoculants (*e.g.*, *Arthrobacter nicotinovorans* SA40) have been shown to improve nickel phytoextraction by the hyperaccumulator *Alyssum pintodasilvae* (Cabello-Conejo et al. 2014). Similarly, the inoculation of ultramafic soils with *Microbacterium arabinogalactanolyticum* AY509224 increased soil Ni extractability and uptake by *Alyssum murale* (Abou-Shanab et al. 2006). The inoculation with TE-resistant plant growth-promoting bacteria has been reported to enhance the biomass of different plant species (*e.g.*, *Brassica juncea, Ricinus communis, Helianthus annuus,* and *Sedum alfredii*) growing in TE-polluted soils (Dell'Amico et al. 2008; Jiang et al. 2008; Mastretta et al. 2009; Zaidi et al. 2006). Likewise, Kolbas et al. (2015) reported the positive effects of endophytic bacteria for Cu phytoextraction by sunflower plants. Truyens et al. (2015) found that inoculation of *Agrostis capillaris* plants with endophytes can be beneficial for their establishment during phytoextraction and phytostabilisation of Cd polluted soils. The co-inoculation of *Paenibacillus mucilaginosus* and *Sinorhizobium meliloti* in Cu-contaminated soil planted with *Medicago sativa* improved alfalfa growth and decreased Cu accumulation in shoots, compared to the uninoculated control (Ju et al. 2020). The inoculation of *Pseudomonas vancouverensis* promoted As accumulation efficiency in *Pteris vittata* and *Pteris multifidi* (Yang et al. 2020b). However, the effect of bacterial inoculants on plant growth and TE-accumulation has been shown to be plant species-specific (Becerra-Castro et al. 2012).

Apart from the recognized conservation value of metallophytes present at TE-polluted sites, plant species and, interestingly, microbial (bacteria, fungi) populations living in soils polluted with organic compounds must also be considered for conservation purposes, owing to their intrinsic value as well as their potential use for the rhizoremediation (*i.e.*, degradation of pollutants by rhizosphere bacteria) (Barrutia et al. 2011b; Lacalle et al. 2018; Brereton et al. 2020) and bioremediation of organically polluted soils (Garbisu et al. 2017; Anza et al. 2018; Meglouli et al. 2019; Villaverde et al. 2019). Since bacterial and fungal strains isolated from organically-polluted soils can then be used for bioremediation via bioaugmentation purposes, after thoroughly testing their degrading capabilities and potentials, it is recommended to keep them in a microbial bank for possible future biotechnological applications.

4. Managing biodiversity during phytomanagement

4.1. ***Phytomanagement under the current scenario of climate change***

The negative consequences of climate change can nowadays be undoubtedly identified in the more frequent alteration of natural and agricultural ecosystems, owing to, for instance, higher temperatures, extreme droughts and storms, and an increased likelihood of heat waves and heavy precipitation episodes (Alkorta et al. 2017). Not surprisingly, plant survival and growth are being significantly altered under changing climatic conditions. Furthermore, increasing CO_2 concentrations in the atmosphere are currently changing the physiology of plants, affecting, among other aspects, their growth rate.

Specifically, regarding the choice of plant species for phytomanagement in semi-arid and arid regions (*e.g.*, southern Europe) (Pulighe et al. 2019), and taking into account the critical importance of an adequate water regime for the success of revegetation programs, special attention should be paid to the selection of drought-resistant plant species and ecotypes, since the duration and frequency of extreme droughts is nowadays increasing in many semi-arid and arid regions (Risueño et al. 2020).

The possibility of irrigating phytomanagement crops is decidedly controversial, since water is an increasingly scarce resource in many parts of the world. A proper sustainable management of water resources is currently one of the greatest challenges for our society worldwide. Above all, we must first ensure availability of good quality water for human consumption and agricultural purposes. Regrettably, in the coming decades, the problem of water scarcity will probably get worse than it is now. Predictably, an increase in the world human population will imply more water for human consumption and agricultural production (agriculture accounts for around 70% of the water currently used in the world). Then, it follows that the consumption of good quality water for irrigation of phytomanaged sites is, in general, not considered a valid option, especially in semi-arid and arid regions. An alternative is the use of wastewater for irrigation. An appealing, and currently attention-grabbing, option is the possibility of treating such wastewater by means of rhizofiltration (*i.e.*, the use of plant roots and associated microbes to absorb, concentrate and precipitate pollutants, especially TEs, from polluted effluents and waters) and/or biodegradation, notably using constructed wetlands (CWs) and floating islands (Dushenkov et al. 1995; Schröder et al. 2007; Zhang et al. 2007; Olguin et al. 2017).

Urban wastewater is known to contain nitrogen, phosphorus and other nutrients, leading to an extra beneficial effect for plant growth through fertilization. Irrigation with wastewater is only recommended for non-food and non-fodder crops, and then it would be an ideal option for phytomanagement. Evidently, it would be beneficial to have an efficient urban wastewater treatment plant closed to the site to be phytomanaged, so that the wastewater, directly or preferably after rhizofiltration in a CW does not need to be transported a long distance. Interestingly, CWs can also be used to treat acid mine drainage and then there is the possibility of reusing the treated water to eventually irrigate mine tailings (Pat-Espadas et al. 2018).

Therefore, especially in semi-arid and arid regions of the world, for phytomanagement purposes, it is recommended to select plant species that are resistant to water stress, extreme droughts and heat waves for increasing the long-term success of the phytomanagement strategy (Risueño et al. 2020). For instance, as water supply and its distribution during the crop cycle is a key limiting factor for crop production in SW France, the sunflower and tobacco ability to stand more frequent heat waves and long droughts is certainly an advantage (Kidd et al. 2015; Mench et al. 2018). Although hundreds of plant species are suitable candidates for phytoremediation and/or phytomanagement purposes, there is nowadays an urgent need to identify those which can successfully be used under the current scenario of climate change.

The potential indirect effects of climate change on the soil biota present in phytomanaged sites must be also considered, through the abovementioned climate change-induced alterations in plant growth and physiology. Although higher levels of atmospheric CO_2 are *a priori* not expected to directly affect soil microbial communities (*i.e.*, CO_2 concentrations in the soil are much higher than in the atmosphere), higher atmospheric CO_2 concentrations can indirectly impact on soil microbial communities through higher plant growth, increases in litter deposition and rhizodeposition (often resulting in a stimulation of soil microbial biomass and activity), faster nutrient uptake and water use efficiency (Phillips et al. 2011; Bardgett et al. 2013; Burns et al. 2013; Alkorta et al. 2017). Such climate change-induced variations of rhizodeposition patterns, and concomitant changes in the composition and activity of rhizosphere microorganisms, can modify TE bioavailability in soils (Rajkumar et al. 2013), thus potentially affecting plant performance during phytomanagement.

The consequences of climate change (via higher atmospheric CO_2 concentrations, heat waves, extreme droughts, higher temperatures, etc.) on beneficial plant-microorganism interactions (*e.g.*, plant growth-promoting rhizobacteria, endophytes, and mycorrhiza) are increasingly being studied (Compant et al. 2005, 2010; Classen et al. 2015; Cavicchioli et al. 2019; Risueño et al. 2020). Plant growth-promoting bacteria and fungi can positively affect water-stressed plants and, then, their inoculation should nowadays be strongly considered for phytomanagement. On the other hand, climate-induced changes in soil temperature and moisture can alter soil processes, such as organic matter decomposition and nutrient cycling (Burns et al. 2013), supported, to a great extent, by the activity of soil microorganisms.

Some phytomanagement practices (*e.g.*, the application of organic amendments, low- or no-tillage practices, grassland implementation and afforestation) have great potential for carbon sequestration and, hence, climate change mitigation. The incorporation of trees in phytomanagement initiatives (*e.g.*, as part of intercropping systems) has also acknowledged positive effects in this respect (Schoeneberger et al. 2012; Alam et al. 2014; Zeng et al. 2019a,b; Brereton et al. 2020).

In theory, a possible option for adaptation to climate change in phytomanagement is to incorporate to the planting scheme as many plant species as possible, and preferably from different vegetation types: grasses, shrubs and trees. Nevertheless, in many situations this is not a realistic, feasible option because the specific plant assemblages established for phytomanagement purposes are determined, to a great extent, on the future land use and on the particular non-food crops intended to be delivered to the local chains processing the harvestable biomass.

Moreover, the conservation of plant biodiversity (*e.g.*, the aforementioned metallophyte diversity) is crucial for adaptation to climate change as part of an "insurance policy": different species, varieties and ecotypes may be needed in the future as environmental conditions are altered by climate change.

4.2. ***Promotion of biodiversity under phytomanagement***

Under the inherent constraints inevitably derived from the phytomanagement goals, it is certainly possible to promote biodiversity in phytomanaged sites by means of, for instance, growing as many plant species and varieties/ecotypes from different vegetation types (grasses, shrubs and trees) as possible (Table 1). Interestingly, the establishment of different plant species in phytomanaged sites can result in the generation of a wider variety of valuable products and ecosystem services (Evangelou et al. 2015; Pandey and Bauddh 2018).

Many additional benefits can be obtained when combining different plant species for the phytoremediation and phytomanagement of polluted sites. For instance, the combination of *Pteris vittata* with *Morus alba* and *Broussonetia papyrifera* not only increased the phytoextraction of trace elements but alleviated phytotoxicity as well (Zeng et al. 2019b). In a similar study, the co-planting of *P. vittata* with *Arundo donax, M. alba* and *B. papyrifera* resulted in an improvement of soil health (Zeng et al. 2019a). Furthermore, intercropping with *Paspalum miliaceum* and *Axonopus affinis* was efficient in promoting gravevine growth in Cu-polluted soil by reducing metal bioavailability (De Conti et al. 2019). Interestingly, the combination of tree, shrub and grass species in a metal polluted soil resulted in a more efficient employment of water resources and a higher biodiversity of soil microorganisms (Parraga-

Aguado et al. 2014). In contrast, the co-planting of *Odontarrhena chalcidica* or *Noccaea goesingensis* with *Lotus corniculatus* for Ni removal led to reduced values of shoot biomass (Rosenkranz et al. 2019).

The beneficial effects of co-planting have also been reported for organically-polluted and mixed-polluted soils. Wang et al. (2013) reported an enhanced degradation of PAHs, in the presence of trace elements, when *S. alfredii* was combined with *Lolium perenne* or *Ricinus communis*. In agreement with these results, in their studies on intercropping with *Medicago sativa* and *Festuca arundinacea*, Sun et al. (2011) observed higher PAH degradation values under intercropping *vs.* monoculture. Likewise, intercropping with *M. sativa*, *L. perenne* and *F. arundinacea* improved the degradation of phthalic acid esters. Finally, *F. arundinacea* was also co-planted with *Salix miyabeana* and *M. sativa*, finding out that when crops were cultivated in pairs they showed an enhanced rhizosphere community in terms of the presence of plant growth-promoting bacteria (Brereton et al. 2020).

Aboveground and belowground organisms are closely linked: plants provide organic carbon for soil decomposers and resources for root-associated organisms; in turn, soil decomposers break down dead plant material and regulate plant growth by determining the nutrient supply (Wardle et al. 2004). Different plant species differ in the quantity and quality of litter and root exudates, thus affecting the biomass, activity and diversity (mainly, composition) of soil microbial communities. A more diversified vegetation leads to a higher number of ecological niches and, hence, biodiversity (Risueño et al. 2020). Indeed, higher plant richness results in a higher variety of root exudates and types of litter, thus stimulating biodiversity belowground (Wardle et al. 2004; Haichar et al. 2008). Nonetheless, Li et al. (2015) found no relationship between plant and soil bacterial diversity in an early successional forest, and a negative correlation in a late successional forest. Similarly, Kowalchuk et al. (2000) reported a negative correlation between grassland plant and soil ammonia-oxidizing bacterial diversity. These contradictory results point out to the vast complexity of the multiple links and interactions between aboveground and belowground diversity (Wardle et al. 2004; De Deyn and Van der Putten 2005; Kardol and Wardle 2010), which, at the moment, are far from being well understood. Phytomanagement, apart from increasing soil microbial biomass and activity, can induce shifts in the bacterial community structure at both the total community and functional group levels (Touceda-González et al. 2017a). In a study on the effectiveness of dolomite and compost as amendments for enhancing Cu phytostabilization with *Populus trichocarpa* x *deltoides* cv. Beaupré and *Agrostis gigantea* L., Cu stabilization and phytomanagement induced positive changes in the microbial community of soil leachates, enriching this community with plant beneficial bacteria (Giagnoni et al. 2020).

The presence of phytopathogens and root herbivores in the rhizosphere can produce a negative feedback on plant growth, whereas mycorrhizal fungi and plant-growth promoting rhizobacteria can have a positive one on plant growth (Sessitsch et al. 2013; Sura-de Jong et al. 2015). In any case, the evidence for positive or negative links between aboveground and belowground biodiversity is mixed, and not all of the mechanisms by which aboveground organisms affect belowground diversity and vice versa necessarily lead to correlations of species richness in both domains (Hooper et al. 2000). The common perception that belowground biodiversity should follow similar patterns to those of plant diversity during ecosystem development is challenged by Delgado-Baquerizo et al. (2020).

A higher richness of plant species can, for instance, be used to promote the biodegradation of aged polycyclic aromatic compounds in soil: oxygenated PAHs (some of which are more toxic than their related PAHs) can, however, accumulate in soils during such a plant-assisted remediation process (Bandowe et al. 2019).

Through an increase in plant diversity and, hence, in the number of ecological niches and possible habitats, it is also desirable to promote the aboveground and belowground diversity of animals (*e.g.*, arthropods: insects, arachnids, myriapods, etc.; earthworms; nematodes; mammals; birds; and so on), of course, always paying close attention to the potential risk of pollutant bioaccumulation and biomagnification (*e.g.*, TE biomagnification along the trophic chain) (Peterson et al. 2003). Interestingly, these animals can act as phytomanagement crop auxiliaries, helping to fight pests, pollinate the cultivated plants, etc. (Verkerk et al. 1998; Ferron and Deguine 2005).

Finally, intercropping systems have been extensively investigated for phytoremediation purposes (Sun et al. 2011; Ma et al. 2013; Wang et al. 2013; Alam et al. 2014; De Conti et al. 2019; Zeng et al. 2019a,b; Brereton et al. 2020) with additional benefits in terms of aboveground and belowground diversity. Likewise, as individual plant species repeatedly possess a limited range of TE phytoremediation capacities, functional complementarity principles could be of value for the phytoremediation of soils polluted with multiple TEs by means of using assemblages of species (Desjardins et al. 2018).

Biodiversity provides a wide range of values, some of them indirectly such as aesthetic value, cultural value, spiritual value, scientific value, educational value, etc. Arguably, from an anthropocentric point of view, the most important value of biodiversity comes from the ecosystem services it provides.

Biodiversity preserves the structure and integrity on which healthy ecosystems depend to provide the vital ecosystem services on which we rely on.

Among other values of biodiversity, the following two are often discussed when dealing with the conservation of biodiversity and the human use of natural resources: (1) *intrinsic value:* as such, we have the moral responsibility to preserve biodiversity (well-known nature writers such as Henry David Thoreau, John Muir, Aldo Leopold, etc. have emphasized the intrinsic value of biodiversity); and (2) *utilitarian value:* as such, focused on the commercial and subsistence benefits (*e.g.*, food, medicines, raw materials, energy, etc.) of biodiversity to humankind. Within this utilitarian perspective, the idea is to protect biodiversity so that we can utilize it later for our own benefit. Obviously, this utilitarian value of biodiversity is inextricably link to the phytomanagement concept. In any case, when designing a phytomanagement initiative, it is unquestionably possible to promote biodiversity within the limits imposed by the specific phytomanagement objectives (*e.g.*, by means growing as many plant species as possible) with the concomitant potential benefit of obtaining a wider variety of products and ecosystem services.

Anyhow, the biodiversity concept is anything but simple. Among others, it includes the following aspects: *richness* (or the number of species), *evenness* (relative abundances resulting in rare and dominant species), *composition* (in terms of taxonomic groups), phylogenetic relatedness/distinctiveness, and spatial and temporal distribution. Regarding species composition, biological species are certainly not all equal: there are keystone species, foundation species, umbrella species, flagship species, charismatic species, ecosystem engineers, invasive species, indicator species, chemical engineers, biological regulators, etc., leading us to the difficult and arduous challenge to prioritize among them (Vane-Wright et al. 1991).

Some authors proposed to assign more value to those species that lack close relatives, as by maximizing the conservation of evolutionary diversity, we maximize genotypic, phenotypic and functional diversity, and, hence, provide ecosystems with the most options to adapt to a changing world (Vane-Wright et al. 1991; Cadotte et al. 2010). Besides, some species appear to perform phylogenetically narrow processes (*e.g.*, nitrification, atmospheric nitrogen fixation) while others perform phylogenetically broad processes (*e.g.*, denitrification). The former show a lower degree of functional redundancy, compared to the latter.

To assess the influence of phytomanagement practices on biodiversity (such a broad concept) is anything but easy. There are still many unanswered questions that research is yet to answer, *e.g.* What number of species is a good number? What species composition is best? What degree of phylogenetic distance is more adequate? How differently should we value the different types of species? Are functionally redundant species less valuable than non-functionally redundant species? These questions being answered, we must not take only richness into consideration when promoting biodiversity at polluted sites under phytomanagement. To the best of our expertise and capacities, we must try to consider other relevant aspects also included within the biodiversity concept.

To further complicate matters, biodiversity is difficult to quantify, at least partly, due to the multitude of indices available to measure it (*e.g.*, species richness, Shannon-Wiener entropy, Simpson's index, Berger-Parker index, etc.). This is not surprising because of the abovementioned complexity of all the aspects of biodiversity, which inevitably leads to the fact that no single perfect indicator for biodiversity can be devised (Duelli and Obrist 2003). As a matter of fact, the choice of index often depends on the question(s) to be answered, as well as on the specific aspect(s) or entity of biodiversity to be evaluated. Paradoxically, most diversity indices have traditionally relied on three untrue assumptions: (i) all species are equal; (ii) all individuals are equal; and (iii) species abundances have been correctly assessed with appropriate tools and in similar units (Magurran 2004). In any case, although the choice of index(es) depends, to a great extent, on the specific questions and objectives of the study, three of the most commonly used indices are the Margalef´s index for richness, the Shannon-Weaver´s index for diversity and the Simpson´s index for dominance.

Similarly, the use of indices for quantifying functional biodiversity (functional richness, functional evenness, and functional divergence) is essential to better understand the links between biodiversity, ecosystem functioning and environmental constraints (Mouchet et al. 2010). Indeed, many studies on the impact of disturbances (*e.g.*, agronomic practices, contamination, climate change, nitrogen deposition, etc.) on biological diversity are focused exclusively on structural biodiversity (usually, of only one or a few taxonomic groups). But phytomanagement has a strong functional component related to the provision of ecosystem services. Thus, it is highly beneficial to include both types of biodiversity, *i.e.* structural and functional diversity, when promoting biodiversity under phytomanagement. Apart from a selection, as wide as possible, of taxonomic groups, an analysis of functional groups, traits, guilds and so on must be included in phytomanagement initiatives (Kumpiene et al. 2014; Durand et al. 2017; Touceda-González et al. 2017a,b; Xue et al. 2018; Burges et al. 2020).

Although the identification of links between structural and functional biodiversity is undoubtedly a challenging task, such identification is of much value from both an academic/scientific and management point of view. Statistical multivariate analyses, applied to the group of variables used to measure structural and functional diversity, are suitable tools for the establishment of hypotheses regarding the abovementioned links.

The topic of the selection of the best indices to quantify both structural and functional biodiversity is not within the scope of this document. Nonetheless, we encourage the use of various indices for covering as much as possible the different aspects of the term biodiversity: richness, abundance, phylogenetic relatedness, functional traits, etc. Ideally, one should make the best efforts possible to evaluate the effect of phytomanagement practices on the various levels of biodiversity: genetic, species, populations and communities/ecosystems. However, biodiversity is not simply the sum of all ecosystems, species and genetic material, as it represents the variability within and among them.

In particular, for soil microorganisms, the assessment of genetic diversity is indispensable for microbial ecologists since: (i) all the current definitions of "species" are inadequate for prokaryotes, among other reasons due to the transfer of genes by horizontal gene transfer; and (ii) most microorganisms cannot be cultivated and so we have no other choice than to study them by means of the application of molecular biology techniques. In consequence, most soil microbial ecologists are nowadays focused on the use of next generation sequencing techniques (*e.g.*, metabarcoding, metagenomics) for the quantification of soil microbial diversity. But next generation sequencing has still many technical limitations and then we must be cautions when drawing conclusions about the effect of disturbances or practices (*e.g.*, phytomanagement) on soil microbial diversity.

Most studies on the effect of phytoremediation or phytomanagement practices on microbial diversity are focused on soil microbial communities, especially rhizosphere microorganisms. In this respect, more attention should be paid to plant microbiota and plant microbiomes (*e.g.*, in the phyllosphere) under phytomanagement (Imperato et al. 2019).

Similarly, concerning genetic diversity, there are still many unanswered questions, such as, for example: The more genes the better? Are all genes equally important? Can we talk about "good" genes (*e.g.*, genes involved in contaminant biodegradation pathways) and "bad" genes (*e.g.*, antibiotic resistance genes)? How can be combined data from metagenomic, metatranscriptomic and metaepigenomic studies?

Likewise, when dealing with ecosystem diversity (*i.e.*, the richness and complexity of biological communities, including trophic levels and ecological processes, together with the chemical and physical environment), additional questions emerge: How many trophic levels do we need? Are all of them equally important? How many species per trophic level are needed?

Regarding the critical links between biodiversity and ecosystem functioning, one should take into consideration the concept of emergent properties, *i.e.* those new qualities that appear on higher integration levels and represent more than the sum of the low-level components (Reuter et al. 2005). For understanding these emergent properties, the interaction between the different elements must be closely studied (Reuter et al. 2005). In consequence, when possible, key biological interactions should be identified and studied during phytomanagement initiatives, since they support the functioning of the ecosystem and are the basis of emergent properties.

On the other hand, when promoting biodiversity under phytomanagement, it is important to always include organisms from the different levels of the trophic chain. Instead, when evaluating their effect on biodiversity, most phytomanagement initiatives only pay attention to aboveground botanical diversity (richness, composition, vegetation structure) and, occasionally, include some belowground soil biota, in many cases just microorganisms owing to their well-known key role in critical soil processes and, hence, functions and ecosystem services (Xue et al. 2018; Burges et al. 2020). Nonetheless, for a biodiversity assessment with more ecological relevance, it is desirable to include taxonomic groups from the different levels of the trophic chain. As a matter of fact, we should study as many taxonomic groups from the food web as possible (Garrouj et al. 2018; Ali et al. 2019; Prins et al. 2019).

Simplifying, the aboveground food web includes producers (plants), primary consumers (herbivores) and secondary consumers (predators). Regarding consumers, there is an unresolved debate regarding the benefits and disadvantages associated to the presence of animals in phytomanaged sites. Actually, when dealing with the remediation of polluted sites, in many cases, animals are deliberately excluded, in an attempt to avoid possible ecotoxic effects on exposed animals and, also, to minimize the risk of bioaccumulation and biomagnification (Mann et al. 2011). But animals (*e.g.*, arthropods, earthworms, mammals, birds, etc.) can act as phytomanagement crop auxiliaries, helping to fight pests, pollinate the cultivated plants, etc. (Verkerk et al. 1998; Ferron and Deguine 2005).

Pertaining to the soil ecosystem, an amazing diversity of soil organisms make up its food web: bacteria, fungi, algae, protozoa, nematodes, micro-arthropods, earthworms, insects, small vertebrates (mice, moles), etc. Like all food webs, the soil food web is fueled by primary producers such as plants, lichens, mosses, photosynthetic bacteria (*e.g.*, cyanobacteria) and unicellular algae. The remaining members of the soil biota obtain energy and carbon by consuming the organic compounds produced by primary producers.

*4.3. **Adaptive monitoring during phytomanagement***

When dealing with long-term monitoring programs, such as the one for assessing the influence of phytomanagement practices on biodiversity, as time passes, it is inevitable that (i) new analytical techniques, methods and equipments might appear in the market; (ii) different approaches, concepts, ideas, etc. might come up; (iii) changes in the ecosystem developmental stage will occur; (iv) unexpected environmental threats might emerge; (v) budget fluctuations might threaten the initiative, and so forth (Epelde et al. 2014). For that reason, we propose that the paradigm of adaptive monitoring (this paradigm enables monitoring programs to evolve iteratively as new information emerges and research questions change) should be incorporated to the long-term monitoring of the effect of phytomanagement practices on biodiversity.

To this purpose, among other aspects, (i) well-formulated, clear and tractable questions must be established at the beginning of the phytomanagement initiative; (ii) a rigorous statistical design must be implemented from the onset of the study, notably accounting for the spatial variability of soil contamination, contaminant exposure and pollutant linkage; and (iii) a conceptual model of the site under phytomanagement must be created (Cundy et al. 2016).

As part of the adaptive monitoring program, periodically (the time period will depend on the specific phytomanagement initiative), an expert judgment analysis must be organized to revise and, if necessary, update the different aspects that make up the biodiversity monitoring program. Expert judgment analyses often encourage the forging of partnership between researchers, policy-makers and resource managers, an aspect of the utmost importance in phytomanagement (Cundy et al. 2013).

The accomplishment of economic, social and environmental benefits is a key aspect of phytomanagement (Cundy et al. 2016). In particular, the provisioning of ecosystem services (carbon sequestration, improvement of soil fertility, control of soil erosion, improvement of air quality, climate and water regulation, production of atmospheric oxygen, provision of habitat, etc.) is a crucial component of phytomanagement initiatives (Burges et al. 2018).

The provision of ecosystem services is underpinned by a variety of ecological processes and functions which themselves are driven by biodiversity. Although changes in biodiversity can affect ecosystem processes and, hence, the provision of ecosystem services, in some situations, biomass, species composition, functional traits, etc. are more important than biodiversity itself for the provisioning of those services. Nonetheless, trade-offs between biodiversity and ecosystem services might arise in some situations (Bandowe et al. 2019). Also, trade-offs and conflicts between the different ecosystem services themselves might also emerge, and, then, it is desirable to select from the onset what specific ecosystem services to promote, and implement measures that minimize conflicts.

Conclusion

Paraphrasing the three well-known M´s of successful trading (Mind, Money management, and Method), we can visualize the links between biodiversity and phytomanagement according to three M´s of successful phytomanagement: (1) *Mind:* for effective phytomanagement, we must use our mind and creativity to design the best strategy for each specific site and casuistry (here, following the medical aphorism “there are no diseases but sick people”, we can state that “there is no pollution but polluted areas”; all of them are different and require a site-specific assessment). In this respect, biodiversity provides ideas, models and strategies (tested through millions of years of evolution) that we can learn from; (2) *Management:* for successful phytomanagement, we must apply scientifically-based adaptive management, especially under the current scenario of climate change. Biodiversity provides a myriad of species, metabolic capabilities, functional traits, etc. which we can use in response to changing conditions; and (3) *Money:* a fruitful phytomanagement will provide economic value through products (crops for biomass-processing technologies) and ecosystem services which can help fuel our bioeconomy. Interestingly, the promotion of biodiversity in phytomanaged sites can result in the generation of a wider variety of valuable products and ecosystem services, while minimizing pollutant-induced environmental risks.

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Table 1: Ten examples of effects of biodiversity under phytomanagement.

Plant species	Contaminants	Main finding	Reference
Pteris vittata co-planted with *Morus alba* and *Broussonetia papyrifera*	As, Cd, Pb and Zn	Co-planting alleviated toxicity and improved phytoextraction	Zeng et al. 2019b
Pteris vittata co-planted with *Arundo donax, Morus alba* and *Broussonetia papyrifera*	As, Cd, Pb, and Zn	Co-planting enhanced *P. vittata* growth and metal(oid) accumulation, and improve soil quality	Zeng et al. 2019a
Co-planting *Sedum alfredii* with *Lolium perenne* or *Ricinus communis*	Metals and PAHs	Co-planting *S. alfredii* with ryegrass or castor enhanced pyrene and anthracene dissipation	Wang et al. 2013
Intercropping: *Medicago sativa* with *Festuca arundinacea*	PAHs	Removal PAHs under intercropping was higher than under monoculture	Sun et al. 2011
Odontarrhena chalcidica or *Noccaea goesingensis* co-planted with *Lotus corniculatus*	Ni	Intercropping with *L. corniculatus* tended to decrease the shoot biomass of both species	Rosenkranz et al. 2019
The grass *Piptatherum miliaceum,* the shrub *Helichrysum decumbens*, and the trees *Pinus halepensis* and *Tetraclinis articulata*	Metal(loid)s	A diverse set of plant species with contrasting life forms may result in a more efficient employment of water resources and a higher biodiversity not only in relation to flora but also soil microbes	Parraga-Aguado et al. 2014
Medicago sativa, Lolium perenne and *Festuca arundinacea*	Phthalic acid esters (PAEs)	Intercropping with the three species was the most effective treatment for PAEs removal	Ma et al. 2013
Monocultures and polycultures of *Festuca arundinacea, Medicago sativa* and *Salix miyabeana*	Ag, As, Cd, Cr, Cu, Pb, Se and Zn	Co-cropping with the three species was the most robust scenario for remediation of multiple trace element contaminated soil	Desjardins et al. 2018
Grapevine was grown in monocropping, intercropping with *Paspalum plicatulum* and intercropping with *Axonopus affinis*	Cu	Intercropping with *P. plicatulum* and *A. affinis* was efficient in promoting the growth of grapevines at moderate and low levels of Cu contamination by reducing its bioavailability	De Conti et al. 2019
Co-cropping of *Festuca arundinacea, Salix miyabeana* and *Medicago sativa*	Trace elements and persistent organic pollutants (POPs)	The crops cultivated in pairs retained rhizosphere microbiome bacteria involved in plant growth promotion, POP tolerance and degradation, and improved nutrient acquisition	Brereton et al. 2020

Figure legends

Figure 1: Evolution from phytoremediation to phytomanagement.