Universitat Autònoma de Barcelona

VALORITZACIÓ DE RESIDUS ORGÀNICS EN TECNOSOLS I AVALUACIÓ DE LA RESTAURACIÓ D'ESPAIS DEGRADATS

Vicenç Carabassa i Closa

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Tesi doctoral

CREAF

SEVERO OCHOA EXCELLENCE

UAB Universitat Autònoma de Barcelona

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Tesi doctoral

Memòria presentada per
Vicenç Carabassa i Closa
per optar al grau de Doctor

Amb el vistiplau del director de tesi:
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A la Guadalupe, a qui tinc present més que mai en aquests dies.

ÍNDEX

Here is the means to end the great extinction spasm. The next century will, I believe, be the era of restoration in ecology. —E.O. Wilson, 1992

Resum

La degradació dels hàbitats, la sobreexplotació dels recursos naturals, la contaminació, les espècies invasores o el canvi climàtic estan afectant la provisió de serveis ecosistèmics arreu del Món, la qual cosa ens obliga a fer plantejaments i reflexions urgents sobre com entenem, valorem i gestionem els ecosistemes. Algunes activitats humanes necessàries per a l'estil de vida actual, com l'extracció de recursos miners, la construcció d'infraestructures de transport, o la pròpia agricultura industrial, provoquen impactes severs sobre els ecosistemes, els quals ens proveeixen de serveis encara més necessaris com la regulació climàtica i hídrica, el control de l'erosió, el filtratge de contaminants, o la provisió d'aigua i la producció d'aliments, entre molts d'altres. Per tant, encara que sigui des d'una visió totalment antropocèntrica, és necessari i urgent reduir els impactes sobre els ecosistemes, a la vegada que restaurem els espais degradats, recuperant així els serveis ecosistèmics. Aquesta tesi pretén contribuir a donar resposta a aquesta necessitat mitjançant propostes d'aprofitament de residus orgànics per a la creació de tecnosols i la validació de mètodes per a la restauració i seguiment d'aquests espais degradats, especialment aquells afectats per activitats mineres.

En un primer treball es valida la utilitat del protocol OCDE-217, basat en l'activitat respiratòria del sòl, per estudiar la toxicitat d'alguns dels metalls pesants habitualment presents en major proporció a les esmenes orgàniques utilitzades en restauració, especialment als fangs de depuradora urbana. En aquest sentit, el protocol OCDE-217 ha estat una metodologia vàlida per a la detecció d'efectes adversos (i també beneficiosos) dels tres metalls estudiats, Cr, Zn i Cu, sobre la respiració del sòl induïda per substrat. No obstant, les toxicitats avaluades per la prova de respiració del sòl OCDE-217 han resultat baixes en el sòl escollit (sòl sorrenc lleugerament àcid), malgrat la seva limitada capacitat d'adsorció i precipitació de metalls pesants que, per tant, romanien potencialment biodisponibles. Tot i així, els valors EC_{50} per Cr, Zn i Cu obtinguts són comparables als presentats per altres autors que han treballat amb sòls de característiques similars. La magnitud dels efectes inhibidors observats també és coherent amb les referències consultades, essent el Cu el metall més tòxic després de 28 dies d'exposició. També s'ha comprovat que l'adició de matèria orgànica fresca emmascara l'efecte dels metalls pesants sobre la respiració del sòl ja que incrementa la biomassa microbiana, així com l'adsorció i la immobilització dels metalls.

En un segon treball de la tesi s'entra en la caracterització de diferents tipus d'esmenes orgàniques procedents de residus urbans, així com en l'avaluació dels seus efectes sobre el sòl i la vegetació quan són usats en la construcció de tecnosols per a la restauració de pedreres, abocadors o talussos de carretera. Així s'avaluen els efectes de l'aplicació de fangs de depuradora urbana a curt (2 anys) i mitjà termini (10 anys), en una selecció de tipus d'actuacions de rehabilitació representatives de les realitzades en pedreres de roca calcària. S'ha demostrat que l'ús de fangs de depuradora permet millorar els resultats de la rehabilitació de sòls degradats, a la vegada que es valoritzen residus orgànics i minerals. La construcció de tecnosols amb fangs de depuradora facilita una ràpida revegetació, especialment pel que fa a espècies

herbàcies, fet que permet l'estabilització ràpida de talussos i controla l'escorrentiu, la qual cosa redueix també l'erosió superficial, que és un objectiu prioritari en talussos de fort pendent com els que trobem en activitats extractives, infraestructures de transport o abocadors.

A mitjà termini (una dècada), els sòls esmenats amb fangs de depuradora afavoreixen l'enriquiment del sòl en matèria orgànica i el segrest de carboni, ja que hem comprovat que contenen fins a cinc vegades més de carboni orgànic que els sòls homòlegs no esmenats. Aquest enriquiment és el resultat de l'augment de la producció primària a causa dels nutrients aportats pels fangs, que encara és evident després de deu anys. Aquest augment de la producció primària es tradueix també en un major recobriment vegetal que protegeix el sòl de l'erosió. En general podem dir que l'aplicació de fangs a dosis moderades, d'acord amb el protocol anomenat RESTOFANGS, afavoreix un major desenvolupament de la vegetació, especialment de l'estrat herbaci en els primers anys, però també progressivament de l'arbustiu i l'arbori, sense limitar el reclutament a mig termini de les plantes pròpies del sistema natural de la zona.

Pel que fa als bioestabilitzats, resultants de l'estabilització aeròbica de la fracció orgànica de residus municipals, s'ha constatat que existeix una important heterogeneïtat en relació als principals paràmetres estudiats. Aquesta heterogeneïtat es dona entre diferents plantes de tractament de residus, però també entre diferents lots d'una mateixa planta, cosa que suposa una important restricció a l'hora de fer generalitzacions per a l'ús dels bioestabilitzats. Els digests, procedents de la digestió anaeròbica de la matèria orgànica obtinguda per la separació mecànica de la fracció resta dels residus urbans, presenten un contingut baix d'impureses i una major concentració de N que els bioestabilitzats, la qual cosa els fa més adequats per a ser aplicats al sòl. L'elevat contingut en N dels digests afavoreix el desenvolupament i el creixement de les plantes, la qual cosa el fa especialment interessants en aquells casos en que cal garantir una coberta herbàcia que protegeixi el sòl de l'erosió. Podem dir que els digests s'assemblen més als fangs EDAR que als bioestabilitzats, malgrat l'origen comú amb els darrers. A més, la composició dels digests és més estable entre plantes de tractament i lots. Pel que fa a la dosificació, s'ha comprovat que les dosis superiors a 20 $g \cdot kg^{-1}$ no milloren el desenvolupament i el creixement de la vegetació, tant en bioestabilitzats com en digests, i també augmenten el risc de contaminació de les aigües subterrànies i la toxicitat sobre la fauna del sòl. El missatge és clar, les esmenes orgàniques generen beneficis als sòls degradats o deficitaris en matèria orgànica, però no té sentit aplicar-les a dosis superiors a les establertes en els protocols que hem publicat o les indicades en d'altres codis de bones pràctiques agràries.

El darrer treball de la tesi es dedica a l'establiment i validació d'un protocol per a l'avaluació de la restauració d'activitats extractives, anomenat RESTOQUARRY. Diferents actors vinculats a la pràctica de la restauració ecològica apunten a la necessitat d'establir protocols de seguiment de la restauració, que han de ser senzills i aplicables a un ampli ventall de situacions o tipologies de restauració. El protocol RESTOQUARRY pretén contribuir a donar-hi resposta al ser una eina adreçada al personal de les empreses extractives i de l'Administració, que ajuda a incrementar el compromís i la implicació dels enginyers de mines i el personal encarregat d'executar la restauració, la qual cosa es tradueix en una major qualitat de les restauracions efectuades. La major part dels indicadors proposats en el protocol avaluen de manera indirecta serveis i/o funcions ecosistèmiques, permetent en alguns casos la seva quantificació. Fins i tot alguns dels indicadors més generals (indicadors del conjunt de l’àrea restaurada), com són els relacionats amb impactes o integració visual, poden ser considerats aproximacions a la valoració de serveis ecosistèmics culturals.

El protocol RESTOQUARRY permet distingir restauracions de qualitat, d’aquelles que necessiten l’adopció de mesures correctores. També permet identificar problemes crítics que posen en risc l’èxit de la restauració i que requereixen de canvis profunds en el disseny i l’execució dels treballs. La major part de les restauracions avaluades amb RESTOQUARRY requereixen de mesures correctores per tal de millorar algun aspecte de la restauració, fet que demostra l’eficàcia del protocol.

Abstract

Habitat degradation, overexploitation of natural resources, pollution, invasive species or climate change are affecting the provision of ecosystem services around the world, forcing us to make urgent approaches and reflections on how we understand, value and manage ecosystems. Some human activities necessary for the current lifestyle, such as open-pit mining, the construction of transport infrastructures, or industrial agriculture, cause severe impacts on ecosystems. These ecosystems provide us necessary services, such as climate and water regulation, erosion control, pollutant filtration, or water supply and food production, among many others. Therefore, even from a totally anthropocentric point of view, it is necessary and urgent to reduce these impacts as much as possible, while recovering the ecosystem services through the restoration of the degraded lands. This thesis aims contributing to fill in the lack of this need by taking profit of organic wastes for technosols construction and validating methods for the restoration and monitoring of these degraded areas, especially those affected by mining activities.

A first work validates the usefulness of the OECD-217 protocol, based on the soil respiratory activity, to establish safe concentrations of some of the heavy metals usually present in greater proportion in the organic amendments used in restoration, especially on sewage sludge from urban wastewater treatment plants. In this sense, the OECD-217 protocol has been a valid methodology for the detection of adverse (and also beneficial) effects of the three studied metals, Cr, Zn and Cu, on substrate-induced soil respiration. However, the toxicities assessed by the OECD-217 soil respiration test were low in the reference soil (slightly acidic sandy soil), despite its low adsorption and precipitation capacity of heavy metals which therefore remained potentially bioavailable. However, the EC_{50} values for Cr, Zn and Cu obtained are comparable to those presented by other authors who have worked with soils of similar characteristics. The magnitude of the inhibitory effects observed is also consistent with the references consulted, with Cu being the most toxic metal after 28 days of exposure. The addition of fresh organic matter masked the effect of heavy metals on soil respiration as it increased microbial biomass as well as adsorption and immobilization of metals.

A second work of the thesis goes into the characterization of different types of organic amendments from urban waste, as well as the evaluation of the effects on soil and vegetation of their use in the construction of technosoils for the restoration of quarries, landfills or road slopes. Thus, the effects of the application of sewage sludge in the short (2 years) and medium term (10 years) are evaluated in a selection of types of restorations representative of limestone quarries. The use of sewage sludge improved the rehabilitation of degraded soils, while at the same time contributing to the valorization of organic and mineral wastes. The construction of technosoils with sewage sludge facilitated fast revegetation, especially with regard to herbaceous species, which allowed the fast stabilization of slopes and the reduction of water erosion.

In the medium term (a decade), sewage sludge amendment favors soil enrichment in organic matter and carbon sequestration. We found that amended soils contain up to five times more organic carbon than unamended homologous soils. This enrichment is the result of increased primary production due to the

nutrients provided by the sludge, which is still evident after ten years. This raise in primary production also increased plant cover that protects the soil from erosion. In general, we can say that the application of sewage sludge in moderate doses, according to the protocol called RESTOFANGS, favors a greater development of the vegetation, especially herbaceous in the first years, but also shrubs and trees later, without limiting the recruitment of native plants from the surroundings.

With regard to compost-like-output (biostabilized), resulting from the aerobic stabilization of organic matter obtained by the mechanical separation of the remaining fraction of urban waste, there is a significant heterogeneity in relation to the main parameters studied. This heterogeneity occurs between different waste treatment plants, but also between different batches of the same plant, which is a restriction when making generalizations about the use of these wastes. Digestates have a low content of impurities and a higher concentration of N than biostabilized, which makes them more suitable for be applied to the soil. The high N content of digestates favors plant development and growth, which makes it especially interesting in those cases where it is necessary to guarantee an herbaceous cover that protects the soil from erosion. In addition, the composition of the digestates is more stable between treatment plants and batches. Regarding doses, applying these wastes above 20 $g \cdot kg^{-1}$ do not improve the development and growth of vegetation, both biostabilized and digestate, increasing the risk of groundwater pollution and the toxicity to soil fauna. The message is clear, organic amendments generate benefits to soils poor in organic matter, but it does not make sense to apply them at doses higher than those established in the protocol.

The last work of the thesis is devoted to the establishment and validation of a protocol for the evaluation of the restoration of extractive activities, called RESTOQUARRY. Different actors linked to the practice of ecological restoration point to the need to establish protocols for monitoring restoration, which must be simple and applicable to a wide range of situations or types of restoration. The RESTOQUARRY protocol aims to help filling this gap by being a tool designed for the staff of extractive companies and public Administration. The protocol helps to increase the commitment and involvement of mining engineers and staff responsible for carrying out the restoration, which indirectly contributes to increase the quality of the restorations carried out. Most of the indicators proposed in the protocol indirectly evaluate ecosystem services and/or functions, allowing in some cases their quantification. Even some of the more general indicators (indicators of the restored area as a whole), such as those related to impacts or visual integration, can be considered approaches to the assessment of cultural ecosystem services.

The RESTOQUARRY protocol makes it possible to distinguish quality restorations from those that require the adoption of corrective measures. It also identifies critical issues that jeopardize the success of the restoration and require profound changes in the design and execution of the work. Most restorations evaluated with RESTOQUARRY require corrective actions in order to improve some aspect of the restoration, which demonstrates the effectiveness of the protocol.

1. Introducció general

Sobre la necessitat de restaurar els espais degradats

La degradació dels hàbitats, la sobreexplotació dels recursos naturals, la contaminació, les espècies invasores o el canvi climàtic estan afectant la provisió de serveis ecosistèmics arreu del Món (Pereira *et al.*, 2012). S'estima que el 60% dels ecosistemes mundials estan més o menys degradats o sobreexplotats (MEA, 2005). La degradació dels ecosistemes terrestres i marins afecta directament 3.200 milions de persones i té un cost aproximat en pèrdues d'espècies i serveis ecosistèmics del 10% del PIB mundial anual (UN, 2019a). Els ecosistemes que desapareixen o es degraden prestaven importants serveis per a l'alimentació i el benestar de la humanitat, incloent l'abastament d'aigua dolça, la protecció contra els riscos naturals, que darrerament ha estat especialment invocat, o la provisió d'hàbitats per a diferents espècies de peixos i pol·linitzadors, entre molts d'altres (UN, 2019a). Existeixen multitud de dades i estudis que alerten de la crítica situació de molts hàbitats i de la degradació de la provisió de serveis, i de la pròpia biodiversitat que els habita i suporta (EEA, 2010, 2015; Díaz *et al.*, 2019). Potser un dels exemples més dramàtics d'aquests impactes el tenim en la desertificació, és a dir, la conversió de terres més o menys fèrtils cap a deserts (desertització) per causa de l'acció humana. Segons l'Agència Ambiental Europea diverses regions de la Península Ibèrica mostren nivells alarmants de desertificació (EEA, 2005).

La capacitat humana per afectar i modificar el medi ambient de manera cada vegada més gran ens obliga a fer plantejaments i reflexions urgents sobre com entenem, valorem i gestionem els ecosistemes (Hilderbrand *et al.*, 2005). Algunes activitats humanes necessàries per a l'estil de vida actual, com l'extracció de recursos miners, la construcció d'infraestructures de transport, o la pròpia agricultura, provoquen impactes severs sobre els ecosistemes, els quals ens proveeixen de serveis ecosistèmics encara més necessaris, com la regulació climàtica i hídrica, el control de l'erosió, el filtratge de contaminants, o la provisió d'aigua i la producció d'aliments, entre molts d'altres. Per tant, les accions impactants que com a societat fem sobre els ecosistemes naturals per tal d'extraure recursos ens detrauen a la vegada serveis encara més necessaris per a la nostra supervivència en aquest planeta, fet que ens obliga a reflexionar i actuar si no volem comprometre el nostre futur i el de les properes generacions. En aquest sentit, no hem de perdre de vista l'antropocentrisme del concepte de "serveis ecosistèmics" (Costanza *et al.*, 1997; MEA, 2005), però això tampoc ens ha de portar a donar-los per garantits. Des de 1970, les tendències en la producció agrícola, la pesca, la producció de bioenergia i l'extracció de materials han augmentat, però 14 de les 18 categories de contribucions de la natura al benestar humà que es van avaluar en el darrer informe de la Plataforma Intergovernamental en Biodiversitat i Serveis Ecosistèmics(IPBES), han disminuït (Díaz *et al.*, 2019).

La degradació del sòl, juntament amb el canvi climàtic, és un dels principals reptes als que s'enfronta la humanitat (Richard *et al.*, 2015). La degradació del sòl es tradueix en una disminució de la capacitat de l'ecosistema de proporcionar béns i serveis als seus beneficiaris (FAO and ITPS, 2015). El segellat del sòl, la destrucció de tots els seus components degut a la mineria, la salinització, la contaminació, o l'erosió són només alguns exemples de degradació del sòl. El canvi d'ús del sòl associat a l'activitat humana és la causa principal de l'erosió accelerada que pateix, i que té implicacions en els cicles de nutrients i carboni, la productivitat agrària i, a la vegada, les condicions socioeconòmiques mundials (Borrelli *et al.*, 2017). La quantitat de sòl perduda anualment per processos d'erosió hídrica s'estima en 20–30 Gt any^{-1}, mentre que l'erosió eòlica mobilitza fins a 2 Gt any^{-1} de pols. L'erosió hídrica és problemàtica a gran part de les zones muntanyoses i de relleu ondulat que s'utilitzen com a terres de cultiu a tots els continents, fins i tot en aquelles regions on s'han fet importants esforços de conservació de sòls com al mitjà oest dels EUA (FAO and ITPS, 2015).

La contaminació de l'aire, el sòl i les aigües és un problema creixent a nivell mundial (Baselt, 2014). Existeixen diverses fonts de contaminació local i difusa del sòl per metalls pesants, derivats del petroli i productes de síntesi, associades a activitats industrials, vessaments per accidents, pràctiques agrícoles inapropiades i l'aplicació reiterada de residus urbans o ramaders, que provoquen la inhibició més o menys intensa de l'activitat biològica del sòl, l'alteració de la descomposició de les restes orgàniques, i en darrer terme la reducció del rendiment dels cultius (FAO and ITPS, 2015; Stanners and Bourdeau, 1995).

Històricament és conegut que la contaminació per metalls pesants ha provocat efectes adversos sobre el medi ambient i la salut humana. Alguns exemples dramàtics d'aquest tipus de contaminació són els brots de malalties a Minamata a causa del Hg (Ekino *et al.*, 2007; Takaoka *et al.*, 2008) i a Itai-Itai a causa de Cd (Baselt, 2014). La contaminació del sòl per causes humanes ha arribat a nivells alarmants. Ha estat identificada com la tercera amenaça més important per les funcions del sòl a Europa i Euràsia, la quarta al nord d'Àfrica, la cinquena a Àsia, la setena al pacífic nord-oest, la vuitena a Amèrica del Nord i la novena a l'Àfrica subsahariana i a Amèrica llatina (FAO and ITPS, 2015). Només a la Unió Europea s'han inventariat més de 650.000 emplaçaments potencialment contaminants dels que només un 10% s'han remeiat (Payá and Rodríguez, 2018). A Àsia aquesta xifra s'estima en un total de més de 3 milions d'emplaçaments potencialment contaminats, i al conjunt del planeta s'eleva a 5 milions (IRIC, 2015).

Per tant, la degradació dels ecosistemes afecta directament el nostre benestar, però el que és encara més important, el benestar de les generacions futures. Per tant, encara que sigui des d'una visió totalment antropocèntrica, és urgent i necessari en primer lloc reduir en la mesura del possible aquests impactes, a la vegada que recuperem aquests serveis ecosistèmics mitjançant la restauració dels espais degradats. No obstant, no hem de caure en la trampa de no afrontar el primer, que implica canvis profunds en el nostre model econòmic i social, excusant-nos en el segon (Hilderbrand *et al.*, 2005). Un exemple d'això, tot i que segurament excessivament simplista, podrien ser les plantacions que realitzen indústries altament contaminants per tal de compensar emissions de carboni, quan aquestes es podrien evitar o reduir mitjançant models de producció/consum alternatius.

Just aquest any encetem la dècada 2020 – 2030, declarada per les Nacions Unides com de la Restauració dels Ecosistemes. Es proposa un període de deu anys per dur a terme la restauració de 350 milions d'hectàrees d'ecosistemes terrestres i aquàtics degradats que generaran una major prestació de serveis ecosistèmics. La restauració dels ecosistemes genera beneficis per a la seguretat alimentària, la mitigació del canvi climàtic i la adaptació, i també pot prevenir conflictes i migracions provocades per la degradació ambiental, entre molts d'altres (UN, 2020). Aquesta tesi pretén ser una petita contribució a aquesta iniciativa, generant eines i coneixements per a la pràctica de la restauració ecològica.

La restauració ecològica: definició, principis i atributs

Segons la Societat internacional per a la Restauració Ecològica (SER), la restauració ecològica és tota activitat encaminada a iniciar o millorar la recuperació d'un ecosistema des del punt de vista de la seva "salut", integritat i sostenibilitat. Freqüentment, els ecosistemes sotmesos a un procés de restauració ecològica han patit processos de degradació, han estat malmesos, transformats o completament destruïts de manera directa o indirecta com a resultat d'activitats humanes. En alguns casos, aquests impactes han estat causats o agreujats per agents naturals com incendis, inundacions, riuades, grans tempestes o erupcions volcàniques, fins a un punt en què l'ecosistema no pot recuperar de manera espontània l'estat previ a la pertorbació ni la trajectòria històrica que l'hauria de permetre recuperar aquest estat (Clewell *et al.*, 2004) en un període de temps no massa llarg des del punt de vista de la percepció humana. Per tant, segons la definició de la SER, l'objectiu principal de la restauració és retornar l'ecosistema a la seva "trajectòria històrica" entesa com el punt de partida idoni per assolir amb èxit aquest procés. No obstant, la trajectòria històrica d'un ecosistema en un estat de degradació avançat pot ser difícil o inclús impossible de determinar de manera acurada. Tot i així, la tendència general i les fronteres d'aquesta trajectòria es poden establir amb una combinació del coneixement de l'estructura, funcions i composició preexistents, mitjançant l'estudi d'ecosistemes poc pertorbats, la informació sobre les condicions ambientals de la regió i l'anàlisi d'altra informació ecològica i històrica. La combinació d'aquestes fonts permet fixar la trajectòria històrica o les condicions de referència, així com la seva translació mitjançant models predictius en el propi procés de restauració, que hauria de permetre millorar la salut i integritat de l'ecosistema segons la visió d'aquests autors (Clewell *et al.*, 2004).

Una crítica habitual a aquesta definició i manera d'entendre la restauració és que parteix d'una concepció dels sistemes naturals com a entitats intactes o prístines, sense tenir massa en compte, o minimitzant, l'activitat humana com a agent transformador del paisatge, i per tant dels sistemes naturals. Es tractaria de la visió amb el que podríem anomenar un "biaix americà", fruit de la relativament llarga trajectòria de la restauració ecològica als Estat Units, que de fet és un dels llocs on es va desenvolupar en major mesura aquesta disciplina de l'ecologia de la restauració[1], i molt probablement on té una major implantació. Podríem dir que una visió més "europea" d'aquest concepte és aquella que té en compte l'ésser humà com un element "clau" de l'ecosistema, que el gestiona i modifica segons els seus interessos i necessitats, arribant en determinats casos a sobreexplotar-lo, degradar-lo o inclús a destruir-lo per complet, la qual

[1] Amb el permís de Bradshaw (Bradshaw, A. D. and M. J. Chadwick (1980) The Restoration of Land: The Ecology and Reclamation of Derelict and Degraded Land" (Book Review)." Town Planning Review, 52(4), pp. 478).

cosa dificulta de manera important la definició de quines són o eren les condicions naturals als paisatges culturals europeus (Bradshaw and Holmqvist, 1999; Hannon and Bradshaw, 2000). L'activitat humana, principalment l'agricultura i ramaderia, ha modificat els paisatges europeus des de l'Holocè, i pràcticament no podem trobar en aquest continent sistemes naturals autènticament verges (Peterken, 1996). Per altra banda, el canvi climàtic provocat per l'home que estem vivint des de la revolució industrial, ja en l'Antropocè, també modifica les condicions ambientals i per tant els propis sistemes naturals de referència, de manera que la restauració dels ecosistemes ha d'assumir aquesta realitat i convertir-se també en una eina d'adaptació.

No obstant, aquesta divisió entre maneres d'entendre la restauració ecològica és excessivament simplista i esbiaixada. De fet, els propis principis de la restauració ecològica tal i com els fixa la pròpia SER (Clewell *et al.*, 2004) ja reconeixen que alguns ecosistemes, particularment en països en desenvolupament, han estat i estan gestionats de manera tradicional mitjançant pràctiques culturals sostenibles. En aquests ecosistemes existeix una reciprocitat entre aquestes pràctiques culturals i els processos ecològics, en la mesura que l'activitat humana enforteix la seva salut i sostenibilitat.

Al marge d'aquestes discussions més filosòfiques, podríem dir que els principis de la restauració ecològica, tal i com els fixa la SER, han estat acceptats pels principals organismes ambientals a nivell mundial (Cortina *et al.*, 2015). Segons aquests principis, es pot considerar un ecosistema com a restaurat si compleix els següents atributs:

1. L'ecosistema restaurat conté les espècies i l'estructura que podem trobar a l'ecosistema de referència (pristí o no).

2. L'ecosistema restaurat consta d'espècies autòctones fins al grau màxim factible. En ecosistemes culturals restaurats, es pot ser indulgent amb espècies domèstiques exòtiques i amb espècies ruderals i arvenses que se suposa que coevolucionaran amb les autòctones.

3. Tots els grups funcionals d'espècies necessaris pel desenvolupament i/o l'estabilitat contínua de l'ecosistema restaurat es troben representats, o si no, els grups que manquen tenen el potencial de colonitzar l'espai per mitjans naturals.

4. L'ambient físic de l'ecosistema (biòtop) restaurat té la capacitat de sostenir les espècies necessàries per a la contínua estabilitat o desenvolupament al llarg de la trajectòria desitjada.

5. L'ecosistema restaurat funciona amb normalitat, d'acord amb el seu estat ecològic de desenvolupament, i no hi ha senyals de disfunció.

6. L'ecosistema restaurat s'integra adequadament a la matriu ecològica i al paisatge, amb quals interactua mitjançant fluxos i intercanvis biòtics i abiòtics.

7. S'han eliminat o reduït les amenaces potencials del paisatge adjacent per a la salut i la integritat de l'ecosistema restaurat.

8. L'ecosistema restaurat té una capacitat suficient de recuperació com per aguantar les pertorbacions periòdiques i pròpies de l'ambient local, i que serveixen per mantenir la integritat de l'ecosistema.

9. L'ecosistema restaurat és autosostenible al mateix grau que el seu ecosistema de referència i té el potencial de persistir indefinidament sota les condicions ambientals existents. No obstant, la diversitat d'espècies, estructura i funcionament podrien canviar com a part del desenvolupament normal de l'ecosistema, i podrien fluctuar en resposta a esdeveniments d'estrès i de pertorbacions de major transcendència. Com qualsevol ecosistema ben preservat, la composició de les espècies i altres atributs d'un ecosistema restaurat podrien evolucionar a mesura que canvien les condicions ambientals.

En conseqüència, la restauració ecològica, quan s'implementa de manera eficaç i sostenible, contribueix a: i) protegir la biodiversitat; ii) millorar la salut i el benestar de les persones; iii) augmentar la seguretat alimentària i la provisió d'aigua; iv) augmentar el subministrament de béns, serveis i prosperitat econòmica; i v) la mitigació, resistència i adaptació al canvi climàtic. Per altra banda, els principis de la SER han estat reformulats recentment per tal d'incorporar millor els factors socioeconòmics i culturals que poden afectar de manera rellevant els resultats de la restauració. En aquests principis revisats es prioritza el fet d'involucrar en el procés de restauració el màxim d'agents socials (*stakeholders*), utilitzant en la mateixa mesura els coneixements científics, tradicionals i locals (Gann *et al.*, 2019).

Mancances i oportunitats de la restauració ecològica

Molts programes i projectes de restauració, així com diverses accions intersectorials tenen com a objectiu restaurar els ecosistemes degradats, sia pel seu valor intrínsec com a patrimoni natural, sia pel seu valor per a la provisió d'una gamma de serveis ecosistèmics (Fisher *et al.*, 2019). Com s'ha dit, un darrer exemple d'això és la declaració de la dècada 2020-2030 per part de les Nacions Unides com la Dècada per a la Restauració dels Ecosistemes (UN, 2019a). Tanmateix, molts d'aquests esforços no aconsegueixen els seus objectius (Ockendon *et al.*, 2018) i podem trobar molts exemples de restauracions fallides en diferents tipus de projectes de restauració (Kentula, 2000; Lockwood and Pimm, 2009; Zedler and Callaway, 1999). En aquest context, comprendre les limitacions efectives de la restauració pot donar resposta a les qüestions plantejades en diferents àmbits de la restauració, l'enginyeria ecològica i l'economia circular, la gestió de l'aigua i del paisatge (Fisher et al, 2019). Per altra banda, aquest avenç en la pràctica de la restauració, ha de permetre també avançar en la mitigació i adaptació al canvi climàtic, alhora que s'incrementa la seguretat alimentària. Tots aquests avenços tenen un impacte directe en la creació d'oportunitats per a les indústries i els sectors econòmics que depenen dels actius naturals i els serveis ecosistèmics que se'n deriven, la qual cosa s'hauria de traduir en una millora del benestar humà (Eklipse Secretariat, 2018).

Hilderbrand *et al.* (2005) apuntaven ja fa uns anys algunes mancances o idees errònies, el que ells van anomenar mites, sobre els quals s'havien dissenyat i executat molts projectes de restauració: simplificació excessiva dels objectius (*carbon copy*), focalització única en condicions físico-químiques (*field of dreams*), acceleració de la successió i el desenvolupament dels ecosistemes (*fast forward*), metodologies insuficientment validades (*cookbook*), tractament dels símptomes en comptes d'abordar el problema d'arrel (*command and control*). En general aquestes idees giren entorn d'una simplificació excessiva dels sistemes naturals i el seu funcionament per tal de facilitar el disseny i execució dels projectes de restauració, cosa que per altra banda és totalment comprensible i habitual en moltes disciplines, especialment en l'àmbit de la ciència aplicada.

Per altra banda, tot i que la SER fixa els atributs que ha de tenir un ecosistema restaurat correctament, el que caracteritza l'èxit de la restauració i com cal mesurar-lo és un tema de debat entre els membres de la comunitat científica (Wortley *et al.*, 2013; Crouzeilles *et al.* 2016), tot i que existeix un consens en què cal implementar procesos de seguiment de les restauracions i millorar els existents (Hagen and Evju, 2013; Halldórsson *et al.*, 2012; Suding, 2011, Fisher *et al.*, 2019). A la literatura científica podem trobar diferents mètodes per avaluar els atributs que ha de tenir un ecosistema restaurat, focalitzats principalment en la composició i estructura de la vegetació, la biodiversitat i els processos ecològics (Ruiz-Jaen and Aide, 2005; Wortley *et al.*, 2013). No obstant, la realitat és que en molts casos no es realitza una avaluació i un seguiment efectiu o prou complet dels treballs de restauració, la qual cosa constitueix una barrera per avançar en la pràctica de la restauració ecològica (González *et al.*, 2015; Suding, 2011). Per altra banda, quan es realitza un seguiment d'aquests projectes, en molts casos es tendeix a focalitzar-se en una descripció de les àrees restaurades, realitzada poc després de finalitzar els treballs de restauració (Suding, 2011), quan seria necessari un seguiment continu durant tot el procés de restauració (Allen *et al.*, 2002; Pander and Geist, 2013) i també un temps després.

En aquest context, els professionals que executen els projectes de restauració i en fan el seguiment demanen a la comunitat científica procediments pràctics basats en indicadors objectius (Beier *et al.*, 2017; Clewell and Rieger, 1997). Com s'ha apuntat anteriorment, l'existència de protocols clars i senzills d'aplicar facilitarien la implantació dels esquemes de seguiment i avaluació dels projectes de restauració, millorant la transferència d'informació i coneixement cap a altres projectes, així com la seva eficiencia econòmica (Nilsson *et al.*, 2015), i molt probablement es milloraria també la qualitat de les restauracions executades.

El 2018 la iniciativa EKLIPSE de la Unió Europea (http://www.eklipse-mechanism.eu/) va constituir un grup d'experts per tal de donar resposta a la pregunta formulada per la xarxa BIODIVERSA en relació a si la falta de coneixement és un dels factors limitants per a la restauració de la biodiversitat, les funcions i els serveis ecosistèmics (Eklipse Secretariat, 2018). Tot i que aquesta era la pregunta que platejava BIODIVERSA, el secretariat d'EKPLISE va realitzar una primera avaluació el 2017 i va decidir ampliar aquesta pregunta per tal de determinar quins eren els principals factors que limitaven l'efectivitat d'aquests plantejaments (Eklipse Secretariat, 2018). Després de gairebé un any de treball del grup d'experts, una revisió de literatura científica i gris, i un procés participatiu que ha implicat a 141 professionals del sector de 18 països europeus, el 2019 es va fer circular un primer esborrany del document que pretenia posar llum a aquesta qüestió (Fisher *et al.*, 2019). Segons aquest document de referència, la manca de coneixement és només una petita part de la llarga llista de barreres que limiten la pràctica de la restauració. Les barreres que s'han classificat com a més importants en el procés participatiu són: 1) baixa prioritat política per a la restauració, 2) manca de planificació, 3) falta de seguiment dels projectes, 4) finançament baix, 5) drets de propietat que impedeixen la implementació i, 6) conflicte d'interessos. Per contra, en la revisió bibliogràfica es van detectar altres barreres principals: avaluació dels projectes, planificació dels projectes, definició d'objectius, i polítiques i governança. En tot cas, aquest procés de consulta i revisió a fons ha permès detectar els factors clau que limiten la pràctica de la restauració ecològica a Europa, als quals es pot donar resposta majoritàriament mitjançant l'aplicació de pràctiques basades en la gestió co-adaptativa (Fisher *et al.*, 2019), entesa com la gestió conjunta a través de l'aprenentatge per l'acció (Cundill and Fabricius, 2009).

Aspectes clau en la restauració d'espais degradats

La restauració d'espais altament degradats, com poden ser els afectats per activitats extractives, segueix una seqüència bàsica en funció dels objectius que comprèn principalment: modelatge del terreny i construcció de la xarxa de drenatge, aportació i estabilització del sòl, i establiment de la vegetació i la fauna. Evidentment, caldria afegir tant els processos previs com els posteriors, com són el disseny i l'elaboració dels estudis preliminars, el procés de decisió dels objectius i la redacció del projecte, que haurien d'incloure en la mesura del possible processos participatius. També caldria incloure el seguiment de l'execució de la restauració al camp, la monitorització dels resultats i la implementació de mesures correctores, així com el manteniment dels espais en procés de restauració i també dels ja restaurats.

Si ens centrem en les pròpies actuacions restauradores, la reposició d'un sòl adequat als objectius de la restauració és un dels aspectes clau per garantir l'èxit de la revegetació. En aquest sentit, el decapatge previ a l'impacte és un dels processos que permeten disposar de sòl de qualitat per a la restauració, sempre que això sigui possible, com és el cas de noves activitats extractives o obres d'infraestructures viàries, per exemple. Ara bé, en molts casos no es disposa de prou sòl de decapatge per a la restauració de tota la superfície afectada, o bé no és de prou qualitat. Per suplir aquesta mancança s'utilitzen sovint materials minerals de rebuig com a formadors de sòls, especialment en el context de les activitats extractives. Aquests materials solen presentar granulometries desequilibrades i deficiències de nutrients que cal corregir fent mescles amb altres terres o materials, i amb la incorporació d'esmenes orgàniques o adobs (Alcañiz *et al.*, 2009).

Tecnosols: economia circular, valorització de residus i restauració d'espais degradats

Les activitats econòmiques vinculades a l'economia verda (*Green economy*) s'han incrementat de manera important en els darrers anys (UN, 2019b). L'economia circular estaria dins del paraigües de l'economia verda, promovent un ús més sostenible dels recursos, així com la transformació dels residus d'una activitat econòmica en subproductes d'una altra, posant un èmfasi especial en les dinàmiques intersectorials i cooperatives (D'Amato *et al.*, 2017). L'ecodisseny, la innovació ecològica, la prevenció de residus i la reutilització de matèries primeres poden suposar un estalvi net per a les empreses de la UE de fins a 600 milions d'euros. Les mesures addicionals per augmentar l'aprofitament dels recursos i la valorització dels residus en un 30% pel 2030 podrien augmentar el PIB prop de l'1%, alhora que es crearien 2 milions de llocs de treball addicionals (EC, 2019).

A la UE es produeixen cada any aproximadament entre 120 i 140 milions de tones de bio-residus (*bio-waste*), que corresponen a uns 300 kg per ciutadà de la UE i any (JRC, 2011). Segons la Directiva marc de residus (EC, 2008), els bio-residus corresponen a residus procedents de jardins i parcs, residus d'aliments de cuines de les llars, restaurants, càterings i locals comercials, així com els residus procedents de plantes de processat d'aliments. No inclou en aquesta categoria residus forestals ni agrícoles, fems, fangs de depuradora ni altres residus biodegradables com ara tèxtils naturals, paper o fusta processada. També exclou aquells subproductes de producció d'aliments que mai es converteixen en residus. Pel que fa específicament als fangs d'estacions depuradores d'aigües residuals urbanes, a tot Europa se'n van produir el 2017 prop de 3M de tones de matèria seca (EC, 2020), i unes 120.000 a Catalunya (ACA, 2018).

A tot Europa prop del 40% dels bio-residus es dipositen en abocadors (fins al 100% en alguns estats membres) (EC, 2010), una pràctica que no està d'acord amb els principis rectors de la política de gestió de recursos sostenibles de la UE, és a dir, la "jerarquia de residus" que hauria de fonamentar totes les polítiques nacionals al respecte (JRC, 2011). Pel que fa als fangs de depuradora, la majoria s'apliquen al sòl un cop digerits o compostats, i una part s'incineren (EC, 2020), si bé en els darrers anys s'han desenvolupat també sistemes alternatius com la piròlisi i gasificació per millorar el tractament tèrmic i la valorització energètica dels fangs (Manara and Zabaniotou, 2012; Samolada and Zabaniotou, 2014). D'acord amb una perspectiva de l'economia circular global, els objectius actuals de la Unió Europea fixen que el 65% de tots els residus municipals produïts es reciclin abans del 2030, i només un 10% acabin en abocadors (EC, 1999).

Per facilitar la gestió final dels residus s'utilitzen diferents tractaments mecànics i biològics (Seyring *et al.*, 2015), tot i que globalment encara s'està lluny de poder aplicar totalment el concepte d'economia circular en aquest àmbit. Quan els esquemes de separació en origen no estan completament implementats, el tractament dels residus mitjançant plantes de tractament mecànic-biològic és una bona opció per incrementar el percentatge de residus reciclats i reduir la disposició en abocador (ETC/SCP, 2014) si bé requereix una major inversió en infraestructures. Les plantes de tractament mecànic-biològic intenten separar mecànicament els components biodegradables i no biodegradables (Donovan *et al.*, 2010). Els components no biodegradables s'envien després per reprocessar-los, incinerar-los o es dipositen en un abocador, mentre que els components biodegradables es sotmeten a compostatge, generant així un producte similar al compost anomenat bioestabilitzat (*bioestabilized or compost-like-output*), o bé a digestió anaeròbia generant aleshores el que s'anomena digest (*digestate*).

Les fraccions orgàniques de residus sòlids municipals són heterogènies quant a composició i procedencia. Per tant, la gestió sostenible d'aquests residus representa un repte si es vol alinear amb els principis de l'economia circular, que ha de reduir la incineració i la disposició en abocadors (Abdullahi *et al.*, 2008). Si la recuperació i el tractament mecànic i biològic dels bio-residus es realitza de forma adequada, els productes finals orgànics biodegradables resultants poden complir els requisits de qualitat necessaris per a ser aplicats al sòl. No obstant, cal estar especialment amatent al seu contingut en contaminants orgànics, metalls pesants i impureses (plàstics, vidres, metalls). Pel cas dels fangs de depuradora, també existeix un potencial molt important per utilitzar-los al sòl, sia com a esmena orgànica o com a fertilitzant, degut a l'elevat contingut matèria orgànica i macronutrients, especialment nitrogen i fòsfor (Alcañiz *et al.*, 2009). Tot i així, en alguns casos l'elevat contingut en metalls pesants, contaminants orgànics i altres pot fer més aconsellable la incineració o el dipòsit en abocador. Per altra banda, darrerament s'està posant el focus en el contingut en micro(nano)plàstics, ja que es calcula que l'aplicació de fangs en sòls agrícoles podria ser una de les principals vies d'entrada d'aquests contaminants al sòl (Hurley and Nizzetto, 2018).

Tornant a les repercussions per a l'economia és important remarcar, tal com ja s'ha fet anteriorment, que la restauració d'espais degradats pot constituir un impuls pels sectors econòmics que s'hi vinculin, obrint un ampli ventall d'oportunitats i línies de negoci (Sundstrom, 2011). Un exemple d'aquesta vinculació, en la línia dels principis de l'economia verda i l'economia circular, pot ser l'aprofitament de residus orgànics i minerals per a les tasques de restauració d'espais degradats, tal com es dona sovint en la construcció dels anomenats tecnosols (Schad and Dondeyne, 2017; Mosquera-Losada *et al.*, 2017). Els tecnosols, que també podríem definir com a "sòls tècnics a la carta", són una font viable de substrats quan la disponibilitat de sòls naturals adequats per a la restauració és limitada (Watkinson *et al.*, 2017), per la qual cosa els podem considerar com una ecotecnologia disponible per resoldre alguns dels problemes de la gestió dels sòls a l'Antropocè (Leguédois *et al.*, 2016). L'ús de residus orgànics (fangs EDAR, fems, composts, etc.) per a la construcció de tecnosols és una pràctica força utilitzada en la restauració de mines, infraestructures de transport o abocadors (Asensio *et al.*, 2013; Lomaglio *et al.*, 2017; Watkinson *et al.*, 2017), situacions en què és habitual la falta de sòl natural fèrtil. En aquests casos l'ús de residus orgànics té com a l'objectiu accelerar la colonització biològica d'un substrat relativament inert. Les aptituds com a esmena d'aquests residus orgànics depenen, entre d'altres, de la seva capacitat per contribuir a l'increment del contingut d'humus (matèria orgànica estable) i al manteniment del balanç de nutrients (Magdoff and Weil, 2004), el qual depèn en major mesura de la degradabilitat de la matriu orgànica que contenen (Rumpel *et al.*, 2002). Per altra banda, existeixen bastants residus orgànics i inorgànics disponibles per a ser utilitzats com a esmenes de sòls, però no tots valen pels mateixos objectius (Pérez-Gimeno *et al.*, 2019).

Situat en l'àmbit de l'economia circular i la restauració d'espais degradats, l'ús de fangs d'estacions depuradores d'aigües residuals urbanes (fangs EDAR, o més coneguts com a fangs de depuradora) en la creació de tecnosols és una via de gestió que contribueix a la valorització d'aquests residus (Sopper, 1993; Alcañiz *et al.*, 2009). A més, degut al seu baix cost, és una opció viable per a la restauració a gran escala (Pérez-Gimeno *et al.*, 2019). En aquest context, els fangs de depuradora s'apliquen una única vegada, a una dosi moderada, per actuar tant com a adob com a esmena orgànica, i generalment es barregen amb residus miners o terres inerts (barreges d'horitzons A i/o B, i inclús C dels sòls), que aporten la fracció mineral. Per tant, l'ús de fangs de depuradora és una opció interessant per valoritzar els residus minerals en els processos de creació de tecnosols, a causa de les seves propietats fertilitzants (Van-Camp *et al.* 2004) i els seus efectes positius en l'estabilitat dels agregats del sòl (Caravaca *et al.*, 2002; Ojeda *et al.*, 2008), la retenció d'aigua al sòl (Ojeda *et al.*, 2011, 2010) i el desenvolupament de la vegetació (Ortiz & Alcañiz 2006; Moreno-Peñaranda *et al.* 2004). Tanmateix, com s'ha apuntat anteriorment, l'aplicació de fangs de depuradora requereix d'una supervisió a causa del risc que pot provocar l'exportació d'elements solubles durant les primeres setmanes després de l'aplicació, així com del que es deriva del seu contingut en metalls pesants, contaminants orgànics persistents o microplàstics (Carabassa *et al.*, 2010; Düring and Gäth, 2002).

Els fangs de depuradora, com a residus derivats de biomassa, poden contribuir directament al segrest de carboni al sòl mitjançant la seva estabilització (p. e. formació de complexos amb les argiles) i, indirectament, a través de l'augment de la producció de biomassa vegetal (Ojeda *et al.*, 2015). Tanmateix, l'estabilitat relativament baixa dels fangs de depuradora (Mattana *et al.*, 2014) pot causar efectes transitoris en els *pools* de matèria orgànica del sòl, ja que el segrest de carboni orgànic a mig i llarg termini es basa més en les aportacions generades per les restes vegetals que en el propi carboni aportat pels fangs (Ojeda *et al.* 2015). L'increment del contingut de matèria orgànica al sòl, al seu torn, pot contribuir a d'altres serveis ecosistèmics rellevants com la millora de la producció de matèries primeres (per a combustibles, materials de construcció, etc.), el reciclatge de nutrients, la regulació climàtica o la millora del sòl com a hàbitat pels organismes (Baveye *et al.*, 2016). Tanmateix, encara se sap poc sobre l'eficàcia dels Tecnosols en termes d'emmagatzematge de carboni (Ojeda *et al.* 2015), especialment a mig i llarg termini.

Els processos pedogènics que es produeixen als Tecnosols són similars als dels sòls naturals (Leguédois *et al.*, 2016), tot i que els components utilitzats poden influir fortament en la seva evolució i la seva capacitat de comportar-se com a sòl, i per tant, de proporcionar els serveis dels ecosistemes associats. Tot i això, solen tenir una ràpida evolució en comparació amb els sòls naturals, la qual cosa inclou també l'activitat biològica (Leguédois *et al.*, 2016). Això és especialment rellevant, ja que la fauna i la vegetació del sòl són factors clau per a la prestació de serveis ecosistèmics (Tate, 2005), essent l'activitat biològica el principal factor que controla la pedogènesi del sòl (si més no als horitzons superficials), per exemple a través del seu paper en la dinàmica del carboni i el nitrogen (Frouz *et al.*, 2013) i en l'acceleració dels processos de meteorització.

El sòl com a proveïdor de serveis ecosistèmics

La sobre-explotació del sòl ha donat lloc al col·lapse de civilitzacions i canvis de règim, reduint irreversiblement la capacitat productiva d'alguns ecosistemes (Birgé *et al.*, 2016; Walker and Salt, 2006). Aquests esdeveniments han estat reveladors del principal servei ecosistèmic que proveeix el sòl per a una població humana cada cop més nombrosa i consumidora de recursos: la producció d'aliments, fibres naturals i combustibles (Reicosky, 2017). La degradació del sòl, com a conseqüència de pràctiques agrícoles intensives o no apropiades, redueix eventualment el rendiment dels cultius, i pot donar com a resultat l'increment de l'erosió, la reducció del subministrament de nutrients a través de la mineralització progressiva de la matèria orgànica, i la degradació de la seva estructura, la qual cosa pot agreujar l'estrès per sequera (Lal, 2009). Un altre servei ecosistèmic que proporciona el sòl és la regulació hídrica via la infiltració d'aigua i el seu emmagatzematge a la matriu porosa. La regulació de pertorbacions (p. e. el control d'inundacions) és un altre dels serveis proporcionats pels sòls saludables en els sistemes naturals, amb un valor estimat d'1,8 x 10^{12} dòlars americans a l'any (Costanza *et al.*, 1997). També la conversió de zones humides en camps de conreu i el segellament del sòl per la urbanització de terrenys agrícoles i forestals redueixen o eliminen aquest servei de regulació, incrementant així la vulnerabilitat enfront a possibles inundacions (Reicosky, 2017).

En definitiva, els sòls són responsables d'una varietat de processos naturals coneguts com a funcions del sòl, que són la base per a la provisió dels anomenats serveis ecosistèmics (Adhikari and Hartemink, 2016). Mentre que les funcions del sòl fan referència als beneficis del sòl per a totes les espècies dels ecosistemes, inclosos els humans, el terme servei ecosistèmic es refereix específicament als beneficis pels humans (Adhikari and Hartemink, 2016; Baveye *et al.*, 2016). A causa de les dificultats per a l'avaluació directa dels serveis dels ecosistemes, els components i les funcions del sòl s'han utilitzat com a indicadors de la prestació d'aquests serveis (Baveye *et al.*, 2016). Per exemple, el contingut de carboni orgànic s'ha utilitzat com a indicador del segrest de carboni, mentre que la biodiversitat de grups particulars d'organismes s'ha relacionat amb la funció d'hàbitat del sòl. La prestació de serveis dels ecosistemes és la raó principal de la rehabilitació del sòl, a causa de la seva connexió directa amb el benestar humà (UNEP, 2008). En l'última dècada, científics, mitjans de comunicació i agències governamentals han adoptat amb èxit el concepte de serveis ecosistèmics impulsat per la publicació de l'Avaluació d'Ecosistemes del Mil·lenni (2005). Des d'aleshores aquest enfocament s'ha inclòs en una gran varietat d'aplicacions, des de la planificació del territori i l'avaluació d'alternatives d'ordenació del sòl, fins a l'avaluació de l'èxit de la restauració d'espais degradats. De fet en els nous principis de la SER ja es posa el focus de manera molt clara en la provisió de serveis ecosistèmics, anant fins i tot una mica més enllà, al buscar la maximització dels beneficis socials dels projectes de restauració mitjançant la implicació de tots els actors rellevants en el procés, especialment la població local. Aquest és sens dubte un aspecte que des de l'àmbit de la restauració ecològica s'haurà d'afrontar adoptant una òptica pluridisciplinar (Samonte *et al.*, 2017).

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2. Objectius i abast del treball

El present treball pretén contribuir a la generació de coneixement que sigui útil per a la pràctica de la restauració d'espais degradats. És a dir, la finalitat general és desenvolupar mètodes per facilitar, fomentar, optimitzar, en definitiva millorar la implementació i el seguiment de la restauració d'aquests espais, especialment en aquells casos més complexos, com poden ser activitats extractives, infraestructures de transport terrestre o abocadors, en què cal refer tots els compartiments de l'ecosistema.

Aquesta fita general la podem concretar en dos grans objectius. Per una banda, contribuir a la generació de coneixement en relació a noves tècniques, procediments o materials, aplicables principalment a la rehabilitació de sòls o a la creació de tecnosols. Per l'altra, innovar en l'avaluació i seguiment de la restauració ecològica, incloent és clar noves tècniques o materials, però també des d'un punt de vista més global i conceptual, considerant el conjunt del procés i el resultat de la restauració, així com els indicadors necessaris per dur-la a terme. Per tant, es pretén abastar diferents aspectes de la restauració o rehabilitació d'espais degradats, des de diferents escales: laboratori, hivernacle, parcel·la de camp, talús o vesant, i paisatge o conjunt de zones restaurades en el seu entorn, des de la voluntat de generar eines útils per a la pràctica de la restauració ecològica. En aquest sentit, la major part dels treballs que configuren aquesta tesi, no tant sols han estat publicats en l'àmbit científic internacional, sinó que en alguns casos també s'han publicat en forma de manuals i protocols per facilitar-ne i fomentar-ne la seva implementació.

Bona part dels objectius encaixen en el marc de l'economia circular i el reciclatge de residus, al centrar-se en la valorització dels fangs de depuradora o els bioestabilitzats. Si bé els primers ja fa anys que s'apliquen al sòl, principalment en l'agricultura, no hi ha gaires protocols que estableixin unes pautes clares per a la seva utilització en rehabilitació d'espais degradats i que n'avaluïn el seus efectes a mitjà termini. Pel que fa als bioestabilitzats hi ha molt poca literatura científica i experiència tècnica sobre el seu ús i gestió, tot i què es produeixen en quantitats molt grans i generen importants interrogants pel que fa a la seva valorització en el context de l'economia circular, alguns dels quals s'afronten en aquest treball. També s'incideix en l'aprofitament de determinats residus miners per a la creació de tecnosols. Es completa la tasca seleccionant indicadors i avaluant l'èxit de la restauració de les activitats extractives, que representen un dels casos de major complexitat.

Aquest marc d’objectius generals es concreta en una sèrie d’objectius més específics que corresponen als diferents capítols de la tesi:

- Comprovar la utilitat del protocol normalitzat OCDE-217 basat en la respiració del sòl per determinarel risc de toxicitat per metalls pesants, així com per la monitorització de l’eficàcia de processos de rehabilitació de sòls quan s’apliquen esmenes orgàniques.

- Avaluar els efectes a curt termini, en el sòl i la vegetació, de l'ús de fangs de depuradora en la construcció de tecnosols en condicions mediterrànies.

- Avaluar els efectes a mig termini (10 anys), en dos serveis ecosistèmics clau (segrest de carboni i funció d'hàbitat), de l'ús de fangs de depuradora en la construcció de tecnosols en condicions mediterrànies.

- Determinar les limitacions i les oportunitats de l’ús de bioestabilitzats i digests procedents de la fracció orgànica de la resta (residus no recollits selectivament) per a la construcció de tecnosols en obres de restauració ambiental.

- Seleccionar indicadors clau del procés de restauració i establir un protocol per a l’avaluació i seguiment de la restauració d’activitats extractives.

3. Determination of EC_{50} values for Cu, Zn and Cr on Microorganisms Activity in a Mediterranean Sandy Soil

Aquest capítol té com a principal objectiu determinar la validesa del protocol OCDE-217, basat en la mesura de la respiració del sòl, per tal de detectar i quantificar la toxicitat de tres dels metalls pesants més abundants en fangs de depuradora urbana. També s'estudia com afecta a aquestes mesures l'addició d'una matriu orgánica fàcilment mineralitzable. Aquest capítol ha estat publicat a la revista CLEAN Soil Air and Water (Carabassa, V., Domene, X., Ortiz, O., Marks, E.A.N., Alcañiz, J.M. (2017) Determination of EC50 Values for Cu, Zn, and Cr on Microorganisms Activity in a Mediterranean Sandy Soil. https://doi.org/10.1002/clen.201700617).

Abstract

Ecotoxicity of three potentially toxic elements (PTEs) (Cu, Zn and Cr) in a slightly acidic sandy soil was tested using the soil respiration test (OECD-217) in order to determine EC_{50} values for the carbon transformation activity of microorganisms. Addition of an organic amendment of *Populus* leaves was also crossed with metal spiking in order to investigate possible interaction with metal toxicity. Soil respiration was measured at 1 (1d) and 28 days (28d) after the soil spiking with the PTEs to assess short-term effects on soil microbial activity. Of the three metals tested, Cu showed the highest toxicity at the longest exposure times (28d) and Zn showed a strong inhibitory effect in the short-term (1d), even though later toxicity diminished significantly. Cr was the least toxic studied PTE. Organic amendment outweighs any adverse effects of these metals, increasing soil respiration, even in the treatments with high doses of metals.

Keywords: effective concentrations, heavy metals, metal toxicity, soil microorganisms, soil organic matter

Abbreviations: EC_{50}**,** Median effective concentration; OECD, Organization for Economic Co-operation and Development; CEC, Cation-exchange capacity; SIR, Substrate-induced respiration

Introduction

Diffuse contamination by heavy metals in soils is an increasing problem worldwide. Historically, metal pollution has caused adverse effects on the environment and human health, exemplified by disease outbreaks in Minamata due to Hg [1, 2] and in Itai-Itai due to Cd [3]. Currently, sources of diffuse soil contamination by trace metals are increasing in number, associated with inappropriate agricultural practices and repeated application of urban and industrial wastes. These sources have created major problems (inhibition of soil microorganism activity, inhibition of soil microbial growth and reproduction, inhibition of litter decomposition, reduced crop yields) in European soils [4].

Heavy metals are well known to be toxic to most organisms when present in excessive concentrations [5], including to soil microorganisms which play an important role in soil functions and associated ecosystem services [6, 7]. Toxic effects of pollutants on soil organisms are difficult to estimate if only the total content in soils is known [8]. Ecotoxicological methods are required to reveal the actual toxicity of soil pollutants, associated with the actual bioavailable fraction which is modulated by interactions among toxins and between toxins and soil components [9]. Ecotoxicological tests provide a link between chemical monitoring and risk assessment of potential impacts on humans and biota [10], as well as datasets for the derivation of maximum safe environmental concentrations. In this context, the development of microbial populations under chemical stress conditions constitutes an important research topic in ecotoxicology. For this reason, there is an obvious interest in determining the relation between chemicals' concentrations and the effects on populations [11].

Stabilization or inactivation of heavy metals in soil can be achieved by adding amendments (lime, apatite, zeolites, organic materials or Mn and Fe oxides, etc.) that are able to absorb, complex or (co)precipitate trace elements. The application of organic by-products has been also demonstrated to be useful for contaminant immobilization [31], with some advantages over other methods such as low cost, its 'in situ' applicability and its low environmental impact [12]. Organic amendments are considered especially effective in Cr stabilization, while in the case of other trace elements (As, Cu, Zn, and Pb) they may have both positive and negative effects [29].

Ecotoxicity tests are conducted by exposing organisms to a dose gradient of toxic compounds under controlled experimental conditions [13]. An increasing amount of evidence demonstrates that soil microorganisms are more sensitive to heavy metals (causing biotic stress) as compared to soil fauna or plants [5]. However, sensitivity ranges widely between species and this imply changes in the microbial diversity promoted by ecotoxic effects. These changes in the microbial diversity imply changes in the structure of microbial ecosystems [14, 15]. Despite setbacks, including limited long-term ecological relevance [5], several methods based on microbial activity and measures of biomass enable the assessment of toxic effects on soil microorganisms; namely, these methods are microbial biomass by fumigation-extraction [16], microbial activity by the substrate induced respiration [17] and basal respiration [18], microbial enzymatic activities such as nitrification [19], dehydrogenase activity [20], glucosidase activity [21], phosphatase activity [22], catalase activity [23], arginine ammonification [24] and carbon transformation test [25] using Biolog® system.

Some older studies are of limited use in terms of comparability because they did not employ standard methods [7]. Despite the large body of research on metal toxicity to soil microorganisms and microbial processes, the amount of available data is always insufficient and the interpretation too uncertain to establish risk-based thresholds [26]. In this context, the Organisation for Economic Co-operation and Development (OECD) defined a methodology to evaluate the effect of crop protection chemicals on the C-transforming activity carried out by soil microorganisms (OECD-217) [6], that has not been tested by heavy metals.

The main aim of this study is to validate the OECD-217 standardized protocol for testing heavy metals toxicity. The short-term toxicity of Cu, Zn and Cr (with and without organic amendment) was assessed by the OECD-217 standardized protocol, aiming to provide additional short-term ecotoxicological data on soil microbial activity of three heavy metals which is useful for the derivation of thresholds for heavy metal contents in soils.

Material and methods

The ecotoxic effects of Zn, Cr and Cu were assessed as effects on glucose-induced respiration rates according to the OECD Guideline 217 [6], based on measurements of CO_2 evolution following the addition of glucose. This was assessed in a natural soil relatively poor in organic matter. The activating or inhibiting effect of the addition of a natural organic matter source (fresh plant leaves) on the toxicity of these three metals was also taken into consideration.

Test soil

The soil corresponded to the fine fraction (<2 mm) of the A horizon (0-20 cm) of a *Eutric arenosol* developed from weathering of granodiorite in the Prades mountains (Tarragona, NE Iberian Peninsula). The topsoil layer (20 cm) was collected for the preparation of the tests. Soil was analysed by current standard methods: particle size was determined by sedimentation (Robinson pipette), organic carbon content by wet oxidation in acid dichromate, total nitrogen using Kjeldahl method, microbial-C by fumigation-extraction method, CEC after ammonium acetate cation displacement, electrical conductivity in a 1:5 w:v extract, and pH in a 1:2.5 w:v water suspension. A summary of the main analytical characteristics of soil are shown in Table 3.1. The soil had a sand fraction of about 70%, ensuring a homogeneous spiking with metals and mixing of the organic matter source. The high C:N ratio indicates a low degree of native organic matter transformation due to the presence of *Cistus laurifolius L.* which has a high leaf phenol content [27]. The soil was moderately acidic but pH values and low electrical conductivity are unlikely to limit microbial activity. Due to the low CEC, high bioavailaibility of metals might be expected in this soil. Overall, the soil properties agreed with the requirements of the OECD-217 procedure, except by the organic carbon content (2.6 $g \cdot kg^{-1}$), which is lower than the recommended value (5-15 $g \cdot kg^{-1}$).

Table 3.1. Main soil properties.

Parameter	Units	Value
Coarse sand (2000-200 µm)	%	60.7
Fine sand (200-20 µm)	%	7.4
Silt (20-2 µm)	%	22.2
Clay (<2 µm)	%	9.7
$CaCO_3$	%	0
Organic carbon	%	0.26
Microbial carbon	$mg \cdot kg^{-1}$	108.2
C microbial / C organic	%	4.16
N Kjeldahl	$g \cdot kg^{-1}$	0.11
C/N	-	23.6
CEC	$cmol \cdot kg^{-1}$	7.3
Electrical conductivity (1:5 w:v)	$dS \cdot m^{-1}$	0.26
pH-water (1:2.5 w:v)	-	6.3

After collection, soil was sieved to 2 mm and stored at 4 °C. Before beginning microbial tests soil was moistened to 10% of water holding capacity (WHC) and kept in dark at 21 °C for three days.

Treatments: heavy metal spiking and organic amendment

Heavy metals were added to soil individually in a water solution prepared to provide 50% of the soil's maximum water holding capacity, applied as dissolved salts of $ZnCl_2$, $CuSO_4 \cdot 5H_2O$, and $CrCl_3 \cdot 6H_2O$.

Following the protocol, preliminary assays were conducted in which each metal was incorporated at 7 logarithmically-increasing concentrations, between 0.001 and 1000 $mg \cdot kg^{-1}$ in the case of Zn, and between $4.5 \cdot 10^{-4}$ and 450 $mg \cdot kg^{-1}$ for Cu and Cr. In the definitive test, the heavy metals were incorporated within the range of concentrations where respiration inhibition was observed in the preliminary assays.

For the treatments with organic amendment heavy metals were spiked after adding fresh organic matter to soil at a rate of 350 $g \cdot kg^{-1}$. This rate is very high in order to ensure a clear effect of the amendment. Fresh organic matter consisted of green *Populus nigra* L. leaves, washed with distilled water and dried at

40°C until constant weight. Once dried, leaves were ground and sieved to <200 µm.

A summary of the treatments is provided in Table 3.2.

Table 3.2. Abbreviations and description of treatments.

Acronym	Treatment
Zn OM-	Addition of $ZnCl_2$
Zn OM+	Addition of $ZnCl_2$ and *Populus* leaves
Cu OM-	Addition of $CuSO_4 \cdot 5H_2O$
Cu OM+	Addition of $CuSO_4 \cdot 5H_2O$ and *Populus* leaves
Cr OM-	Addition of $CrCl_3 \cdot 6H_2O$
Cr OM+	Addition of $CrCl_3 \cdot 6H_2O$ and *Populus* leaves

Soil incubation and CO_2 measurements

Three replicates were prepared per metal concentration and treatment (OM- and OM+), each consisting of a 1.2 L plastic container filled with 300 g of soil. All the replicates were incubated in the dark at 21°C for 28 days. To prevent anoxic conditions, the containers were periodically aerated three times weekly. Soil moisture was checked weekly gravimetrically and replaced as necessary with deionized water.

The CO_2 produced by glucose-induced respiratory activity was measured by utilizing NaOH traps [28] for 12 consecutive hours after glucose addition, according to OECD-217 procedure. The amount of glucose required to achieve a maximum respiratory response in this soil had been previously determined using a series of concentrations of glucose (2, 5 and 8 $g \cdot kg^{-1}$). At the beginning of the test glucose was homogeneously mixed with the soil and respiration measurements were carried out at the beginning of the incubation time (1d) and the end (28d). At the end of the test, electrical conductivity and pH were measured.

Effective concentrations estimation

A concentration-response curve was constructed using the response values expressed as percent of the response in controls (soils with 0 $mg \cdot kg^{-1}$ of added heavy metals). The concentration–response curves were adjusted to a regression model (logistic or Gompertz models), chosen according to best fit with the available data. These were then used for the estimation of effective concentrations for each metal and incubation time (1d and 28d). The non-linear estimation module of Statistica 6.0 software (StatSoft ®) was used for this purpose.

Results

Preliminary assays

As expected, the addition of fresh organic matter (OM+) increased glucose-induced soil respiration. Treatments without organic matter addition (OM-) presented a maximum respiration rate slightly lower than 9 mg C-CO_2 ·kg^{-1}·h at day 1 (Cu OM- day 1), whereas the maximum for OM+ soil was upper than 130 mg C-CO_2·kg^{-1}·h for the same day (Cu OM+ day 1) (Figure 3.1). In OM- treatments microbial glucose-induced soil respiration was effected by added Zn, Cu and Cr in the range of studied concentrations, the responses following a lognormal distribution.

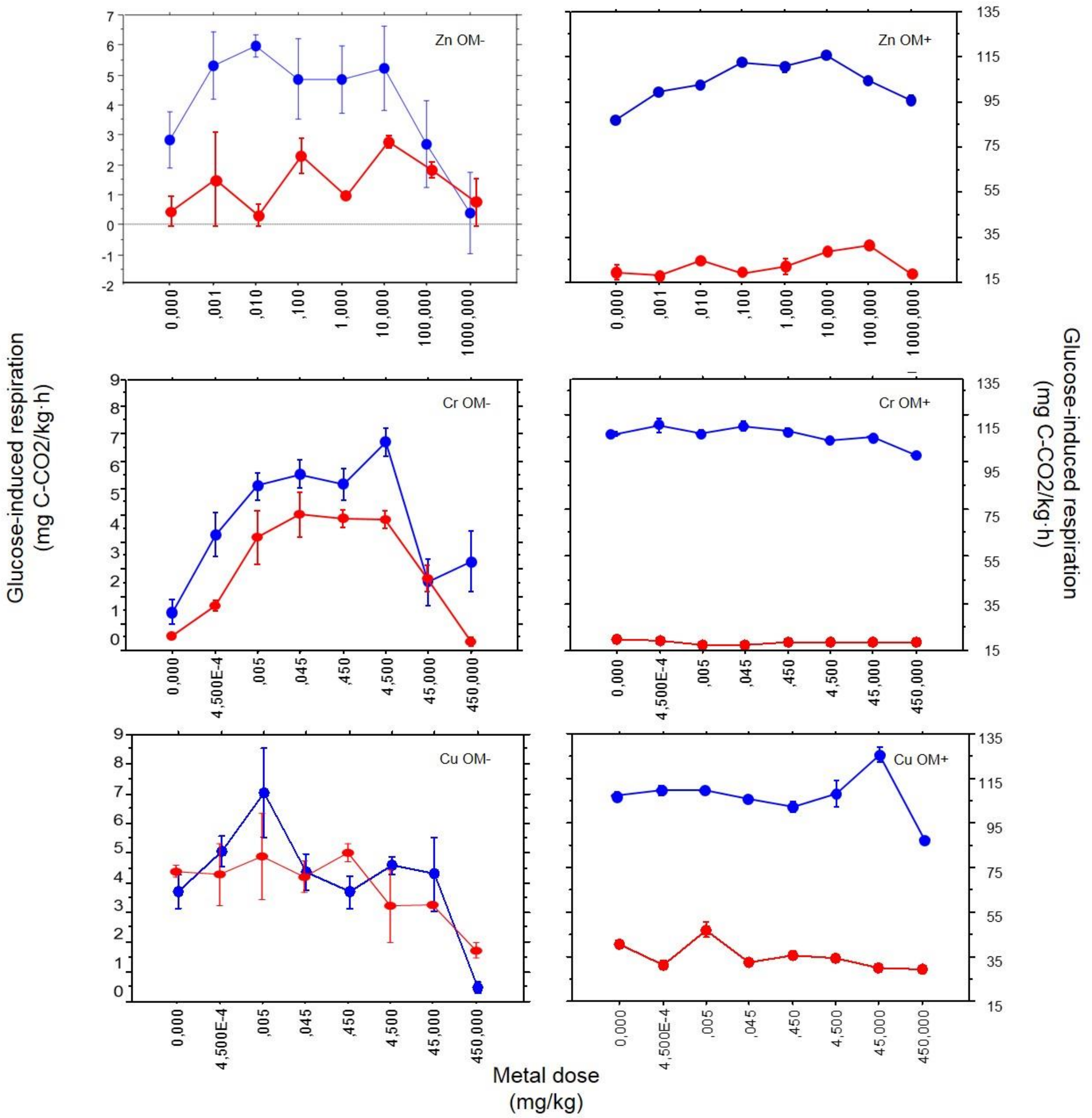

Figure 3.1. Changes of relative SIR (Substrate-induced Respiration) on the preliminary assay on the soil polluted with increasing concentrations of Zn, Cu and Cr, with (OM+) and without (OM-) organic amendment. Measures relative to day 1 are represented in blue, and to day 28 in red. Bars show standard error. Dose is presented in logarithmic scale.

According to the protocol, when results from tests are evaluated and the difference in respiration rates between the treatment and control is equal to or less than 25% at any sampling time after day 28, the tested product can be evaluated as having no long-term influence on carbon transformation in soils. We observed little negative effect on soil respiration at low levels of pollution in the different preliminary assays for all the studied metals. In addition, stimulating effects at low concentrations of Zn were observed. However, this effect was not observed in OM+ treatments.

In the range of studied concentrations inhibitory effects were observed at 1d and 28d in the OM- soil polluted with Cu or Zn. Regarding the OM+ treatments toxicity was not observed even though respiration was lower at high doses. In the case of the soil polluted with Cr (OM+ and OM-), no significant toxicity was detected in the range of studied doses, and in the OM- soil practically all Cr doses had soil respiration higher than the control, statistically significant at 28d measurements.

In light of the above, preliminary assays showed the existence of relevant toxic effects only in absence of organic matter amendment. Particularly, inhibition was detected in the case of Zn OM- (1d), Cu OM- (1d) and Cu OM- (28d). In the preliminary assays carried out with soil polluted with Zn, an inhibitory effect was observed at 28d when the reference soil had not been amended with organic matter. However, the large variability obtained after 28 days of incubation suggested the need to undertake definitive assays to confirm these results. In these cases the definitive assays were carried out using the same range of concentrations from which toxicity was detected in the preliminary assays. The assays corresponding to treatments polluted with Cr (Cr OM+, Cr OM-), soil polluted with Cu or Zn and amended with organic matter (Cu OM+, Zn OM+) did not show toxicity and therefore the definitive assays were not carried out.

Definitive assays

The dosage ranges used in the definitive series were established according to the results obtained in the preliminary assays (Table 3.3). Note that the doses range used in the preliminary assays was different for Cu and Zn.

Table 3.3. Range of concentrations used in the definitive series for the Zn OM- and Cu OM- treatments. See abbreviations meaning in Table 3.2.

Zn OM- ($mg \cdot kg^{-1}$)	Cu OM- ($mg \cdot kg^{-1}$)
0	0
1	100
3	147
10	215
32	316
100	464
316	681
1000	1000

Definitive assays showed that soil respiration decreased as metal dose increased in all assays. The Zn series showed a pronounced toxicity at 1d and 28d (Figure 3.2), whereas the effect at 28d was more pronounced. The same pattern was found for Cu series (Figure 3.2).

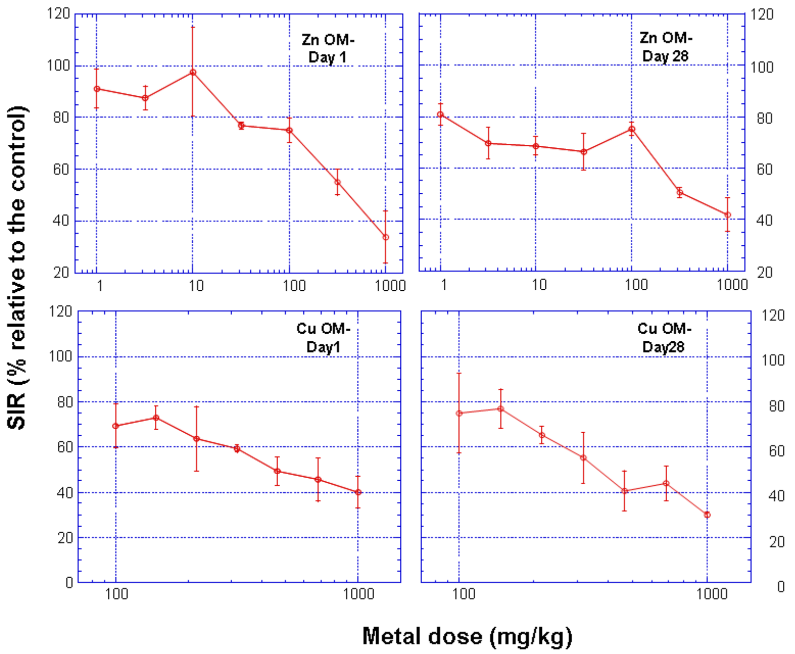

Figure 3.2. Changes of relative SIR (Substrate-induced Respiration) on definitive assay for the soil polluted with increasing concentrations of Zn and Cu. Bars show standard error. Dose is presented in logarithmic scale.

Effective concentrations

Regression models were applied to calculate effective concentrations. The models with the best fits were the Gompertz model for Zn treatments, and the logistic model for Cu treatments. The significance was high for the two models ($p<0.001$). EC_{50} (Median Effective Concentration) values obtained with the regression models presented different exposure time patterns for Cu and Zn (Table 4.3), specifically that Zn toxicity was lower with increased exposure time (lower EC_{50} at 28d) whereas Cu toxicity had the opposite trend (higher EC_{50} at 28d).

Table 4.3. EC_{50} values and 95% confidence interval (in brackets) in the 1 day incubations (1d) and the 28 days incubations (28d).

Metal	1d-EC_{50} ($mg \cdot kg^{-1}$)	28d-EC_{50} ($mg \cdot kg^{-1}$)
Zn	529 [257, 1089]	1285 [552, 2993]
Cu	958 [301, 3051]	517 [115, 2325]

Discussion

When metal toxicity data for soil microbial processes and populations cited in the literature is summarized, an enormous variability becomes apparent [5, 28, 29]. When representing acute and chronic effects, we can classify the curves obtained into different typologies of typical toxicology functions [11]. Therefore, we can identify curves of Type I or "rank without effects series of inhibition", Type II or "stimulation followed by inhibition", Type III or "stimulation" and type IV or "no change". Bååth [30] publishes ranges of No Observed Effect Concentrations (NOEC) for Cd, Cu, Zn and Pb; in forest soils with a low organic matter content, the NOEC for Cu is distributed in a range from 2 $\mu g \cdot g^{-1}$ to 1600 $\mu g \cdot g^{-1}$; for the case of Zn, the NOEC were between 32 $\mu g \cdot g^{-1}$ and 1000 $\mu g \cdot g^{-1}$. Saviozzi *et al.* [33] and Renella *et al.* [34] established a rank order of metal toxicity depending on the metal's inhibition power on soil respiration; in this ranking, Cu is above Zn, therefore being the most toxic for the CO_2 production. Many authors who in these tests have applied organic amendments or used soils with high organic matter contents have not detected toxic effects even at high metal concentrations [31, 32]. However, there exist totally contradictory results, with significant inhibitions of heterotrophic respiration due to the presence of one of the studied metals, despite the addition of organic matter [43].

There are only two groups of factors that can contribute to this variability among the results reported in different previous studies: (1) factors which modify the toxicity of the metals and, (2) differences in the sensitivity of soil microorganisms or in microbial processes [7]. It is extremely difficult to sort out these factors when metal toxicity is studied in soils due to difficulties in assessing (or measuring) the bioavailability of the metals [35, 36, 37], and also due to the complexity of soil microbial communities [5]. In fact, bioavailability can only be estimated since it only can be measured by the growth of the organism of interest and by the absorption or the toxicity of the metal [38]. However, over the last ten years there has been considerable progress in defining bioavailability and in taking account of differences in bioavailability and sensitivity of soil organisms in EU risk assessment research [39]. In any case, the metals used in the present work were incorporated into the soil in a soluble form that is considered to have high bioavailability, independent of subsequent chemical reactions or changes in the soil.

For the present work, in the case of chromium, no evidence of inhibition was observed. The beginnings of a hormetic effect were detected at lower doses, and no inhibition was found at higher doses. This might be due to the fact that chromium was added in the trivalent form, which is less toxic than the hexavalent form [40]. Moreover, the trivalent cations bind strongly to clay particles [41], reducing the bioavailability of chromium. However, the experimental soil had a low proportion of clay and low organic matter contents, and consequently a reduced CEC. For this reason is expected that metal adsorption by these components would be minimal. As for the possibility of the preliminary test concentrations having been excessively low, several authors [42, 43] have found inhibitory effects for Cr at lower concentrations than the preliminary ones in this work. The stimulating effect observed for Cr might be attributable to the fact that this element is essential for many organisms, as it is in relatively high concentrations in RNA. Moreover, the cations of heavy metals can induce the liberation of nutrients adsorbed on the soil, causing indirect positive effects [11] in closed experimental systems.

Regarding the changes of toxicity with exposure time, in the case of the soil polluted with Zn, toxicity was lower after 28 days. This effect, reflected in a relative increase in respiration at the same dose rank, may be attributable to physiological changes or adaptations to metal pollution leading to a recovery or selective effect on the community [44].

The addition of fresh organic matter led to significant increases in SIR (Substrate-induced Respiration) (Figure 3.1). Other studies have shown that microbial biomass increases with the addition of fresh leaves, given the availability of easily-degradable substrate. This increase in microbial biomass is associated with an increase in soil respiration [50]. The addition of organic matter masked any effect of the rank of doses studied. This masking of the toxicity may therefore be due to an increase in the microbial biomass produced by the organic matter addition [37], to the adsorption and immobilization of metals on the added organic matter [51], or a combination of these factors. It has been reported that a broad variety of organic components increase the adsorption of Zn [52], and that the mineralization of the organic matter favours the precipitation of inorganic metallic compounds [53]. On the other hand, it is known that Cu associates easily with humified or stable organic matter [54, 55] to which it can be strongly adsorbed [56, 57]. It is for this reason that the addition of organic amendments is a common practice for the immobilization of heavy metals, improving the quality of the soil and facilitating revegetation of polluted soils [53, 58]. In this manner, the obtained results contribute to the validation of this practice.

When measuring toxicity at 1d in the OM+ treatments, taking into account a short period for interaction of the metal, the main cause of the observed lower toxic effect is probably the availability of labile substrate which favoured the growth of the bacterial biomass and increased its mineralisation activity, concealing any decrease in microbial populations. In this way, the inhibitory effect of the metal would be offset by the activating effect of the organic matter. This decrease of the toxic effect is observed in 1d as well as 28d measures, even though after 28 days the amount of emitted CO_2 diminishes significantly.

The decrease of SIR observed in OM+ at 28 days may also be owed to the r-K species transition, as we have appointed for 28d measures Cu OM- treatment. Stenström *et al.* [49] demonstrated that SIR measured in soils where glucose was added at different pre-incubation times showed a transition of K-strategists towards r-strategists. These species were responsible of 63% of SIR after 4 days of incubation, but later this trend reversed and the contribution of r-strategist microorganisms was reduced (at 46 days of incubation the r strategy group was responsible for only of 16% of the SIR). These results agree with the observations of Ritz *et al.* [59], who detected that after the addition of an easily decomposable substrate to soil the SIR increased significantly for only 25 days.

No inhibitory effects were observed in OM+ treatment SIR measurements (1d and 28d). The fact that relative differences (to the control) between the measures in OM+ treatments at 1d and 28d were not observed can be due to the fact that organic matter, as a source of mineralisable carbon (independent of its recalcitrance), had not run out completely. Only in the case of the soil polluted with Cu and amended with organic matter a slight decrease of the relative respiration was observed at 28 days. Therefore, the results obtained in the OM+ assays polluted with metals agree with the results of the OM- soils, since Cu proved to be the most toxic element. Moreover, some authors [60] have demonstrated that in presence of soluble organic matter, Cu is the more available of the studied metals.

In light of the results, the protocol OECD-217 is a valid methodology for the detection of adverse effects of the three metals studied on substrate-induced soil respiration. Cr, Zn, and Cu toxicities, evaluated by the OECD-217 soil respiration test, were low in the slightly acidic sandy test soil chosen for its potential high bioavailability of toxic metals. The EC_{50} values obtained in the present work are comparable to those presented by other authors who have worked with soils of similar characteristics [51, 42]. The magnitude of the inhibitory effects that have been observed for the three metals studied (Cu>Zn>Cr) is also coherent with the consulted references, Cu being the most toxic metal after 28 days of exposure [33, 30]. However, the microbial responses produced in the short-term assays were unpredictable and bore little resemblance to the long-term effects observed in the field in other studies [7].

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4. Sewage sludge as an organic amendment for quarry restoration: short-term effects on soil and vegetation

En aquest capítol s'estudien els efectes de l'ús de fangs de depuradora urbana com a esmena orgánica en la construcció de tecnosols de mina. En general els fangs afavoreixen un ràpid desenvolupament vegetal, inclús quan s'apliquen en materials estèrils com són els residus miners, la qual cosa redueix de manera molt significativa el risc d'erosió, afavorint així l'estabilització dels talussos i creant les condicions per l'establiment posterior de vegetació arbustiva i arbòria. Aquest capítol ha estat publicat a la revista Land Degradation and Development (Carabassa V, Ortiz O, Alcañiz JM. (2018) Sewage sludge as an organic amendment for quarry restoration: Effects on soil and vegetation. Land Degrad Dev. 2018;1–7. https://doi.org/10.1002/ldr.3071).

Abstract

Quarry restoration in Mediterranean environments usually needs organic amendments to improve the substrates used for technosol construction. Digested sewage sludge from municipal wastewater treatment plants are rich in organic matter, N and P and constitute an available an economically interesting alternative for substrate amendment. However, their pollutant burden and labile organic matter content involve an environmental risk that must be controlled. Moreover, ecological succession in restored areas can be influenced by the use of sludge and should be assessed. To minimize these risks, a new sewage sludge dose criteria relating to its labile organic matter and heavy metal content has been established. Sewage sludge doses currently range between 10 and 50 $Mg \cdot ha^{-1}$. In order to verify the suitability of this dose criterion, sixteen areas rehabilitated using sewage sludge located in limestone quarries in a Mediterranean climate in Catalonia (NE Iberian Peninsula) have been assessed. These evaluations focused on physicochemical properties of rehabilitated soils, land degradation processes and ecological succession. In the short term, six months after sludge application, an increment of organic matter content in the restored soils was observed, without significant increases in electrical conductivity or heavy metals content, and with a dense plant cover that contributes to effective soil erosion control. Two years after, ruderal plants were still present but later successional species colonized the restored zones in different degrees. These results suggest that sewage sludge, used as a soil amendment according to the proposed methodology, can safely improve technosol quality without constraints that compromise ecological succession.

Keywords: soil rehabilitation, organic amendment, stability degree, Technosol, erosion control, quarry restoration

Introduction

Restoration ecologists have long recognized the role of soil, particularly its physical and chemical properties, in the successful revegetation of degraded sites (Jordan *et al.*, 1987; Heneghan *et al.*, 2008). Starting from this premise, ecological restoration principles applied to quarry restoration implies in many cases the use of their own mine spoils (Tedesco *et al.*, 1999; Ram *et al.*, 2006; Jordán *et al.*, 2008), mainly when topsoil stripping is not possible or does not give a sufficient quantity of soil. However, these materials usually do not meet the minimum fertility requirements for their direct use as soil substitutes in land restoration and have to be improved using organic amendments. In this context, the use of sewage sludge as organic amendment could represent an economically and environmentally effective alternative to create a new fertile substrate, currently named technosol, for plant growth. Sewage sludge contains nutrients and trace elements essential for plant growth (Ortiz & Alcañiz, 2006), and organic matter, which can act as a soil conditioner to improve the physical properties, such as soil aeration and water-holding capacity (Sort & Alcañiz, 1999; Singh & Agrawal, 2008). However, their pollutant burden, comprising heavy metals and a variety of organic compounds, needs to be controlled. In this sense, there are some legislative regulations (European Union, 1986) and proposals (European Union, 2000) that establish maximum levels of heavy metals and organic pollutants in sewage sludge and receiving agricultural soils.

Organic amendments with high labile organic matter content are not suitable for land rehabilitation as this type of organic matter can be quickly mineralised, releasing large amounts of nutrients, which limits its positive effects to a short time. Moreover, in studies carried out with different types of organic wastes, a negative correlation between the degree of stability of organic matter (the proportion of stable, not labile, organic matter) and toxicity to plants and/or soil fauna has been described (Fuentes *et al.*, 2004; Domene *et al.*, 2007; Ramírez *et al.*, 2008). Furthermore, a strong correlation between the stability degree and total nitrogen, hydrolysable nitrogen and NH_4-N content has been found (Ramírez *et al.*, 2008). At the field level, high amounts of available nitrogen in rehabilitated soils promote ruderal plant species predominance (Moreno-Peñaranda *et al.*, 2004), which makes ecological succession difficult, and poses a risk for groundwater contamination (Navarro-Pedreno *et al.*, 2004). Regarding groundwater pollution by heavy metals, these do not pose a real risk because heavy metals mobility in water is very low (Hornburg & Brummer, 1994).

In order to limit or avoid these unintended situations, a new dose calculation protocol for organic amendments to be used as soil amendments has been proposed (Alcañiz *et al.*, 2009; Carabassa *et al.*, 2010). Relating to organic amendment characteristics, only stabilised amendments are allowed, with a recommendation for stable organic matter content greater than 30% (i.e. amendments containing a maximum amount of labile organic matter of 70%). In order to prevent heavy metal pollution, the protocol recommends avoiding sewage sludge and soils with metal concentrations above the limits proposed by the draft of the European Commission (2000), and do not reach these limits on the resulting technosoils. Moreover, in soils or substrates having more than 20 $g \cdot kg^{-1}$ of organic matter, sewage sludge amendment is not recommended.

On the other hand, alongside this dose calculation protocol, a site-aptitude evaluation methodology has been designed with the objective of avoiding applications on unsuitable areas such as sites vulnerable to pollution, protected groundwater recharge areas, etc. The items (topics) included in this evaluation procedure are the proximity of the zone to be restored to wells or watercourses, the location of a quarry in a zone with aquifers vulnerable to nitrate contamination, the water table depth, the accessibility and the space for storage and mixing of sludge with soil, site visitation frequency, farming utilization and the proximity to inhabited sites. Additionally, if the evaluation procedure determines that a site is suitable for sewage sludge use, a maximum area of 2 ha per year may be restored using this amendment. This protocol is currently being used by the Waste Agency of Catalonia (NE Iberian Peninsula) and by waste management companies to calculate sludge doses and sludge application conditions for their use in quarry restoration works (DTS, 2015).

The main goal of this paper is to introduce the dose criteria proposed and to evaluate the effects of sewage sludge application as a technosol amendment from an ecological restoration point of view. To do this, soil quality parameters, degradation processes and plant composition, as indicators of ecological succession, have been evaluated.

Methods

Sludge-dose criteria

According to the protocol described in Alcañiz *et al.* (2009) and Carabassa *et al.* (2010), sewage sludge is dosed depending on its organic matter stability, determined by weight loss-on-ignition (LOI) at 550°C after acid hydrolysis. The protocol proposes 5 g·kg^{-1} as a maximum amount of labile organic matter added by the sludge on the amended soil. Other parameters such as the organic matter content of the sludge, the thickness of the technosol layer to be applied (0.4 m maximum), the bulk density and the percentage of the <2 mm size fraction are considered in the dose-calculation formula:

$$D_{DM} = T \cdot BD_E \cdot FE \cdot \left(5 + \frac{5 \cdot S}{1 - S} \right) \cdot \frac{1}{OM_S} \cdot 10$$

Where D_{DM} is the dose of sludge (Mg·ha^{-1}, dry weight); *T* is the desired thickness of the substrate layer to be applied (m); BD_E is the bulk density of the mineral substrate (Mg·m^{-3}); *FE* is the proportion of <2 mm size particles of the mineral substrate (g·kg^{-1}); *S* is the degree of stability of the sludge (g·kg^{-1}) and OM_S is the proportion of organic matter in the sludge (g·kg^{-1}).

Moreover, as a preventative measure, this protocol fixes a maximum dose of sludge (50 Mg·ha^{-1}, dry weight), based on earlier experience obtained from ecotoxicological assays using plants and soil fauna (Domene *et al.*, 2007, Tarrasón *et al.*, 2008).

Restored zones selected

A set of 16 areas located in 11 quarries that were restored using their respective mine spoils and amended with sewage sludge according to the current protocol were selected (table 4.1). These quarries are located in Mediterranean environments encompassing a climatic gradient from semiarid to sub-humid. Mining activities exploiting diverse calcareous materials that could influence the restoration processes were included. Mine spoils were the main substrate used to create a technosol for topsoil rehabilitation. All the substrates were calcareous (30% to 70% of carbonate content) and stony, but with more than 30% of <2 mm particle-size fraction and a loamy texture.

Table 4.1. General description of the mine sites located in the NE Iberian Peninsula.

Quarry	Latitude (N)	Longitude (E)	Mean annual rainfall (mm)	Mean annual temperature (℃)	Previous vegetation	Parent material
Aiguamolls	41° 28´ 31´´	0° 49´ 39´´	450-500	14-15	Shrubland dominated by *Thymus vulgaris* and *Rosmarinus officinalis* as dominant shrubs	Marl
Calvari	41° 30´ 26´´	0° 57´58´´	350-400	14-15	Rainfed tall fruit trees	Marl
Ponderosa	41° 15´ 41´´	1° 09´ 27´´	550-600	15-16	*Pinus halepensis* forest	Limestone
Antonieta	42° 16´ 06´´	1° 22´ 11´´	750-800	11-12	*Pinus nigra* forest	Limestone
Lázaro	41° 12´ 07´´	1° 28´ 57´´	500-550	15-16	*Pinus halepensis* forest	Limestone
Orpí	41° 31´ 22´´	1° 36´ 15´´	600-650	13-14	*Pinus halepensis* forest	Limestone
Montlleó	41° 41´ 35´´	1° 49´ 39´´	600-650	13-14	*Thymus vulgaris* and *Rosmarinus officinalis* shrubs	Sandstone
Vallcarca	41° 15´ 13´´	1° 52´ 25´´	500-550	15-16	Mixed forest: *Quercus ilex* and *Pinus halepensis*	Marl and limestone
Falconera	41° 15´ 40´´	1° 53´ 12´´	500-550	15-16	*Pinus halepensis* forest	Limestone
Cubetas	41° 20´ 24´´	1° 53´ 33´´	600-650	13-14	*Pinus halepensis* forest	Dolostone
Cuevas	41° 16´ 16´´	1° 54´ 13´´	500-550	15-16	*Quercus coccifera* shrubs	Limestone

The average sewage sludge dose was 40 ± 13 Mg ha^{-1} (mean ± SD, dry weight basis). All sludge applied came from municipal wastewater treatment plants close to the mining areas, having been subjected to an anaerobic digestion process and partially dehydrated (table 4.2). The organic matter content of these sludges and their stability degree was relatively high. They had high concentrations of nitrogen and phosphorous, similar to those currently applied to agricultural soils. The heavy metals content was low and always met the requirements of the European Union (2000).

Steep slopes (60-75%) are the predominant restored surfaces in the mine sites selected, with some exceeding 100%. On these slopes, a layer of about 20 cm of amended substrates (mainly mine spoils mixed with sewage sludge) was spread on top. The average area of restored slopes per site is 4,500 m^2.

Table 4.2. Analytical characterization of sewage sludges applied on the selected mine sites.

Parameter	Average	Max.	Min.	SD
Dry matter (%)	24.5	26.8	22.5	12.6
Organic matter (%)	57.7	70.7	39.2	27.1
Stability degree (%)	48.1	60.1	31.8	17.2
Conductivity (dS m^{-1} 25°C)	2.0	3.0	1.0	0.7
pH (water 1:10 w:v)	7.7	8.5	6.9	0.6
N-Kjeldhal (g kg^{-1})	36.8	63.4	11.5	19.8
N-ammonia (g kg^{-1})	10.7	21.5	1.8	7.3
P- total (g kg^{-1})	38.3	64.4	24.5	23.5
K (g kg^{-1})	2.8	5.7	1.0	1.5
Cu (mg kg^{-1})	322.3	580.0	102.0	170.6
Ni (mg kg^{-1})	23.0	43.3	15.5	13.9
Cr (mg kg^{-1})	56.6	85.6	12.5	35.7
Pb (mg kg^{-1})	54.3	64.2	29.5	28.1
Hg (mg kg^{-1})	2.0	4.2	0.2	1.3
Cd (mg kg^{-1})	3.7	10.0	0.9	3.6
Zn (mg kg^{-1})	1037.6	2199.0	375.0	512.6

Evaluation parameters

The parameters evaluated in the restored zones are related to soil quality, degradation processes and vegetation. Soil samples were taken 4-6 months after sludge application but degradation processes and vegetation data were assessed after 24 months. Soil sampling involved taking a composite sample of cores (n =12-20, d=0-20 cm) for each restored zone. The analysed parameters in soil samples were: particle size determined by sedimentation-Robinson pipette (Gee & Or, 2002), equivalent $CaCO_3$ by CO_2 volume released after HCl addition -Bernard calcimeter method (Nelson, 1982), electrical conductivity of 1:5 water extract (Rhoades, 1982), organic carbon content by acid dichromate oxidation (Nelson & Sommers, 1982), total nitrogen using the Kjeldahl method (Bremner & Mulvaney, 1982), available phosphorous-Olsen phosphorous (Olsen & Sommers, 1982) and potassium (Knudsen *et al.*, 1982), and heavy metals by ICP-MS analysis (Thomas, 2004). Geotechnical and soil degradation processes were estimated through direct measures and observations in the field, and erosion losses were estimated by measuring the rill volume. Vegetation measures were taken by establishing transects and sampling plots (10m transects, $100m^2$ plots, 3 per area).

Results and discussion

Soil quality indicators of technosoils were evaluated after one growing season (spring or autumn) (table 4.3). The electrical conductivity of amended soils remained low, with the only exception of two cases that were greater than 2 dS m^{-1}. Organic C contents were almost always higher than 10 g kg^{-1}, with an average increment of 11.2 ± 7.8 g kg^{-1} (mean ± SD) caused by the addition of the sludge to the mineral substrate that constitutes the mineral fraction of the technosol. Phosphorous content tended to be high and correlated with the amount of sewage sludge added. Total nitrogen concentrations were balanced to the organic matter content, having a C/N ratio close to 10, and relating to the sludge dose applied. Available potassium content tended to be low, especially in very rich calcium soils. Heavy metals content was low in the amended soils. Minor increases in the concentrations of these elements were observed after the addition of sewage sludge, all below the European upper limits for agricultural soils (table 4.4).

Table 4.3. Summary of physical and chemical parameters of amended substrates (Technosols). N=28. Data related to the fine fraction (<2 mm), except coarse particle size fractions that are reported as percent of whole soil sample (ws).

Parameter	**Average**	**Max.**	**Min.**	**SD**
Fraction >5 cm (% ws)	3	10	0	3.5
Fraction 5-1 cm (% ws)	23	40	8	9.1
Fraction 1-0.2 cm (% ws)	21	42	8	8.1
Fine earth <0.2 cm (% ws)	53	87	31	15.7
Sand (%)	35	60	16	12.9
Silt (%)	41	63	25	11.3
Clay (%)	23	28	14	4.4
$CaCO_3$ eq. (%)	52	80	29	14.7
pH water (1:2.5 w:v)	8.1	8.5	7.9	0.2
Electrical conductivity (1:5 extract w:v, dS·m^{-1} 25°C)	1.02	2.94	0.3	0.8
Organic carbon (g·kg^{-1})	11.3	23.4	2.1	4.1
N-Kjeldhal (g·kg^{-1})	1.1	1.9	4.0	3.0
P-Olsen (mg·kg^{-1})	66	103	13	25.4
K-available (mg·kg^{-1})	116	244	76	41.0

Table 4.4. Heavy metals concentration on amended soils and average and maximum increments relating to original mine spoils (nd: no detected; units: mg·kg^{-1}). Z1, Z2 and Z3 refers to diverse slopes or zones in the same quarry. n=28.

Zone code	Cd	Cu	Cr	Hg	Ni	Pb	Zn
Average	0.100	20.8	25.1	0.062	19.1	27.2	77.5
Max.	0.500	50.0	40.0	0.097	29.0	55.0	190.0
Min.	0.100	6.0	7.0	0.041	11.0	11.0	18.0
SD	0.005	10.7	9.4	0.024	5.7	14.9	39.9
Limits EU*	1.5	100	100	1	70	100	200
Average increase	nd	0.3	1.8	nd	2.1	2.1	4.4
Maximum increase	nd	2.1	11.5	nd	12.0	11.5	14.5

* Limits proposed for agricultural alkaline soils (EU, 2000).

Soil quality parameters indicate that sewage sludge application causes a substantial improvement of organic matter and nutrient content in the amended technosoils, as reported in other works (Albiach *et al.*, 2001; Heras *et al.*, 2005). Despite the noticeable increase in the soil nutrients due to sludge mineralisation, electrical conductivity did not rise greatly. Available phosphorous levels were increased by sludge addition but were in a high-medium range compared with agricultural soils of the region (Peñuelas *et al.*, 2009). However, partial immobilization could take place due to the alkaline pH of these highly calcareous soils. According to sludge composition, mineral nitrogen levels may rise just after sludge application, particularly in ammonium form (not measured in our soil samples). This ammonium nitrogen may rapidly turn into nitrates in these calcareous soils (Kleber et al., 2000), which could then leach from the soil and contaminate aquifers. However, nitrogen leaching, if it occurs, should take place mainly in the first four months after sludge application due to the high mineralization rates of organic-N from sludge. After this, leaching risk should decrease quickly due to nitrate absorption by roots of the growing plants, reducing the risk of aquifer eutrophication (Tarrassón *et al.*, 2008; Jordán *et al.*, 2017). Moreover, it has to be taken into account that this risk is relatively low due to the small surface restored (<2 ha in a quarry per year) and because sludge is applied only once, compared to agricultural applications where sludge is applied recurrently at the same plot. On the other hand, a fraction of organic matter from sewage sludge persists in soil due to its recalcitrant composition (e.g. some lipids and aromatic hydrocarbons), and a labile organic matter fraction could also remain in soil protected as aggregates or by sorption on mineral surfaces (Ojeda *et al.*, 2015).

The total amount of heavy metals in the amended technosoils never exceeded the concentrations fixed by the European Union proposal (2000), which is stricter than the current European Directive that regulates the agricultural use of sewage sludge. This is due to the relatively low concentration of these elements in the sewage sludge but also to the moderate doses applied. Moreover, the metal bioavailability in the studied technosoils is expected to be low, according its pH and lime content (Ortiz & Alcañiz, 2006).

No noticeable water erosion or other soil degradation processes were observed 24 months after sludge application. Only in two cases, stability issues (landslides and fallen rocks) were reported due to the excessive slope (greater than 100% in some sites). Rill erosion was detected in seven restored zones although the estimated erosion rates were always below 6 $Mg \cdot ha^{-1} \cdot y^{-1}$, which can be considered an acceptable rate in slopes and surfaces of recently restored areas (Verheijen *et al.*, 2009), with the one exception of Las Cuevas quarry, where the Z1 zone was severely affected (table 4.5).

Table 4.5. Surface of soil affected by water erosion, erosion rates and soil loss on the evaluated zones of quarries where soil erosion has been detected (Z1, Z2 and Z3 refer to diverse slopes or zones in the same quarry).

Zone code	**Affected area (m^2)**	**Erosion rate ($Mg \cdot ha^{-1} \cdot year^{-1}$)**	**Soil loss ($Mg \cdot ha^{-1}$)**
Aiguamolls Z1	1225	2.3	5.3
Aiguamolls Z2	1870	0.3	0.7
Aiguamolls Z3	1950	2.2	5.0
Lázaro	1800	0.4	0.9
Cubetas	8040	1.9	4.5
Cuevas Z1	2000	17.3	55.4
Cuevas Z2	4400	3.2	10.0

Dense plant cover was observed in the evaluated zones two years after sludge application. The average plant cover was 70 ± 24 % and herbaceous plant height 0.45 ± 0.30 m. Concerning species richness, more than 20 plant species were identified in the evaluated zones. Regarding herbaceous vegetation (see figure 4.1), grasses were the most frequent functional group of plants ($P < 0.015$). However, some reported herbaceous species can be considered as ruderal plants that grow in nutrient-rich and disturbed habitats, usually resulting from human activity. Legumes are also well represented in sewage sludge revegetated zones at a similar frequency to Asteraceae and ruderals. No exotic or invasive species were observed in the evaluated zones, despite the presence of some individuals of *Arundo donax* in one zone (Falconera quarry) before sludge application.

The observation of vegetation succession showed that native species started to colonize the amended zones two years after sludge amendment. Herbaceous species were the main colonizers, being found in half of the amended zones. Shrub species appeared in four restored zones, especially *Santolina chamaecyparissus* and *Rosmarinus officinalis.* Moreover, in approximately 60% of the rehabilitated zones where shrubs were planted, spontaneous reproduction of these species was observed.

Vegetation cover is a key parameter for soil stabilization and erosion control (Merlin *et al.*, 1999; Bochet *et al.*, 2010; Espigares *et al.*, 2011), mainly in major civil works such as the construction of motorways, the rehabilitation of quarries or dumps, and even the creation of ski slopes. Several authors (Albiach *et al.*, 2001; Pond *et al.*, 2005) have demonstrated that soils amended with sewage sludge favor a fast vegetal recovery and plant growth, especially for herbaceous vegetation, which is the best way to control soil erosion in steep slopes. Moreover, sewage sludge promotes soil aggregation (Sort & Alcañiz, 1996; Sort & Alcañiz, 1999; Ojeda *et al.*, 2008), reducing soil erodibility. These combined beneficial effects of sewage sludge on vegetation development and soil structure are probably the main reasons explaining the reduced erosion rates found on steep slopes that are especially vulnerable to soil erosion, in the studied quarries.

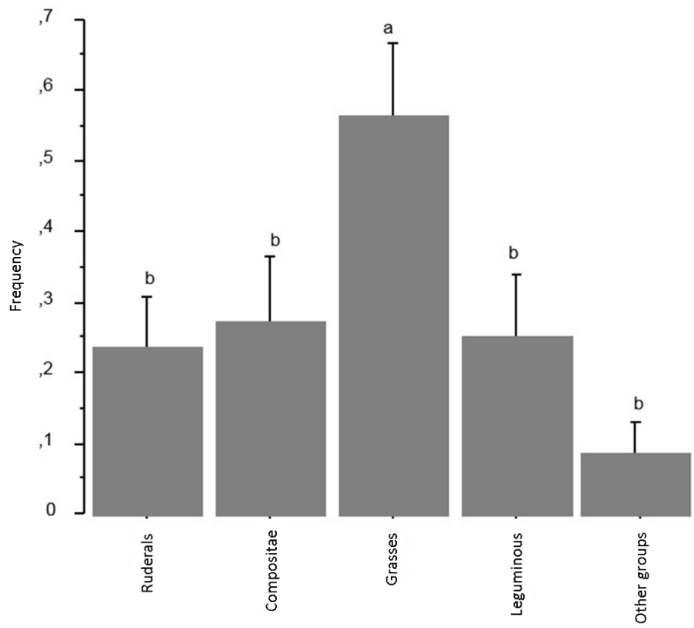

Figure 4.1. Frequency of different herbaceous plant functional groups in the evaluated zones of quarries. The error bars indicate standard error ($p < 0.02$).

The relatively lower frequency of ruderal species found in this work (see figure 3.1) compared with a previous study of the same team (Moreno-Peñaranda *et al.*, 2004) that showed a dominance of ruderal plants on sewage sludge amended zones must be discussed. This apparent discrepancy may be explained by the different doses of sludge applied, which were lower in the present work (calculated following the protocol reported in Carabassa *et al.*, 2010) compared to other previous studies (Sopper, 1993; Alcañiz *et al.*, 1996; Barnhisel *et al.*, 2000; Jorba & Andrés, 2000; Morera *et al.*, 2002; Moreno-Peñaranda *et al.*, 2004). Therefore, the main difference is the quantity and quality of the organic matter added to the mineral substrate. As explained in the methods section, the new procedure calculates the dose of sludge according to its concentration of stable organic matter (stability degree), and fixes the maximum dose at 50 $Mg{\cdot}ha^{-1}$. These criteria imply an addition of a limited proportion of labile organic matter and a relatively low addition of total organic matter associated with sludge application, which contribute to reduce the development of ruderal plants currently associated with over-fertilized soils. However, the presence of ruderal plants in restored areas may be common also when organic amendments are not used (Hobbs & Adkins, 1988; Alpert *et al.*, 2000; Moreno-Peñaranda *et al.*, 2004). Therefore, the use of sewage sludge in appropriate doses should not be considered as a barrier regarding plant community succession towards the natural surrounding vegetation. Furthermore, the noticeable recruitment of native shrub species may suggest a plant community convergence with adjacent undisturbed habitats in the medium term. However, this process should be monitored in the long term, as it is one of the main objectives of ecological restoration (SER, 2004). Moreover, an increasing emphasis will be focused on the proper ecological functionality of a restored site, and to a lesser extent on returning a restored site back to previous conditions based on species composition (Harris *et al.*, 2006).

Conclusions

For the range of climatic and soil conditions tested in this work, the use of sewage sludge for vegetation recovery purposes in restoration works is a good alternative that allows the valorization of sewage sludge and increases the quality and stability of restored areas, reducing the risk of soil erosion. One of the most important parameters to take into account for sewage sludge dosage is the amount of labile organic matter, in order to avoid compromising encroachment and reduce the risk of nitrate contamination. Moreover, an aptitude evaluation of sludge, mineral substrates and location is mandatory before sludge application in order to prevent contamination and detrimental impacts on inhabited zones.

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5. Mid-term effects on ecosystem services of quarry restoration with Technosols under Mediterranean conditions: 10-year impacts on soil organic carbon and vegetation development

Aquest capítol té com a principal objectiu avaluar els efectes a mitjà termini de l'ús de fangs de depuradora en la construcció de tecnosols de mina. S'observa que els sòls on s'ha aplicat fangs segresten més carboni i afavoreixen un major desenvolupament vegetal, sense comprometre el reclutament. Aquest capítol s'ha publicat a la revista Restoration Ecology (Carabassa, V., Domene, X., Díaz, E., Alcañiz, J.M. (2019). Mid-term effects on ecosystem services of quarry restoration with Technosols under Mediterranean conditions: 10-year impacts on soil organic carbon and vegetation development. Restoration Ecology; https://doi.org/10.1111/rec.13072).

Abstract

The use of Technosols for the restoration of limestone quarries overcomes the usual 'in situ' scarcity of soil and/or its poor quality. The use of mine spoils, improved with mineral and/or organic amendments, could be an efficient and environmental friendly option. Properly treated sewage sludge from urban wastewater treatment plants could be a suitable organic amendment and fertilizer (rich in N and P) whenever its pollutant burden is low (heavy metals and/or organic pollutants). Its appropriate use could improve essential soil physical and chemical properties and, therefore, promote key ecosystem services of restored areas, such as biomass production and carbon sequestration, as well as biodiversity and landscape recovery. However, the mid-term impacts of these restoration practices on soil functioning and their services have rarely been reported in the available literature. In this study we assess the mid-term effects (10 years) of the use of sewage sludge as a Technosol amendment on soil organic carbon (SOC), nutrient status, and plant development in several restored quarries. Soils restored using sewage sludge showed a three-fold increase in SOC compared to the corresponding unamended ones, despite the moderate sludge dosage applied (below 50 $Mg \cdot ha^{-1}$). Plant cover was also higher in amended soils, and recruitment was not affected by sludge amendment at these doses. This study demonstrates that, used at an appropriate rate, sewage sludge is a good alternative for the valorization of mine spoils in quarry restoration, improving some important regulatory ecosystem services such as carbon sequestration, without compromising woody plant encroachment.

Keywords: Quarry restoration; Technosol; Organic amendments; Sewage sludge; Soil organic carbon sequestration

Implication for practice

- The use of sewage sludge as organic amendment in mine spoil-based Technosols is an environmentally friendly and economically-suitable option for the restoration of land degraded by quarrying and could allow a fast recovery of the ecosystem services lost.
- After 10 years, sludge-amended Technosols boosted primary production and promoted a three-fold increase in organic carbon stocks without compromising woody plant encroachment.
- The use of digested municipal sewage sludge with high stability degree (>30%), at moderate doses (ca. 45 Mg ha^{-1}), in Technosols construction minimizes environmental risks and maximize ecosystem services in terms of carbon sequestration and plant biodiversity.

Introduction

Quarrying activities produce severe impacts on important ecological functions that provide ecosystem services contributing to human wellbeing. In most of the scenarios, restoring ecosystem services after finishing the exploitation implies the recovery of vegetation in sites where soil fertility levels have been depleted (Moreno-Peñaranda *et al.* 2004). Manufactured soils (Technosols) could be a viable soil source when the availability of suitable natural soils is limited (Watkinson *et al.* 2017), with this technology being emblematic of the issues we face for the management of the soils of the Anthropocene (Leguédois *et al.* 2016). The use of organic waste for Technosol construction is a widely used practice in mine restoration (Asensio *et al.* 2013; Watkinson *et al.* 2017; Lomaglio *et al.* 2017), with the aim of speeding up the biological colonization of a relatively inert initial substrate. Specifically, the use of sewage sludge for quarry restoration is a management option that contributes to the valorization of mine spoils and sludge from urban wastewater treatment plants (Sopper, 1993), in agreement with the EU principles for a more circular economy (Mosquera-Losada *et al.* 2017). When a Technosol approach is taken, sewage sludge is applied once, at a moderate dosage, to act both as fertilizer and organic amendment, and usually mixed with mining debris before their spreading as a topsoil layer in the restored areas. Sewage sludge is then an interesting option for valorizing mine spoils due to its fertilizing properties (Van-Camp *et al.* 2004) and its known positive effects on soil aggregate stability (Caravaca *et al.* 2002; Ojeda *et al.* 2008), soil water retention (Ojeda *et al.* 2010, 2011), and vegetation recovery (Ortiz & Alcañiz 2006; Moreno-Peñaranda *et al.* 2004; Carabassa *et al.* 2018). However, sewage sludge application also requires strong supervision and monitoring due to certain environmental risks related to its potentially harmful heavy metals and persistent organic pollutant content (Düring & Gäth 2002; Carabassa *et al.* 2010).

Pedogenic processes occurring in Technosols are similar to those of natural soils (Leguédois *et al.* 2016), although the components used can strongly influence their evolution and their capacity to behave as a soil, and therefore, to provide the associated ecosystem services. However, they tend to have a fast evolution compared to natural soils, including biological activity (Leguédois *et al.* 2016). The last is of importance since soil fauna and vegetation are key factors for the provision of ecosystem services in soils (Tate 2005), but biological activity also drives soil pedogenesis itself, e.g. through its role in carbon and nitrogen dynamics (Frouz *et al.* 2013).

Regarding carbon sequestration in soil, it is generally achieved by any biomass input that originated through a process causing a net removal of atmospheric CO_2-C by plants, and then stored as stable soil organic matter. The storage efficiency of the different pools of organic matter is highly influenced by its biochemical recalcitrance, its stabilization as organomineral aggregates, the occlusion in soil aggregates, or its transportation into the subsoil (Lal 2003). Sewage sludge, as a biomass derived residue, may directly contribute to soil carbon sequestration through its stabilization in soil, and indirectly through the increase in plant biomass production and litter intake (Ojeda *et al.* 2015). However, the relatively low stability of sewage sludge (Mattana *et al.* 2014) is expected to cause transient effects on soil organic matter pools, as mid-term soil organic carbon (SOC) sequestration relies more on the subsequent evolution of organic carbon (OC) inputs from plant debris intake to soil rather than on the OC provided by the sludge (Ojeda *et al.* 2015). Organic matter improvements, in turn, can contribute to other relevant ecosystem services in rehabilitated land such as raw materials production (for fuel, construction materials, etc.), nutrient cycling, climate regulation or improving soil as habitat for organisms (Baveye *et al.* 2016). However, little is known about the efficiency of Technosol approaches in terms of soil carbon storage (Ojeda *et al.* 2015), and especially in the mid and long-term, which could be of interest for offsetting the emissions of mining activities closely linked to the cement industry, one of the main contributors to industrial CO_2 emissions (Imbabi *et al.* 2012).

The use of sewage sludge in Technosols is expected to enhance the biological colonization in the initial stages of the restoration. In previous studies, the use of sewage sludge was shown to strongly influence plant community structure in the short-term in a shrubland intended to be converted into a dehesa (Tarrasón *et al.* 2014; Ferreiro-Domínguez *et al.* 2011). On the contrary, no significant effects on diversity were found when the reference was the neighboring forest or shrubland (Moreno-Peñaranda *et al.* 2004). Some negative ecological effects of the use of sewage sludge have been reported elsewhere, such as declines in microbiota, mesofauna and macrofauna sensitive taxa (Giller *et al.* 2009; Andrés *et al.* 2011; Barrera *et al.* 2001), the promotion of exotic species (Alpert *et al.* 2000), and decreases in plant biodiversity (Ferreiro-Domínguez *et al.* 2011). This is of concern given the ecosystem services driven by plants (Lavorel 2013) and soil organisms (Lavelle *et al.* 2006).

To prevent the negative effects of excessive dosing of sewage sludge that might compromise the provision of ecosystem services of rehabilitated land, some recommendation protocols have been proposed, such as the one used since 2008 in Catalonia by the local environmental authorities (Alcañiz *et al.* 2009; Carabassa *et al.* 2010; Department of Territory and Sustainability, 2015). This guideline takes into account the mineral substrate characteristics, the stability of the sewage sludge used (recommended to be over 30%), its pollutant burden, the site aptitude (sewage sludge should not be used near to wells, water courses, nitrate vulnerable zones or highly frequented or inhabited areas), among other environmental restrictions. The protocol also states a dosing limit for sludge (50 Mg ha^{-1}) and a maximum increase in labile organic matter in the receiving soils or substrates (0.5%).

The main goal of this article is to assess the mid-term effects (10 years), on two key ecosystem services (carbon sequestration and habitat function), of the use of sewage sludge in Technosol construction for limestone quarry restoration, under Mediterranean conditions.

Methods

Study sites

A set of seven limestone quarry sites, restored ten years ago using Technosols, were selected, all of them located in the Mediterranean climatic area of Catalonia (NE Iberian Peninsula) (Figure 5.1). Each experimental site corresponded to a Technosol constructed using sewage sludge, and a neighboring control area corresponding to a Technosol with the same mineral substrate but without adding sludge (Table 5.1). Climatic conditions in the different sites mostly differed in terms of water availability, since mean annual precipitation ranged from 400 to 700 mm (from wet to semiarid Mediterranean climate). The reference ecosystem for the restoration was a Mediterranean forest that predominates in the study area, dominated by Aleppo pine (*Pinus halepensis*) usually mixed with holm-oak (*Quercus ilex*) and accompanying shrub species (Table 5.1). Prior to restoration, the areas were used for limestone exploitation for aggregate or concrete production. The evaluated sites (sludge amended and controls) had an average surface area of 3000 m^2, and covered many facings, from the most favorable (N face) to the most challenging (S face). The dominant geomorphologic (landform) type is the terrace/berm embankment with steep slopes, some approaching 45°. The subsoil of embankments primarily consisted of fine and/or rocky fractions from extraction debris or excavations, and sometimes with blasting debris.

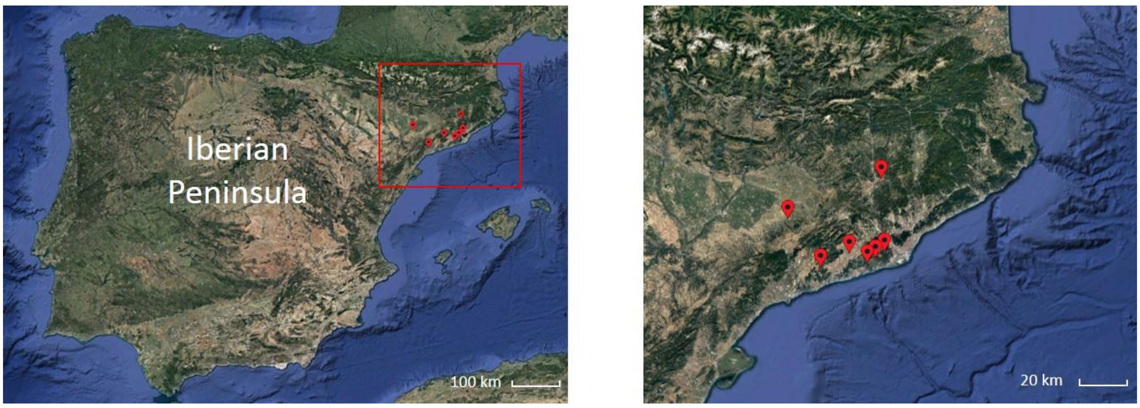

Figure 5.1. Location of the evaluated quarries on the NE Iberian Peninsula.

Table 5.1. Quarry sites, their location, climatic conditions (precipitation and temperature), slope characteristics (orientation, morphology, steepness, filler material and lithology) and reference system (plant community of the surrounding area). *material used to fill the excavation hole for giving the final morphology (geomorphic reclamation) to the restored areas (De la Vergne, 2006).

Site	Latitude (N)	Longitude (E)	Mean annual precipitation (mm)	Mean annual temperature (°C)	Orientation	Landform	Maximum slope (°)	Filler*	Dominant lithology	Reference system
Aiguamolls	41° 28´ 31´´	0° 49´ 39´´	451	14.9	W	Terrace/berm embankment with steep slope	42	Rocky debris	Marl	*Thymus vulgaris* and *Rosmarinus officinalis* shrubland with *P. halepensis*
Ponderosa	41° 15´ 41´´	1° 09´ 27´´	589	14.9	E	Terrace/berm embankment with steep slope	42	Soil and rocky debris	Limestone	*Pinus halepensis* forest
Lázaro	41° 12´ 07´´	1° 28´ 57´´	505	15.9	SE	Terrace/berm embankment with steep slope	43	Rocky debris	Limestone	*Pinus halepensis* forest
Fou	41° 21´ 32´´	1° 54´ 34´´	684	13.2	NE	Terrace/berm embankment with steep slope	33	Rocky debris	Limestone	*Pinus halepensis* forest
Vallcarca	41° 15´ 13´´	1° 52´ 25´´	535	15.6	NO	Terrace/berm embankment with steep slope	42	Without filler, blasting debris	Marl and limestone	Mixed forest: *Quercus ilex* and *Pinus halepensis*
Falconera	41° 15´ 40´´	1° 53´ 12´´	545	15.5	NO	Terrace/berm embankment with steep slope	33	Soil and mining wastes	Limestone	*Pinus halepensis* forest
Montlleó	41° 41´ 35´´	1° 49´ 39´´	630	13.7	S	Terrace/berm embankment with steep slope	40	Rocky debris	Marl	*Thymus vulgaris* and *Rosmarinus officinalis* shrubland with *Pinus halepensis*

Technosols construction

The mineral substrate used for Technosol construction mostly consisted of rocky debris, sometimes mixed with topsoil kept aside from mine topsoiling or excavations. In some cases, the stoniness was very high (over 80%), having a high proportion of carbonates and very low organic matter content (Table 5.2).

Sewage sludge consisted of an anaerobically digested sludge coming from medium size municipal wastewater treatment plants. All of them had enough quality to be used in agriculture, i.e. had relatively high stability (48% as average) and low heavy metal content (Table 5.3). As usually found in sewage sludge, P concentrations were very high. Sludge was dosed according to its organic matter content, stability degree, and the properties of the mineral substrate (stoniness and bulk density), following Alcañiz *et al.* (2009) and Carabassa *et al.* (2010). The average sludge dose used in the different sites was around 45 $Mg \cdot ha^{-1}$ (d.w.), and its field application was conducted between autumn 2006 and spring 2007.

Field sampling and laboratory analysis

The parameters assessed in the rehabilitated areas reflect the sludge-based Technosols ability to improve soil quality, minimize degradation processes and promote vegetation development. Soil samplings were carried out in 2007 (4-6 months after sludge application), and in May-June 2017 (10-11 years after sludge application). Vegetation measures were carried out in May-June 2017 after 10-11 years of the sludge application. Soil sampling involved taking a composite sample of cores (n =12-20, d=0-20 cm) for each restored zone with an Edelman auger.

The soil parameters analyzed were particle size determined by sedimentation-Robinson pipette (Clarke Topp & Ferré 2002), equivalent $CaCO_3$ by CO_2 volume released after HCl addition -Bernard calcimeter method, electrical conductivity of 1:5 (w:v) water extract, SOC content by acid dichromate oxidation, total nitrogen using the Kjeldahl method, available phosphorous-Olsen and available potassium (American Society of Agronomy 1982). In each restored area, vegetation measures were taken according to Carabassa *et al.* 2019: establishing 6 x 10 m transects and measuring the main cover types (herbaceous, shrubs, trees, organic debris, bare soil) by contact-point each 20 cm; shrub and trees density identifying and counting all the seedlings in 3 x 100 m^2 plots; flora inventories identifying all the species present; species abundance by qualitative observation of its respective cover.

Data analysis

Analysis of Variance (1 way-ANOVA and repeated measures ANOVA) was used to examine differences between treatments (amended and control soils) regarding soil properties (organic matter, N and P contents), proportion of soil cover in each category (herbaceous, total vegetal, organic debris) and herbaceous development (height), using p=0.05 as the cut-off for statistical significance throughout the manuscript.

Table 5.2. Properties of the soils and mining debris used for the Technosols' construction.

Site	< 2 mm (%)	Sand* (%)	Silt* (%)	Clay* (%)	Carbonate* ($g \cdot kg^{-1}$)	pH	Organic matter* ($g \cdot kg^{-1}$)	Electrical conductivity (1:5 extract, $dS \cdot m^{-1}$)	Bulk density ($Mg \cdot m^{-3}$)
Aiguamolls	22	14	50	36	421	8,8	1.0	0.48	1.8
Ponderosa	45	34	33	34	314	8.3	7.0	0.90	1.5
Lázaro	17	50	22	28	734	8.3	2.7	0.27	1.6
Fou	22	21	30	49	426	8.2	7.0	0.58	1.4
Vallcarca	57	46	22	32	271	8.3	3.7	0.37	1.5
Falconera	55	58	21	21	207	8.1	11.9	1.16	1.4
Montlleó	75	41	40	19	390	8.4	8.0	1.96	1.2

*referring to the fine fraction (< 2 mm)

Table 5.3. Physico-chemical characterization of the sewage sludge used for Technosol construction.

Parameter	Average	Max.	Min.	SD
Dry matter (%)	24.5	26.8	22.5	12.6
Organic matter (%)	57.7	70.7	39.2	27.1
Degree of stability (%)*	48.1	60.1	31.8	17.2
Conductivity (1:5 extract. $dS \cdot m^{-1}$ 25 °C)	2.0	3.0	1.0	0.7
pH (water 1:10 w:v)	7.7	8.5	6.9	0.6
N-Kjeldhal ($g \cdot kg^{-1}$)	36.8	63.4	11.5	19.8
N-ammonia ($g \cdot kg^{-1}$)	10.7	21.5	1.8	7.3
P- total ($g \cdot kg^{-1}$)	38.3	64.4	24.5	23.5
K ($g \cdot kg^{-1}$)	2.8	5.7	1.0	1.5
Cu ($mg \cdot kg^{-1}$)	322.3	580.0	102.0	170.6
Ni ($mg \cdot kg^{-1}$)	23.0	43.3	15.5	13.9
Cr ($mg \cdot kg^{-1}$)	56.6	85.6	12.5	35.7
Pb ($mg \cdot kg^{-1}$)	54.3	64.2	29.5	28.1
Hg ($mg \cdot kg^{-1}$)	2.0	4.2	0.2	1.3
Cd ($mg \cdot kg^{-1}$)	3.7	10.0	0.9	3.6
Zn ($mg \cdot kg^{-1}$)	1037.6	2199.0	375.0	512.6

*percentage of organic matter resistant to acid hydrolysis.

Results

Soil properties

After ten years of the Technosols construction, sludge-amended soils had significantly higher organic matter, N, and P contents compared to the unamended ones (Figure 5.2), achieving two order of magnitude increase in one of the sites. Some unamended Technosols were clearly deficient in organic matter, with SOC values below 0.5%. These differences were even stronger in the case of N, where sludge-amended Technosols had more than three times higher contents compared to unamended ones, which were extremely poor in N, especially when only mine spoils were used as mineral substrate. Moreover, P contents showed the largest contrast between amended and control soils, with very high contents in the sludge-amended Technosols, which were ten times higher than in unamended soils where values showed a clear deficiency.

Regarding soil C:N ratio (Figure 5.3), most soils were well balanced, with a ratio between 8 and 12, which are typical values for A horizons of calcareous Mediterranean forest soils. In general, ratios were similar for amended and unamended soils, except in the Lázaro control treatment, where C:N ratio was extremely high due its very low N content. Regarding N:P ratio, it has been calculated using Kjeldhal N and available P. Controls were deficient in available P and highly unbalanced, mainly when rocky debris without including topsoil were used for Technosol construction.

Comparing the soil properties values shortly after the Technosols construction (4-6 months) with the results after 10 years (Figure 5.4), it was shown that SOC increased in both amended and unamended soils (control areas) but more strongly in the sludge-amended plots, with a 2.1-fold increase compared to the 1.6-fold in the unamended Technosols. After 10 years, the control Technosols also failed to achieve the initial SOC levels of sludge-amended one. Amended soils tended to show significant increases in N content after 10 years, but this trend was not observed for P, with non-significant changes, despite a tendency to increase.

Considering only the topsoil layer (first 20 cm depth), the average organic C sequestration in sludge-amended Technosols was 28 Mg C·ha^{-1} after 10 years, and 9 Mg C·ha^{-1} was contained in the unamended ones, which represents a three-fold increase in the sequestered C.

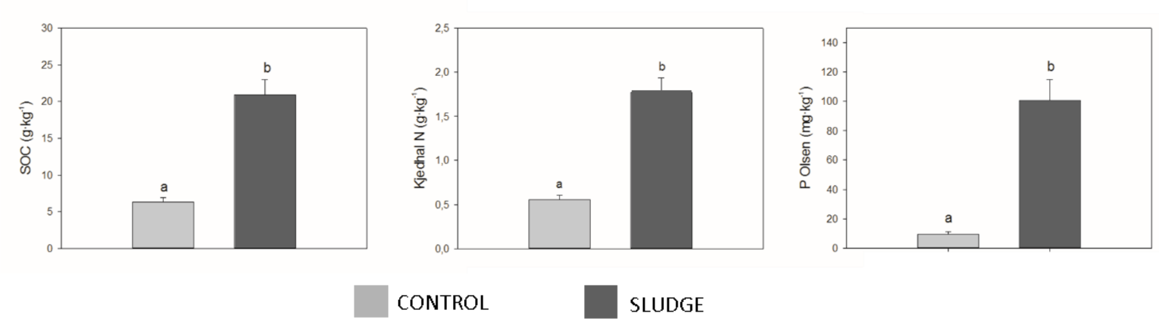

Figure 5.2. SOC, Kjeldhal N and Olsen P content in sludge-amended and control Technosols 10 years after their construction. The error bars correspond to standard error and different letters indicate significant differences at a p<0.05 level.

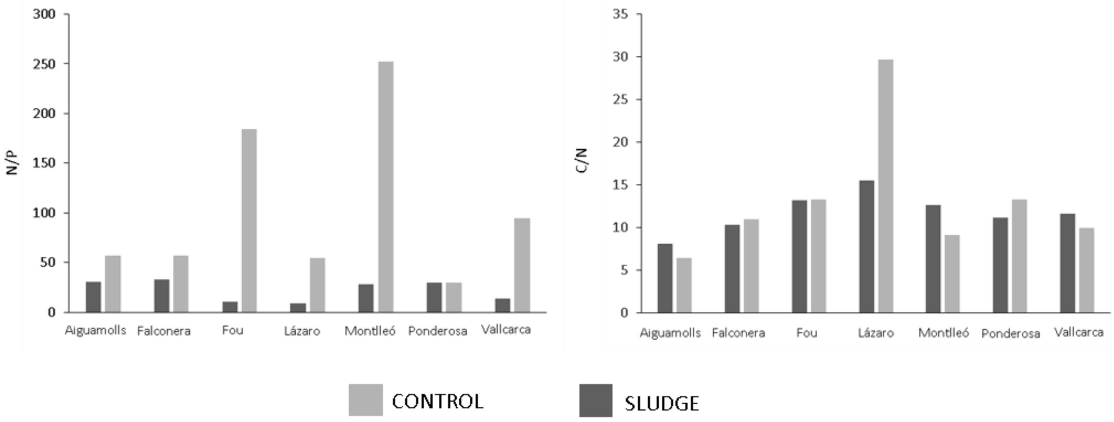

Figure 5.3. C:N and N:P ratios of sludge-amended and control Technosols ten years after construction, in the different sites studied. Regarding N:P ratio, it has been calculated using Kjeldhal N and available P.

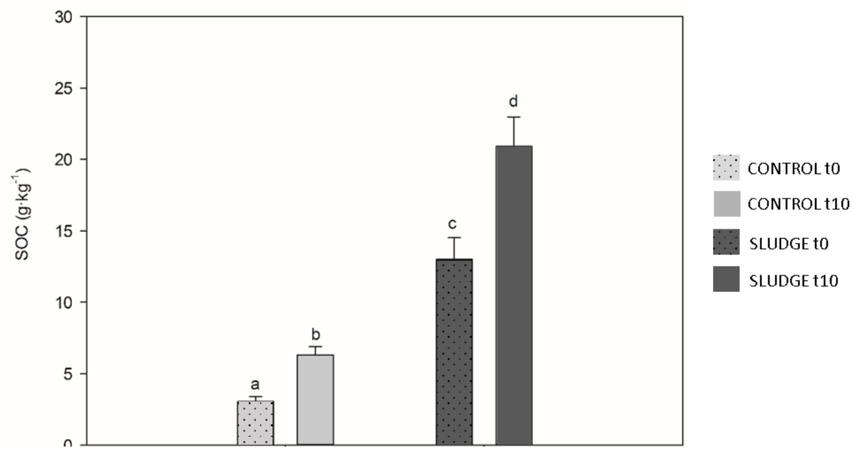

Figure 5.4. Initial (t_0, 4-6 months) and ten years (t_{10}) soil organic carbon (SOC) contents in sludge-amended and control Technosols of studied quarries. The error bars correspond to standard error and different letters indicate significant differences at a p<0.05 level.

Plant community and development

After 10 years, herbaceous cover was still dominant in most of the areas evaluated, irrespective of controls or sludge-amended plots. The average herbaceous cover over all plots was less than 50% (Figure 6.6). However, herbaceous vegetation was still more developed in amended plots, where organic debris accumulation was also higher (Figure 6.6). Regarding herbaceous species composition, sludge-amended Technosols showed a higher frequency of ruderal species, such as *Chenopodium album*, *Malva sylvestris* and *Cardus* spp. (Table S1), but they were not dominant. Colonization by native neighboring species was observed in both Technosol approaches. Some silt tolerant and halophyte plants, like *Salsola kali* or *Atriplex halimus* were more frequent in sludge-amended soils despite salinity not being significantly higher. Some invasive species like *Arundo donax* were identified in amended plots, despite the fact that their vegetation cover is minimal, and its presence cannot be attributed to sludge amendment but to its introduction as rhizomes in the exogenous soil used for Technosol construction.

Regarding the shrub and tree strata, sludge-amended plots presented higher woody cover (shrubs and trees), mainly due to enhanced pine growth, which explains the higher total plant cover in this treatment (Figure 6.6). The presence of shrubs and trees is mainly explained by plantation actions carried out after Technosol spreading. However, in most areas, recruitment of at least one wild shrub species took place, while this was also true but less usual for tree species. The most common tree species found was Aleppo pine (*Pinus halepensis*), although in some cases holm oak (*Quercus ilex*) was also present at much lower densities (see Table S1). Regarding shrubs, *Dittrichia viscosa,* an extremely common Mediterranean plant of the first stages of secondary succession, had colonized almost all the areas evaluated. Rosemary (*Rosmarinus officinalis*), mastic (*Pistacia lentiscus*), and cotton lavender (*Santolina chamaecyparissus*) were also found in many of the areas evaluated. Most specimens of these bushes were planted, although natural recruitment was also observed, especially in the case of mastic. There were no notable differences between treatments regarding the recruitment of shrub or tree species (see Table S1).

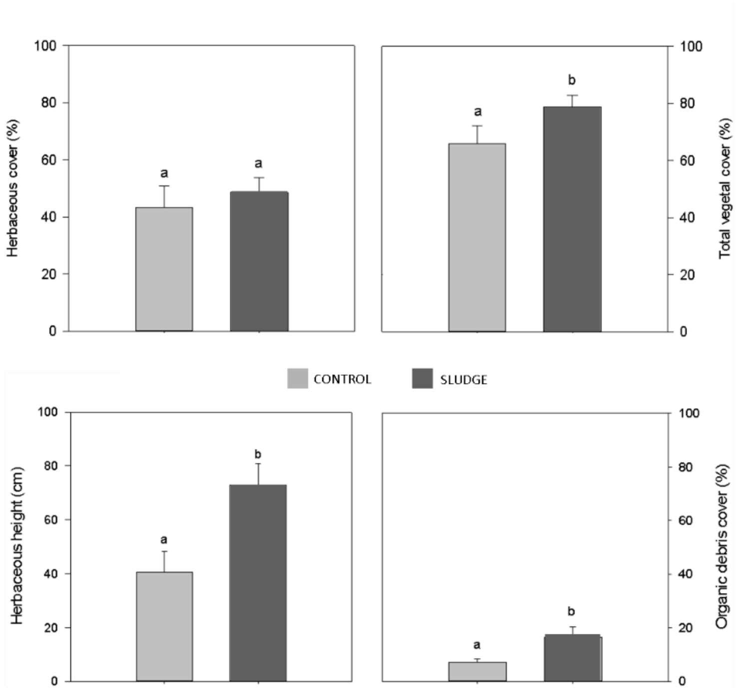

Figure 6.6. Percentage of plant cover types distribution (herbaceous, total vegetal, organic debris) and herbaceous development (height, cm) on sludge-amended and control Technosols 10 years after sludge application. The error bars correspond to standard error and different letters indicate significant differences at a $p<0.05$ level.

Discussion

After ten years, Technosols constructed using sewage sludge as an amendment had three times more SOC than the unamended ones. On one hand, it is clear that sewage sludge has contributed to changes in the organic matter content of the soil, but most of its organic matter is labile, which suggests that the direct effects of sludge on soil have only been transient (Tarrasón *et al.* 2010). On the other hand, plant debris from vegetation grown in restored areas contributes to increase SOC (Muñoz-Rojas *et al.* 2016) that tends to be stabilized and concentrated in the fine fraction (<2 mm). For this reason, the increase in SOC observed in amended plots after ten years is more likely due to the contributions of vegetation growth in the area than from the OC directly provided by the sludge. These differences were higher in extremely stony soils (<20% fine fraction) located in moderately rainy regions (700 mm annual precipitation), that allowed herbaceous vegetation to grow after sludge application and rain events. In contrast, fine soils in semi-arid regions (500 mm annual precipitation) did not accumulate such quantities of C in amended soils, due to the smaller positive impact of the amendment on growth of herbaceous vegetation, and consequently differences respect to control were also lower. This seems to be confirmed by Meersmans *et al.* (2012), who detected that organic carbon concentrations in fine particle-size fraction increase with increasing rock fragment content, and that spreading farmyard manure and slurry induces higher carbon concentrations mostly in wet and stony grasslands. Similarly, Arias *et al.* (2017) found gains of up to 10 Mg C ha^{-1} in the tilled layer (0–30 cm) of stony soils only after two years of irrigation.

After ten years of restoration works, the average SOC of soils amended with sewage sludge was relatively high, whilst the SOC content in the control areas was very low and clearly deficient (Carabassa *et al.* 2015; Carabassa *et al.* 2010). In any case, both amended and unamended soils were still far from the SOC average for Mediterranean forest soils (Rovira & Ramón Vallejo 2007; Lal 2005; Doblas-Miranda *et al.* 2013), and therefore probably far from any C saturation situation, especially considering the relatively high clay content of soils in this study.

Despite the fact that SOC fractioning was not available in this study, previous studies on Technosols from quarries confirmed that SOC tends to be more stabilized over time in sewage-amended soils, doubling the fraction of non-hydrolyzable carbon in the mid-term (Ojeda *et al.* 2015). Considering that many organic compounds of this non-hydrolyzable fraction are hard to mineralize (Rovira & Ramón Vallejo 2007), we state that this SOC fraction of the sludge-amended mine Technosols could be considered as sequestered carbon. Moreover, in this study all the Technosols had a high carbonate content, which contributes to the formation of aggregates and to the physical protection of SOC (Amézketa 1999).

Regarding the total amount of carbon sequestered in soils at a world level, the vast majority of the SOC reservoir is reported to be below 20 cm (Fontaine *et al.* 2007). This deep carbon is highly persistent because it is bonded to soil minerals and it is less accessible for decomposers (Wattel-Koekkoek *et al.* 2003). However, we only considered the first top 20 cm of soil in our carbon stocks estimations. Consequently we underestimated the total SOC, especially on amended plots, due to the strong plant growth in the first years after Technosols construction (Carabassa *et al.* 2018). This plant growth, that included trees, could have increased deep C through root growth, as shown by Simón *et al.* (2018) in mine Technosols in SE Iberian Peninsula 6 years after their establishment. Nevertheless, this plausible underestimation might be limited considering the reduced soil depth of the Technosols included in this study.

Regarding nutrient content, some control Technosols were clearly deficient and unbalanced in N and P. This fact might explain the low vegetation success in sites with relatively low hydric stress (700 mm annual precipitation), as N and P are the two main macronutrients limiting plant primary production in terrestrial ecosystems (Elser *et al.* 2007). This is especially true in Mediterranean forest ecosystems with calcareous soils that reduce P bioavailability (Sardans *et al.*, 2004). Furthermore, nutrient imbalances in the stoichiometric relationship between N and P can have significant impacts on soil functions, affecting the development of vegetation and microbial activity (Sardans *et al.* 2012).

On the contrary, sludge-amended Technosols still presented high N and P concentrations after 10 years, and in the specific case of N, more than in the short-term after sludge amendment. Although nitrogen leaching is plausible with the sludge treatment, it would mainly take place during the first four months due to the high mineralization rates of organic-N from sludge (Carabassa *et al.* 2018). After this period, leaching should decrease quickly due to reduced sludge decomposition rates and the enhanced nitrate absorption by the growing vegetation (Jordán *et al.* 2017; Tarrasón *et al.* 2008). As SOC becomes stabilized over time in sludge-amended Technosols (Ojeda *et al.* 2015), N stabilization in organic forms is also expected, as shown by the balanced C:N ratios observed. Regarding P, and despite its high levels, a low availability is expected due to the alkaline pH of the highly calcareous materials used for the Technosols construction, that causes a fast immobilization of P as calcium triphosphate (Tunesi *et al.* 1999).

Regarding total plant cover, significant differences between amended and control Technosols persisted after 10 years. However, despite herbaceous cover being similar in both treatments, herbaceous vegetation was more developed in sludge-amended areas, which was coupled to an enhanced accumulation of organic debris and the consequent higher content of SOC (Wambsganss *et al.* 2017). Even though the absolute herbaceous cover was less than 50% in both treatments, this was not of concern due to the effective protection against erosion because of the high stoniness of these soils. Moreover, sludge-amended areas had higher cover of woody plants, associated with the high fertilizing effect on pine growth, which in turn also plausibly contributed to the increased SOC. This tree-specific fertilizing effect has been described elsewhere and shown as higher tree growth ratios in soils amended with sludge for restoration purposes (López-Díaz *et al.* 2009; Tarrasón *et al.* 2014).

Regarding plant composition, and coupled to the higher N levels, higher relative abundance of nitrophilous species was observed in sludge-amended soils, though those species were not dominant in the community after ten years. Thus, the species composition in both types of Technosols were similarly ruderal, in agreement with previous studies conducted 5 years after sludge application (Moreno-Peñaranda *et al.* 2004). Despite the slowness of this process and the fact that woody species were also present in unamended Technosols (including natural recruitment), enhanced woody (shrubs and trees) species recruitment was observed in sludge-amended plots, which represents an important goal in the restoration of these areas due to the beneficial effect on ecosystem functioning (Soliveres *et al.* 2014). This show that Technosols constructed with mine spoils without organic amendment are less successful in terms of vegetation development and community complexity, mostly due to the extremely low fertility of the quarry substrates.

In summary, after a decade, Technosols constructed with moderate dosages of sewage sludge boosted soil organic matter enrichment and carbon sequestration, as they contained three times more SOC than the unamended ones. This was the result of increased primary production due to the high nutrient content of sludge, which was still visible after 10 years. Plant cover was also enhanced in Technosols receiving sludge, without causing strong changes in plant community but demonstrating higher development of shrubs and trees that might reflect a speed up in the natural succession process. All of these benefits are clearly linked to the two main soil ecosystem services that are intended to recover in the restoration of quarrying activities, which are biological habitat and carbon sequestration. Furthermore, we demonstrate that the valorization of 'in situ' mine spoils can be successful and improved by the use of sewage sludge, in agreement with the current principles of the circular economy.

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Supplementary information

Table S1. List of species identified in the areas evaluated and relative abundance depending on the contribution to the vegetal cover (1=minimal; 2=cover<25%; 3=cover 25-75%; 4= cover>75%). See the site and treatment abbreviations in Table 5.1.

Specie	**Family**	AIC	AIFA	FAC	FAFA	FOC	FOFA	LAC	LAFA	MOC	MOFA	POC	POFA	VAC	VAFA
Acer monspessulanum	Sapindaceae														2
Ajuga chamaepitys subsp. chamaepitys	Lamiaceae		1												
Anagallis arvensis	Primulaceae		1												1
Antirrhinum barrelieri	Plantaginaceae							1	1						
Arbutus unedo	Ericaceae														2
Arundo donax	Poaceae				1								1		
Asparagus acutifolius	Liliaceae				2	4			1						
Asphodelus fistulosus	Xanthorrhoeaceae			1											
Pallenis spinosa	Asteraceae						1		1		1	1		1	1
Atriplex halimus	Chenopodiaceae												1		
Avena sterilis	Poaceae		1	2	2	1	4	2	3		3	3	3	3	3
Avenula versicolor	Poaceae											1			
Brachypodium retusum	Poaceae		1												
Bromus madritensis	Poaceae	4	3												
Bromus sp.	Poaceae					4	3					1	1	1	1
Cardus sp.	Asteraceae			1			1		1		1		1		1
Catapodium rigidum	Poaceae													1	
Centaurea aspera subsp. stenophylla	Asteraceae											1			
Centranthus ruber	Valerianaceae					1	1		1						
Chenopodium album	Chenopodiaceae										1		1		
Chenopodium sp.	Chenopodiaceae				1				1						
Cistus albidus	Cistaceae							1							
Cistus sp.	Cistaceae	4													
Convolvulus althaeoides	Convolvulaceae			1									1		1

Specie	**Family**	AIC	AIFA	FAC	FAFA	FOC	FOFA	LAC	LAFA	MOC	MOFA	POC	POFA	VAC	VAFA
Coriaria myrtifolia	Coriariaceae												1		
Crepis sp.	Asteraceae	1	1	1	1			1	1	1	1		1	1	1
Cynodon dactilon	Poaceae			1				1	1						
Cynoglossum creticum	Boraginaceae														
Dactylis glomerata	Poaceae	1	1				1								
Datura stramonium	Solanaceae								1						
Daucus carota	Apiaceae						4	1						1	1
Delphinium peregrinum	Ranunculaceae		1												
Dittrichia viscosa	Asteraceae	1	2		2	3	3	4	1			1			2
Dorycnium hirsutum	Fabaceae					3									
Dorycnium pentaphyllum	Fabaceae						4								2
Echium vulgare	Boraginaceae		1	1	1			1	1					1	
Erica sp.	Ericaceae			2				1							
Eryngium campestre	Apiaceae		1												
Euphorbia sp.	Euphorbiaceae					1	3	1	1			1	1	1	1
Euphorbia terracina	Euphorbiaceae			1											
Foeniculum vulgare	Apiaceae			2	1		1					1	1	1	1
Galactites tomentosa	Asteraceae		1	1	1		3	1	1		2		1	1	1
Geranium. sp	Geraniaceae		1												
Globularia alypum	Plantaginaceae	3	4												
Helianthemum hirtum	Cistaceae	1													
Helichrysum stoechas	Asteraceae		2	1		4					1	1			
Hordeum vulgare	Poaceae			1								1	1		
Hyparrhenia hirta	Poaceae							1							
Hypericum sp.	Hypericaceae			1											
Laminacea	Lamiaceae								1						

Specie	**Family**	AIC	AIFA	FAC	FAFA	FOC	FOFA	LAC	LAFA	MOC	MOFA	POC	POFA	VAC	VAFA
Lolium rigidum	Poaceae									2	3				
Malva sylvestris	Malvaceae			1	1				1						1
Marrubium vulgare	Lamiaceae	1	1												
Medicago sp.	Fabaceae			1			2			2	1			1	1
Melica ciliata	Poaceae		1												1
Olea europaea subsp. sylvestris	Oleaceae			2	2										2
Parietaria judaica	Urticaceae														1
Phalaris minor	Poaceae		1												
Phillyrea latifolia	Oleaceae				2										
Phleum pratense	Poaceae						2					1			
Picris echioides	Asteraceae			1			1					1			1
Pinus halepensis	Pinaceae		4	4	4		1	1	4	3	4		4	4	4
Piptatherum miliaceum	Poaceae		2	3	4	1	4	2	3	2	3	3	3	3	3
Pistacea lentiscus	Anarcadiaceae			4	2	1		1	1				2	4	4
Plantago lanceolata	Plantaginaceae			1					1						
Plantago major	Plantaginaceae													1	
Psoralea bituminosa	Fabaceae			1				2	1			1			
Quercus coccifera	Fagaceae			2											
Quercus ilex	Fagaceae			2			2			3				4	4
Retama sphaerocarpa	Fabaceae					1	1								
Rhamnus alaternus	Rhamnaceae			2	4	3		1	2				1		
Ricinus communis	Euphorbiaceae														4
Rosmarinus officinalis	Lamiaceae	4	4	3			1								
Rubia peregrina	Rubiaceae			2				1							1
Rubus ulmifolius	Rosaceae														3
Rumex crispus	Polygonaceae						1								

Specie	**Family**	AIC	AIFA	FAC	FAFA	FOC	FOFA	LAC	LAFA	MOC	MOFA	POC	POFA	VAC	VAFA
Rumex sp.	Polygonaceae										1				1
Salsola kali	Amaranthaceae	1	4				1								
Sanguisorba minor	Rosaceae		1												
Santolina chamaeciparisus	Asteraceae				1			1	1			1	1		
Sedum sediforme	Crassulaceae							1	1			1			
Sideritis hirsuta	Lamiaceae			1											
Sideritis spinulosa	Lamiaceae		1												
Smilax aspera	Smilacaceae				2			1							
Spartium junceum	Fabaceae											2	1		
Thymus vulgaris	Lamiaceae					4									
Torilis arvensis	Apiaceae				1										
Trifolium sp.	Fabaceae					1									
Tusilago farfara	Asteraceae		2												
Ulex sp.	Fabaceae								1			1			
Verbascum thapsus	Scrophulariaceae		1												
S (species richness)		10	26	29	19	14	22	21	25	6	12	18	19	16	29

6. Soil restoration using compost-like-outputs and digestates from non-source-separated urban waste as organic amendments: limitations and opportunities

En aquest capítol s'avalua l'aptitud de bioestabilitzats (compost-like-outpus or bioestabilized) *i digests* (digestate) *produïts a partir de la matèria orgànica present a la fracció resta dels residus municipals, per a ser utilitzats com a esmenes orgàniques en la construcció de tecnosols. Tot i presentar aptituds per a ser utilitzats com a esmenes, cal controlar molt bé la qualitat d'aquests residus per evitar l'aportació d'impureses i altres contaminants al sòl, així com matèria orgánica poc estabilitzada, especialment quan s'utilitzen bioestabilitzats. Aquesta article s'ha publicat a la revista* Journal of Environmental Management (Carabassa, V., Domene, X., Alcañiz, J.M. (2020) Soil restoration using compost-like-outputs and digestates from non-source-separated urban waste as organic amendments: Limitations and opportunities. J. Environ. Manage. 255. https://doi.org/10.1016/j.jenvman.2019.109909).

Abstract

Soil rehabilitation in the context of the restoration of quarries, dumping sites, or road slopes often requires the prior addition of organic amendments to improve the substrates used for Technosol construction. Bio-wastes coming from advanced Mechanical-Biological Treatment Plants, mainly compost-like-outputs (CLO) and digestates (DGT), are new and suitable sources of organic matter potentially useful as organic amendments for this purpose, in an approach clearly fulfilling the principles of circular economy. In order to assess the suitability of these materials, a complete physicochemical and biological evaluation was carried out, including an ecotoxicological evaluation to discard hazardous effects on key soil fauna groups. Field experiments were also carried out on several road slopes and a dumping site. The stability degree of organic matter and the impurities content could be restricting parameters for the use of CLO in soils. Low stability degree decreased plant development in the initial stages of restoration. Moreover, the high heterogeneity in terms of physicochemical parameters of the different CLOs assessed is a serious constraint to making generalizations about its use. In contrast, composition of DGTs was more stable between plants and batches, and presented low impurities and high N contents that make them more suitable for applying to soil and promoting plant development. Regarding the application rates, DGT application at 20 $g \cdot kg^{-1}$ clearly improved plant growth after sowing, without compromising recruitment. However, application at 80 $g \cdot kg^{-1}$ did not ameliorate seed germination and plant growth, in either CLO or DGT treatments, and increased N-leaching and toxicity risk to soil mesofauna in DGT amended Technosols.

Keywords

Technosol, bio-wastes, Mechanical-Biological Treatment Plants, compost impurities, stability degree, ecotoxicity risk

Highlights

- A case per case evaluation should be done before the application of compost-like-outputs and digestates from advanced Mechanical-Biological Treatment Plants as organic amendments for Technosols construction
- The stability degree and the impurities content of compost-like-outputs could be restricting parameters for their application to soils
- High heterogeneity between batches and production plants hinders making strong generalizations related to the use of compost-like-outputs
- Digestates have a more stable composition and present lower impurities and higher N contents than compost-like-outputs
- Application rates higher than 2% should be avoided

Acronims

- CLO: Compost-Like-Output
- CLO-A: Compost-Like-Output from MBTP A
- CLO-A_{OL}: technosoil from Olost trial constructed using CLO-A
- CLO-B: Compost-Like-Output from MBTP B
- CLO-B_{OL}: technosoil from Olost trial constructed using CLO-B
- CLO-D: Compost-Like-Output from MBTP D
- CLO-D_{OL}: technosoil from Olost trial constructed using CLO-D
- C+DGT: compost-like-output from digestate, produced in MBTP E
- C+DGT_{OL}: technosoil from Olost trial constructed using C+DGT
- CTR: control technosoil used for physicochemical, biological, and ecotoxicological characterization of CLO/DGT mixtures
- CTR_{LM}: control technosoil from Lloret de Mar trial
- CTR_{OL}: control technosoil from Olost trial
- CTR_{TE}: control technosoil from Terrassa trial
- DGT: digestate
- DGT-C: Digestate from MBTP C
- DGT-C_{OL}: technosoil from Olost trial constructed using DGT-C
- DGT-C_{TE}: technosoil from Terrassa trial constructed using DGT-C
- DGT-F: Digestate from MBTP F
- DGT-F_{OL}: technosoil from Olost trial constructed using DGT-F
- DGT-F_{TE}: technosoil from Terrassa trial constructed using DGT-F
- DGT-L_{LM}: technosoil from Lloret de Mar trial constructed using DGT-L
- HD: high dose (80 $g \cdot kg^{-1}$)
- LD: low dose (20 $g \cdot kg^{-1}$)
- LM: Lloret de Mar
- MBTP: Mechanical-Biological Treatment Plant
- OL: Olost
- TE: Terrassa

Introduction

With global economic and social development, human activities like mining, road construction or landfilling have grown rapidly (Yang *et al.*, 2016). As an example, mining activities occupy more than 1% of the Earth's surface and continue to grow (Šálek, 2012). These activities are usually developed in natural landscapes, producing severe impacts on all the ecological compartments, their ecological functions, and the ecosystem services that they provide to the society. In order to recuperate these services (or establish new ones), these spaces should be restored. The most common restoration practices include topographic modeling, spreading of a soil layer, and posterior vegetation establishment (Werner *et al.*, 2001; Blanco and Lal, 2010).

In the context of restoration following these human activities, manufactured soils, so-called Technosols, a taxonomic group of the World Reference Base (Schad and Dondeyne, 2017), could be a viable restoration practice when the availability of suitable natural soils is limited (Watkinson *et al.*, 2017). This technology is emblematic of the issues we face for the management of soils of the Anthropocene (Leguédois *et al.*, 2016). The use of organic waste for Technosols construction is a widely used practice in soil restoration (Asensio *et al.*, 2013; Lomaglio *et al.*, 2017; Watkinson *et al.*, 2017), shown to speed up the biological colonization of a relatively inert initial mineral substrate. Amendment properties of this organic waste are related to the ability of the contained organic matter to contribute in maintaining the soil humus and nutrient balance (Magdoff and Weil, 2004). This ability depends on the degradability of the organic matter, i.e. recalcitrance vs. labile organic fraction contents (Rumpel *et al.*, 2002). Pedogenic processes occurring in Technosols are similar to those occurring in natural soils (Leguédois *et al.*, 2016), although the ingredients used during their construction can strongly influence their evolution, the resulting soil properties and the capacity of the soil to provide ecosystem services. However, they tend to have a fast evolution compared to natural soils, including in terms of biological activity (Leguédois *et al.*, 2016), which is a crucial factor for the provision of ecosystem services in soils (Tate, 2005) and a driver of soil pedogenesis (Frouz *et al.*, 2013).

Approximately 120 to 140 million bio-waste tons are produced every year in the EU, which corresponds to approximately 300 kg per European Union (EU) citizen per year (JRC, 2011). According to the Waste Framework Directive (EC, 2008), bio-waste corresponds to waste coming from gardens and parks, food and kitchen waste from households, restaurants, caterers and retail premises, as well as the waste coming from food processing plants, therefore not including forestry or agricultural residues. Throughout Europe, about 40% of bio-waste is still landfilled (up to 100% in some Member States) (EC, 2010), a practice that is not in agreement with the guiding principles of EU Waste and Sustainable Resource Management Policy, notably the "waste hierarchy" that should underlie all national waste policies (JRC, 2011). According to a global circular economy perspective, the current EU objectives aims for 65% of all municipal waste produced to be recycled before 2030, with only 10% disposed of in landfills (EC, 1999).

In an attempt to reduce the environmental impacts of biodegradable waste, mechanical and biological treatments are being used as a waste management process in many countries. Mechanical-Biological Treatment Plants (MBTP) attempt to mechanically separate the biodegradable and non-biodegradable components (Donovan *et al.*, 2010). The non-biodegradable components are then sent for reprocessing or are landfilled, whereas the biodegradable components are reduced through composting (generating Compost-Like-Output, herein referred to as CLO) or by anaerobic digestion (generating digestate, herein referred to as DGT). Additionally, sometimes DGT is later composted obtaining compost from DGT (herein referred to as C+DGT).

The organic fractions of municipal solid waste are heterogeneous in terms of composition and source. The sustainable management of these waste products represent a challenge due to policy shifts and the increasing pressure on landfills (Abdullahi *et al.*, 2008). If the recovery and biological treatment of bio-waste is properly performed, the resulting organic end-products may comply with quality requirements, while in other cases they can contain impurities (plastics, glass, metals) which might hamper their quality meaning that they do not fit local legislation or preventing their acceptance by the end-users.

The main goal of this article is to assess the limitations and opportunities of the use of CLO and DGT for technosols construction in soil restoration works. The study considers this question at different scales, from the lab to the field, through a detailed characterization of waste products and their respective soil mixtures. This includes an ecotoxicological evaluation at the laboratory and greenhouse scale, and field trials in natural conditions.

Material and methods

DGT/CLO origin and characterization

The study was carried out in Catalonia (NE Iberian Peninsula), a region of 7 million people, representative of the Mediterranean EU, that produced 3.98 Tg of municipal solid waste in 2018 (ARC, 2019). Six of eight of the MBTP existing in Catalonia in 2016 were included in the study. The MBTP selection was carried out in order to include all of the variety of treated municipal organic waste products produced in these plants: three producing CLO (plant codes A, B, D), two producing DGT (plant codes C, F) and one producing C+DGT (plant code E). A composite sample of 15 kg of each batch, to be used for mixing with the soil, was taken from the respective MBTPs, taking sub-samples from different parts of the piles where batches were stocked. Samples were stored at 4 °C in plastic drums (2-3 days). Particle size and impurities content were determined in a 2 kg sample aliquot after drying at 60 °C for three days. For all MBTPs, sub-samples of each batch (1 kg) were sent, refrigerated in plastic bags, to external labs for a complete physicochemical characterization, shown in Table S1. In order to assess any seasonal variability of CLO composition, the MBTPs provided analytical information from representative samples obtained in different seasons and years (Table S2).

The characterization of CLO and DGT included the following parameters. Dry matter, pH, electrical conductivity (EC), organic matter, total nitrogen, ammonia-N, and the maturity test (Rottegrade) that were measured according to EN 13040 (2008), EN 13037 (2012), EN 13038 (1999), EN 13342 (2000), EN 13652 (2002), EN 13039 (2012), and EN 16087-2 (2012), respectively. Elemental analysis of P, K, Ca, Mg, Fe, Cd, Cr, Cu, Hg, Ni, Pb and Zn was carried out by ICP-OES according to ISO 11885 (2007) and EN 13650 (2002). Sample treatment consisted of drying at 110 °C, grinding, calcination, acid digestion with 3N HNO_3 and filtration. Presence of pathogens (*Salmonella* and *E. coli*) was evaluated according to prEN 15215-1 (2006) and ISO 5679 (1997). Particle size distribution content was determined by dry sieving, and impurities by hand sorting, and reported as mass percentage (Huerta *et al.*, 2010). The stability degree was measured as the non-hydrolyzable (stable) organic matter (percentage of organic matter remaining in the sample residue after an acid hydrolysis) as described in Huerta *et al.* (2010) and FCQAO (2002). This method removes the more labile fraction of an organic material, so the remaining organic material is considered as recalcitrant. The method consists of two consecutive steps of hydrolysis with sulfuric acid: the first one using acid at 72% at room temperature where celluloses are hydrolyzed, and a second one using diluted acid at boiling point where other polysaccharides, proteins and lipids are hydrolyzed, with lignin and humic substances remaining.

Physicochemical, biological, and ecotoxicological characterization of DGT/CLO-soil mixtures

The Bw horizon of a clay-loam calcareous soil (*Calcixerept* according to Soil Survey Staff, 2014) was sieved to 5 mm and then mixed with respective CLO/DGT at increasing concentrations (0, 20, 80 $g{\cdot}kg^{-1}$). This amended soil intended to mimic the worst case in soil restoration, when topsoil is not available and more or less sterile mineral fractions are used (mining wastes, deep soil layers from constructions), requiring an organic amendment before its use for restoration purposes (see Table 6.1). The effect on soil water retention capacity was assessed gravimetrically. The organic matter content was determined through acid dichromate oxidation (Walkley-Black) and loss on ignition (American Society of Agronomy, 1982), and its quality by acid hydrolysis as described by Raya-Moreno *et al.* (2017). Microbial activity was assessed by measuring the basal respiration (Alef and Nannipieri, 1995). The soil pH (1:2.5 (w:v) soil:water extract) and salinity (as electrical conductivity of 1:5 (w:v) soil:water extract) were assessed according to the American Society of Agronomy Standards (1982). Finally, soluble elements were determined on 1:5 (w:v) soil:water extracts at the beginning and at the end of the experiment by ionic liquid chromatography (DIONEX DX-100 Ion Chromatograph system).

To either evaluate the habitat function and the potential negative effects on plants of the selected organic amendments, the soil described above was mixed with each CLO or DGT. Thirty-nine pots (three per treatment) used as lysimeters were filled with about 2 kg of soil-organic amendments mixtures (at 0, 20, 80 $g{\cdot}kg^{-1}$). Pots were sown with wheat (*Triticum aestivum var Botticelli*), at a density of 176 $seeds{\cdot}m^{-2}$, and grown in controlled greenhouse conditions at an approximately stable moisture content (50% water holding capacity) for three months.

Table 6.1. Physicochemical characterization of the soil material (Bw horizon) used as substrate to be mixed with compost-like-outputs (CLO) and digestates (DGT) for laboratory and greenhouse experiments.

Parameter	**Units**	**B horizon**
Moisture 105°C	%	1.3
pH (1:2.5 w:v)		8.47
Electrical conductivity (1:5 w:v, 25°C)	dS/cm	0.128
Weight loss (375°C)	%	2.5
Weight loss (550°C)	%	4.0
Organic matter (W&B)	%	0.5
Resistant organic matter	% OM	62
Kjeldahl nitrogen (N)	%	0.032
Equivalent calcium carbonate	%	40
Clay D < 0.002mm	%	27.2
Fine silt 0.002 < D < 0.02mm	%	50.8
Coarse silt 0.02 < D < 0.05mm	%	10.5
Sand 0.05 < D < 2mm	%	11.5
Texture class	USDA	CLAY LOAM
Cr	$mg \cdot kg^{-1}$	36.0
Ni	$mg \cdot kg^{-1}$	22.8
Pb	$mg \cdot kg^{-1}$	11.7
Cu	$mg \cdot kg^{-1}$	25.5
Zn	$mg \cdot kg^{-1}$	136.0
Hg	$mg \cdot kg^{-1}$	<0.4
Cd	$mg \cdot kg^{-1}$	<0.5

Seed germination was monitored for the first two weeks after sowing through direct observation, while plant elongation was determined monthly during the three months of the experiment, measuring all the germinated plants. Plant biomass was measured at the end of this period harvesting all the plants, weighing (wet weight), drying at 60°C for three days, and weighing again (dry weight). After weighing, all the plant spikes were cut and weighed separately. All of the soil in each pot was dried and sieved at 2 mm before proceeding to the soil analysis.

In order to evaluate the habitat function and the ecotoxicological risks for soil fauna of the CLO/DGTs applications, the ISO 11267 survival and reproduction test of Collembola (*Folsomia candida*) (ISO, 1999) was performed. This test was applied to the same samples that were used in the plant bioassay, so the tested dosages were the same.

Field pilot tests

In order to upscale and test the suitability of CLO/DGT from MBTP as soil organic amendments in real restoration scenarios, three pilot field trials were carried out in different degraded land systems: a landfill slope (Lloret de Mar, LM) and two road slopes (Olost, OL; Terrassa, TE), located in a variety of Mediterranean climatic scenarios (Table 6.2) and soil types (Table 6.3). The organic waste amendments were mixed with the local soils at a dose of 20 $g{\cdot}kg^{-1}$, except for DGT-C_{OL}, which was applied at a dosage of 40 $g{\cdot}kg^{-1}$ in the Olost site (Table S3). After mixing, the amended soil (30 -50 m^3) was spread in >50 m^2 plots (three plots per treatment) randomly distributed. The plots were sown using commercial seed mixtures adapted to the specificities of each area (see Table S4) at a density of 30 $g{\cdot}m^{-2}$.

Vegetation cover was monitored four, five and six months after soil seeding, taking orthogonal pictures of the plots and measuring soil covered by vegetation through ENVI image analysis software through photogrammetry. Plant biomass was determined by harvesting all the vegetation in 60 cm diameter circles randomly distributed in triplicate in each plot. Additionally, flora inventories were done by identifying all the species present in the plots, and species abundance was determined by qualitative observation of their respective cover. Soil sampling was done using an Edelman auger and taking 10 sub-samples of the first 20 cm of soil per plot just after amendment application (T0) and 4 months after (T1).

Table 6.2. Pilot test sites, type and location, climatic conditions (precipitation, temperature and Köppen class), slope characteristics (facing, landform, steepness and length) and organic waste tested (CLO= compost-like-output, DGT= digestate, C+DGT: compost from digestate).

Site	**Restoration type**	**Latitude (N)**	**Longitude (E)**	**Mean annual precipitation (mm)**	**Mean annual temperature (°C)**	**Köppen-Geiger class**	**Facing**	**Landform type**	**Maximum slope (°)**	**Maximum slope length (m)**	**Organic waste type**
Olost (OL)	Road slope	41° 59´ 4´´	2° 6´ 13´´	690	10,2	Cfa	W	Steeped slope	45	150	CLO, DGT and C+DGT
Terrassa (TE)	Road slope	41° 33´ 4´´	2° 0´ 4´´	620	14,8	Csa	NE	Steeped slope	43	80	DGT
Lloret de Mar (LM)	Landfill	41° 43´ 23´´	2° 51´ 2´´	650	14,5	Csa	SW	Steeped slope	33	50	CLO

Table 6.3. Physicochemical parameters of the soil materials used as substrate to be mixed with compost-like-outputs (CLO) and digestates (DGT) at the three pilot tests. Unamended substrate of Olost road slope (CTR_{OL}), of Terrassa road slope (CTR_{TE}) and Lloret de Mar landfill (CTR_{LM}).

Parameter	**Units**	**CTR_{OL} (road slope)**	**CTR_{TE} (road slope)**	**CTR_{LM} (landfill)**
pH		7.79	8.32	7.55
Electrical cond. at 25°C	dS/cm	1.385	0.166	0.093
Total Organic Matter (W&B)	%	1.7	<0.50	<0.50
Kjeldhal N	%	0.117	0.021	<0.020
Equiv. calcium carbonate	%	31	19	3
Clay D < 0.002mm	%	16.7	13.9	6.0
Fine silt 0.002 < D < 0.02mm	%	24.6	20.7	7.5
Coarse silt 0.02 < D < 0.05mm	%	13.8	14.9	4.7
Sand 0.05 < D < 2mm	%	44.9	50.5	81.8
Texture class	USDA	Loam	Loam	Sandy loam
Cr	$mg \cdot kg^{-1}$	25.5	17.4	<10.0
Ni	$mg \cdot kg^{-1}$	21.4	17.9	7.4
Pb	$mg \cdot kg^{-1}$	8.2	13.7	7.9
Cu	$mg \cdot kg^{-1}$	<20.0	20.9	<20.0
Zn	$mg \cdot kg^{-1}$	65.9	61.0	32.0
Hg	$mg \cdot kg^{-1}$	0.7	<0.4	<0.4
Cd	$mg \cdot kg^{-1}$	<0.5	<0.5	<0.5

Data analysis

Analysis of Variance (1 way-ANOVA and repeated measures ANOVA) was used to examine differences between treatments (0, 20, 80 $g \cdot kg^{-1}$) and waste types based on physicochemical characterization of soil mixtures (TOC, resistant C, soluble C, C-CO_2 respired), greenhouse experiment measurements (germination rate, plant height, plant and spike weight) and ecotoxicological risk assessment (survival and reproduction). Additionally a correlation analysis was done for chemical and ecotoxicological parameters of CLO/DGT mixtures. Regarding field pilot tests, differences in vegetation cover measures were examined between treatments and time. The cut-off for statistical significance throughout the manuscript was fixed at $p=0.05$.

Results

Suitability as organic amendments and comparison between CLO and DGT

A relatively high heterogeneity was observed in some analytical parameters (humidity, pH, conductivity, impurities, N-forms) between MBTP and between batches in the same plant (Tables S1 and S2). Due to the fact that they had undergone very different treatments, CLO and DGT presented very different moisture contents: while the DGT had a dry matter content below 50%, CLO had a much lower humidity content, generally above 60% dry matter. Regarding salinity, clear differences between CLO and DGT were observed with higher EC values in CLO compared to DGTs, with the exception of CLO-A that had an unusually low EC, particularly considering the values obtained from other batches of the same product (see Table S2 for 2013 batches). However, at least for the batches included in this study (Table S1), electrical conductivity of CLO was less than 7 $dS{\cdot}m^{-1}$, with the exception of CLO-D, that reached 9.2 $dS{\cdot}m^{-1}$. Particle size distribution of the CLO samples (Table S5) showed a clear dominance of the fraction <2 mm, although large differences between the treatment plants were observed in this parameter. The main difference was the predominance (in volume) of a fibrous fraction in CLO-A, mostly consisting of paper remains. There was also a notable contribution of the wood fraction greater than 10 mm in the C+DGT, which resulted from the structural material added for composting, consisting on pruning waste. Regarding DGT, the entire sample passed through a 2 mm sieve except some impurities, which were of a low proportion. This was mainly due to (1) the sieving process of the organic fraction performed before its digestion, which strongly reduced the impurities content and homogenized the organic matrix, and (2) to the anaerobic digestion that promoted a higher degradation (and fragmentation) of organic particles compared to aerobic stabilization. The relatively high impurities content in CLO-D and F was mainly due to the presence of glass fragments (Table S6) that can be explained in CLO-D by a failure in the sieving line as reported by MBTP engineers. Other analyses provided by this plant (see Table S2) showed an average value of impurities close to 5% in 2013 and 3% in 2014, although in 2013 one batch (January) reached 14%.

Regarding heavy metals, almost all the samples included in this study (except CLO-D) are within the Spanish limit values for compost application to agriculture (BOE, 2013), despite the fact that some of them exceed the EU end-of-waste limit values based on the 2008 IPTS pilot study on compost/digestate (IPTS, 2008), which are more restrictive (see Table S7).

In terms of organic matter contents, all the samples contained more than 40% in weight, as determined by loss-on-ignition (EN 13039, 2012). CLO-A had a very high content, around 77%, probably due to fibrous paper pulp remains that are present in a high proportion as this MBTP receives noticeable amounts of this type of waste. On the contrary, C+DGT was the one with the lower content of volatile solids (48%), plausibly linked to the aerobic stabilization following the anaerobic digestion process. Maturity of CLO, measured with the Rottegrade test, showed a wide variation, from degree II (in CLO-A) to V (in CLO-B and C+DGT), sometimes showing higher stability (measured by acid hydrolysis) in less mature materials (CLO-

A vs. CLO-B). Wide variation in the ammonia-N contents was also measured, from 0.29% to 1.29%. (see Table S1).

CLO and DGT effects on soil properties

CLO and DGT increased soil water retention capacity when applied at 80 g·kg^{-1}(see Table S8), while such an effect was not detectable at lower dosages. At the same time, high dosage DGT treatments retained more water than CLO, especially in the case of CLO-C, where this trend was also detected at the low dosages (see Table S8). Both CLO and DGT are prone to increase soil EC due to the abundance of soluble compounds (see Table S8) resulting from the organic matter mineralization during their production at MBTP, but also after their application to soil (do Carmo *et al.*, 2016). Just after application (EC_0) values were not very high, with the exception of CLO-B that reached 1.6 dS m^{-1} at the 80 g kg^{-1} dose. After three months of greenhouse incubation, EC tended to decrease in all the CLO treatments, while in DGT treated soils EC increased significantly, reaching relatively high values at the higher DGT-C dosages (see Table S8). Regarding the composition of soluble elements in water extracts (Table S9), important differences existed between DGT and CLO. Nitrogen forms in digests treatments had a very high concentration of mineral nitrogen (nitrate, nitrite and ammonium) at the end of the greenhouse incubation (3 months), coupled to the conductivity increase, which can be attributed to the mineralization of the organic matter of the digest incorporated in the soil. In contrast, CLO treatments presented low soluble element contents, below those of controls. Concerning dose effects on soluble nitrogen forms, N immobilization was evident in CLO. While the initial mineral-N forms concentrations were clearly higher than controls at the 80 g·kg^{-1} dosage, these concentrations became lower than controls three months after. The C+DGT had the lowest mineral-N concentrations (nitrite, nitrate, and especially ammonium). Concentrations of soluble phosphate were moderate in all the treatments, being the highest in the high dosage of CLO-D (5.1 mg·kg^{-1}), despite the fact that after three months digest C was the one with higher values (3.8 mg·kg^{-1}). Furthermore, sulfate content also increased importantly after three months, agreeing with conductivity values. Similarly, sodium and chloride concentrations were high in all treatments at the high dose. Regarding the pH effects (see Table S8), the application of the CLO/DGT caused a slight acidification compared to controls (from 8.74 to 7.39).

As can see in Figure 6.1, TOC was above 2% in high application dosages except for the C+DGT. This case is probably due to its lower proportion of organic matter derived from the stronger stabilization process already mentioned. In fact, C+DGT had the highest proportion of acid hydrolysis-resistant organic carbon expressed on a TOC basis, and the lowest proportion of soluble C (Figure 6.1). C mineralization, assessed by basal soil respiration measurements, was inversely coupled with the content of resistant organic matter of the treated soils. CLO-B presented the highest respiration rate (see Figure 6.2), while DGT-C also showed high respiration rates during the first 15 days, and then decreased rapidly. DGT-F and C+DGT presented the lowest respiration rates, in agreement with their low TOC and the proportionally high (referred to TOC) resistant organic matter content.

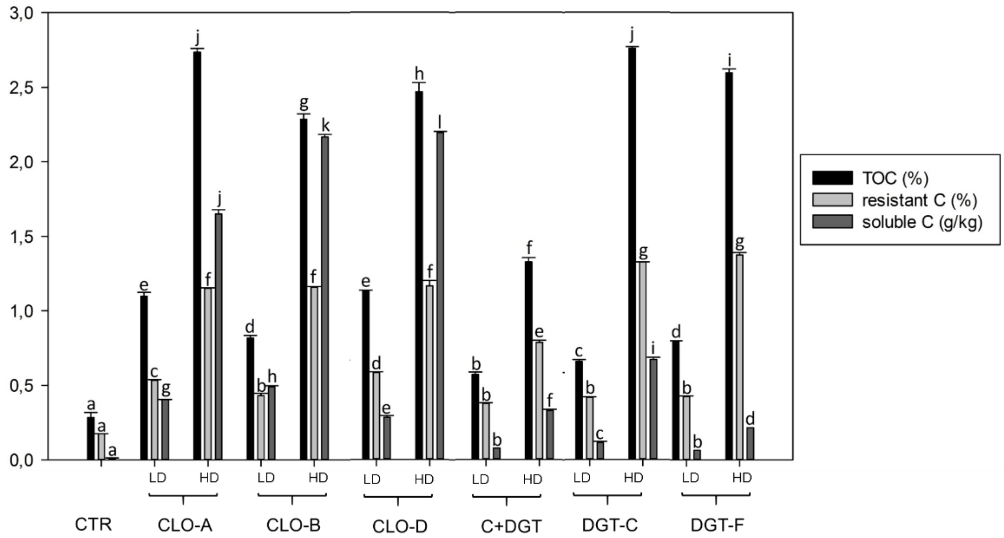

Figure 6.1. TOC (%), soluble-C ($g{\cdot}kg^{-1}$) and resistant C (%) in a soil amended with high doses (HD, 80 $g{\cdot}kg^{-1}$) and low doses (LD, 20 $g{\cdot}kg^{-1}$) of compost-like-outputs (CLO) or digestates (DGT). Error bars correspond to standard error. Letters indicate a significant difference between respective treatments according to Fisher's test ($p < 0.05$). n=39.

At the end of the incubation, respiration rates dropped to that of the control (<1 mg $C\text{-}CO_2{\cdot}day^{-1}$), especially for the treatments with C+DGT and DGT-F. Respiration rates in the controls were low and constant along the incubation time as expected, indicating the lack of mineralizable native carbon sources in this soil (see Table S10). Respiration rates for the less stabilized products (CLO-A, CLO-B, DGT-C, CLO-D) were high, and fitted a biphasic model. In the first phase, labile organic matter was fast mineralized and after this, respiration rate diminished (stabilization phase). In general, mineralization was globally high during the first 30 days (2.8 mg $C\text{-}CO_2{\cdot}day^{-1}$, as average) and after this period remained stable (1 mg $C\text{-}CO_2{\cdot}day^{-1}$ between 30 and 90 days). After the stabilization period, the less stabilized product (CLO-B) respired almost four times more than the most stable one CLO (C-CLO) (see Figure 6.2).

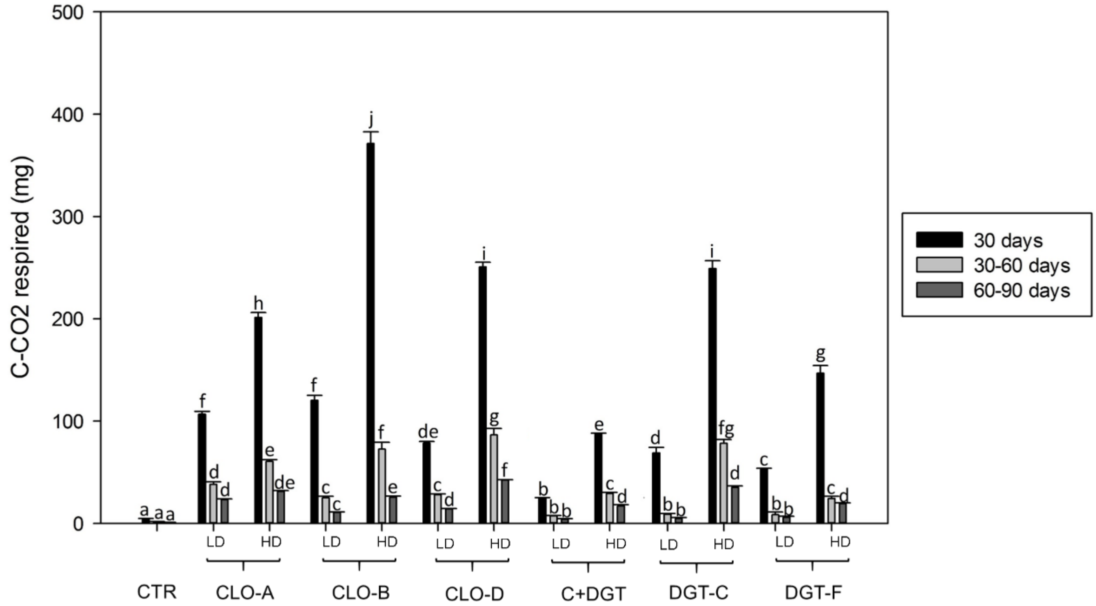

Figure 6.2. C-CO_2 respired (mg) over 90 days incubation in soil samples amended with high doses (HD, 80 g·kg^{-1}) and low doses (LD, 20 g·kg^{-1}) of compost-like-outputs (CLO) or digestates (DGT). Error bars correspond to standard error. Letters indicate a significant difference between respective treatments according to Fisher's test ($p < 0.05$). n=39.

Ecotoxicity risk assessment of CLO and DGT applications

Regarding plant germination, slightly higher rates were detected in both, C+DGT and CLO-B, irrespective of the dosage and the salinity of the soil-organic residue mixtures (Figure 6.3). On the other hand, a slight germination inhibition effect was observed in DGT-F at 80 g·kg^{-1} (HD).

In relation to plant growth, it is worth noting the satisfactory growth in DGT treatments and the poor development in the CLO ones (Figure 6.4). These differences were clearly observable at the first stages of development, one month after sowing, with a two-fold higher growth in DGT treatments than CLO ones, that were below the control level. Moreover, chlorosis was observed in controls and some CLO treatments at this stage. Such strong differences between CLO and DGT treatments after one month of sowing were mitigated at two months (see Figure 6.4), although DGT treatments were still showing more advanced plant development. The initial growth inhibition observed in CLO treatments was restricted to CLO-D and A, although chlorosis of leaves was clear in all the CLO treatments. Two months after sowing, controls and CLO treatments still presented chlorotic leaves, despite it being less marked in CLO than in controls. Aerial plant biomass corroborated the differences between CLO and DGT (see Figure 6.5). Spike biomass followed the same trends observed for total biomass, with DGT treated soils showing correct development of the spikes, while the treatments with CLO produced smaller spikes with lower number of seeds compared with the control (see Figure 6.5).

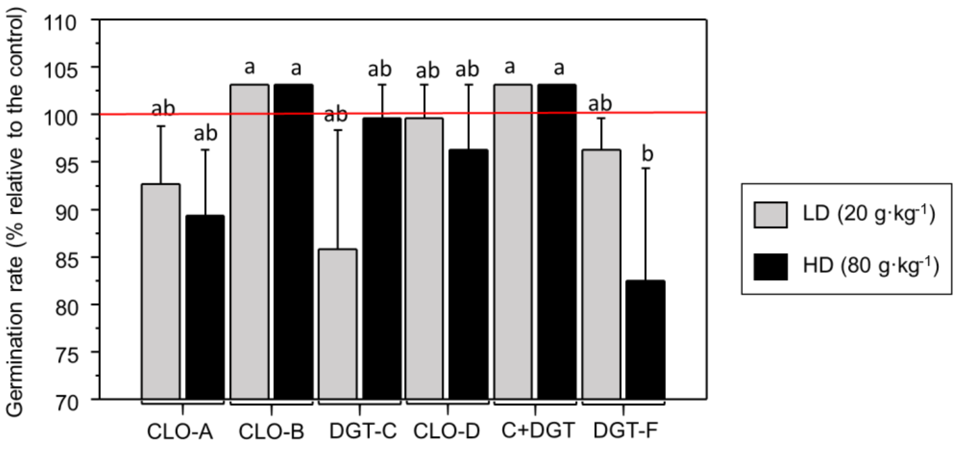

Figure 6.3. Wheat germination (%) relative to control, in soil samples amended with high doses (HD, 80 g·kg^{-1}) and low doses (LD, 20 g·kg^{-1}) of compost-like-outputs (CLO) or digestates (DGT). Red line indicate the reference value (CTR, 100%). Error bars correspond to standard error. Letters indicate a significant difference between treatments according to Fisher's test (p <0.05). n=39.

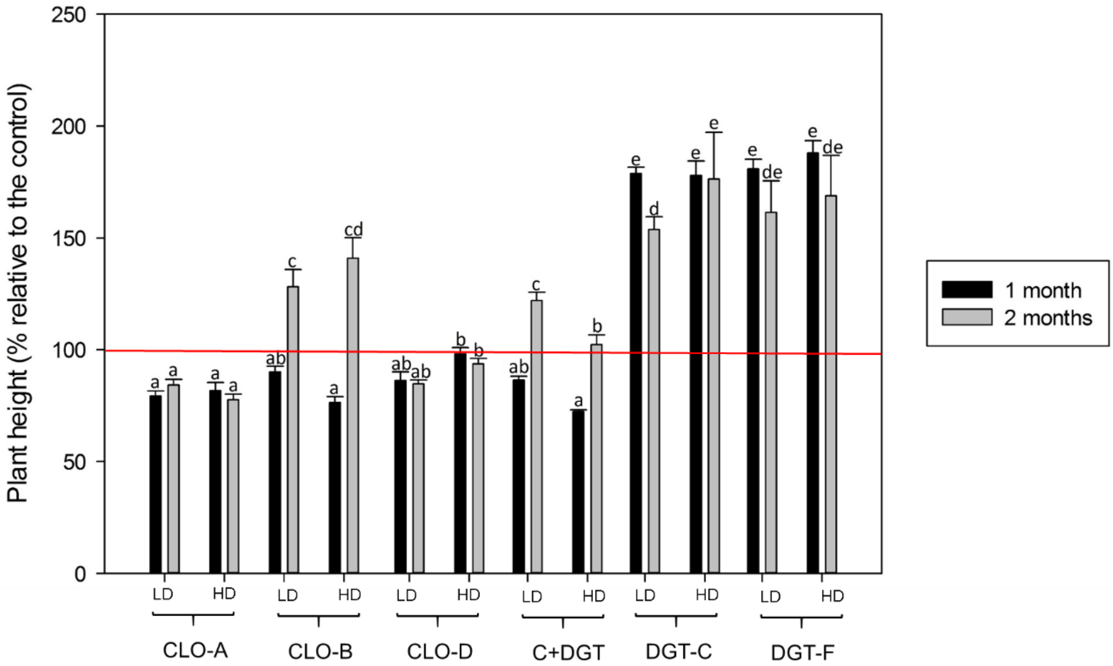

Figure 6.4. Height of wheat plants growth in CLO/DGT amended soil, relative to the control, as a function of the applied treatment, one month and two months after sowing. The treatment codes indicate the waste tested (CLO, DGT, C+DGT) and dose (HD: 80 g·kg^{-1}; LD: 20 g·kg^{-1}). Red line indicate the reference value (CTR, 100%).The error bars correspond to standard error. The letters indicate a significant difference according to Fisher's test (p <0.05). n=78.

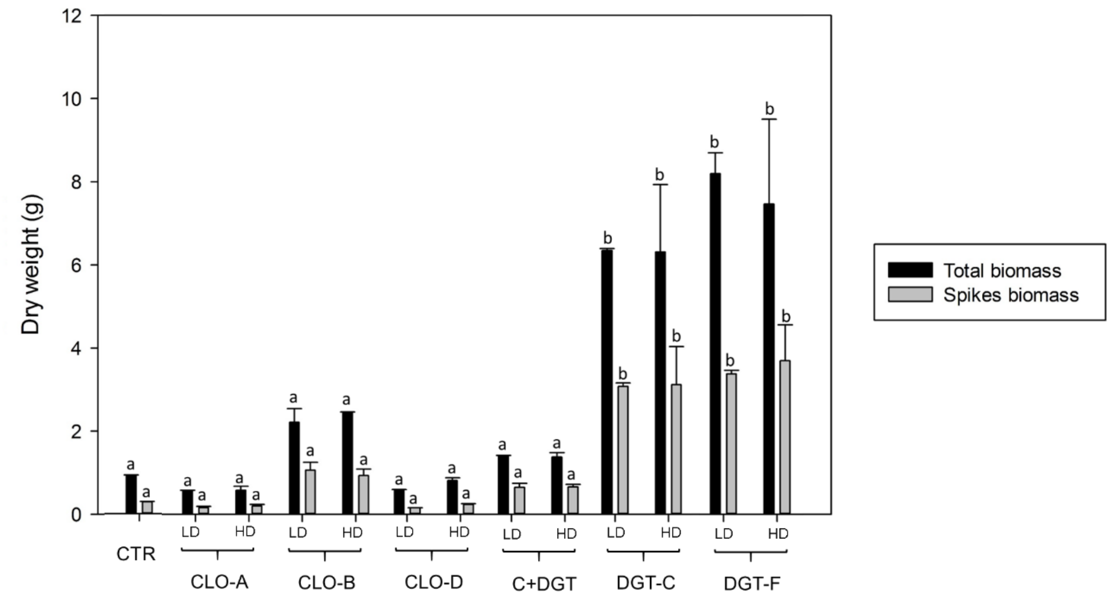

Figure 6.5. Total biomass and spike biomass (dry weight) of wheat plants growth in CLO/DGT amended soil at 90 days of sowing, depending on the applied treatment. The treatment codes indicate the waste tested (control:CTR; compost-like-output:CLO; digestate:DGT; compost-like-output from digestate:C+DGT) and dose (HD: 80 g·kg^{-1}; LD: 20 g·kg^{-1}). The error bars correspond to standard error. The letters indicate significant differences between treatments according to Fisher's test ($p < 0.05$). n=39.

Soil collembolan (*F. candida*) survival in DGT/CLO-soil mixtures after 28 days showed a clear effect of CLO-B mixtures when applied at high doses (see Figure 6). While DGT low dosages were neutral or led to survival promotion over controls, differences with CLO-A (LD), C+DGT (LD) and CLO-D were not statistically significant. These results were consistent with those obtained when *F. candida* reproduction was assessed (see Figure 6). All of the CLO treatmentsexhibited very strong reproduction inhibition at the high dosage. In CLO-B, this inhibition was due to the mortality of adult individuals, while in CLO-A and CLO-D, no significant mortality was detected. Inhibition of reproduction was also explained by chronic toxicity of decomposition products or other unassessed substances in these residues (Domene *et al.*, 2007), or by salinity, slightly coupled to these reproductive effects (Table S11). The correlation with the stability degree of the amendments becomes more evident looking at the results in the C+DGT treatment, the most stable material included in this study, where no inhibition on reproduction was observed.

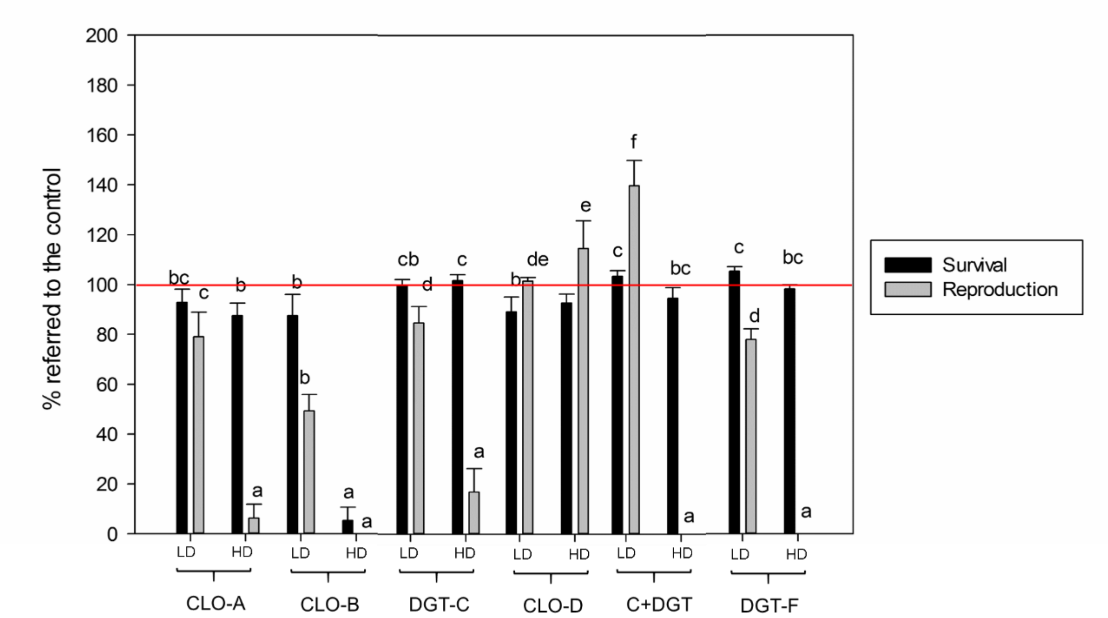

Figure 6. Survival and reproduction rates of *Folsomia candida* in CLO/DGT amended soil, relative to the control, depending on the applied treatment. The treatment codes indicate the waste tested (CLO, DGT, C+DGT) and dose (HD: 80 g·kg^{-1}; LD: 20 g·kg^{-1}). The error bars correspond to standard error. The letters indicate a significant difference according to Fisher's test (p <0.05). n=65.

Upscaling to the field: CLO and DGT applications for soil restoration purposes

One month after the setup of the plots, all treatments showed higher herbaceous cover than controls, except for some treatments in the road slopes of Olost (CLO-A_{OL} and CLO-B_{OL}). Moreover, CLO-A_{OL} and CLO-B_{OL} presented vegetation covers below controls of Terrassa road slope and Lloret de Mar landfill (CTR_{TE} and CTR_{LM}, respectively) which are much more unfertile than CTR_{OL}. Four months after the spreading of the technosols on the slopes, plots with an application rate of 20 g·kg^{-1} of DGT showed almost two-times more vegetation cover than controls (Figure 7). This difference was even bigger when control soils were extremely poor in organic matter and nutrients, such as those in Terrasa. Regarding the CLO treatments, their effects were dependent on the quality of control soil: in relatively fertile ones like those of Olost, all the CLO treatments showed a lower plant cover development than controls, especially CLO-A_{OL} and CLO-D_{OL}, which after four months resulted in plots without vegetation. In contrast, in very poor soils, like those in Lloret de Mar (CTR_{LM}) the CLO application promoted vegetation development.

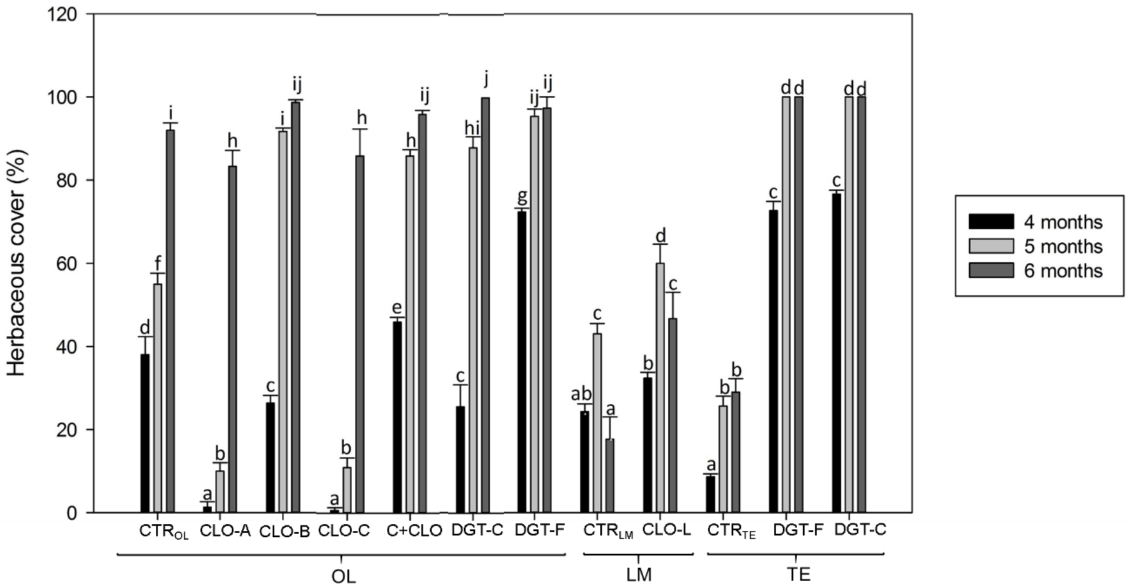

Figure 7. Herbaceous cover on plots restored with soil amended with CLO/DGT, 4, 5 and 6 months after Technosol spreading. Treatment codes indicate the waste tested (control:CTR; compost-like-output:CLO; digestate:DGT; compost-like-output from digestate:C+DGT) and the site (OL: Olost; Te: Terrassa: LM: Lloret de Mar). The error bars correspond to standard error. The letters indicate a significant difference according to Fisher's test ($p < 0.05$) between measures of the same site. n=36.

Regarding differences between doses, treatment with 40 $g \cdot kg^{-1}$ of digestate ($DGT\text{-}C_{OL}$) presented less vegetation cover than treatment with 20 $g \cdot kg^{-1}$ ($C+DGT_{OL}$) on the first sampling. Regarding EC, this increased after applying the amendment and diminished after five months (see Tables S12 and S13).

Looking at plant biodiversity, during the first months after soil restoration (Tables S14, S16 and S17), sown species dominated in all the plots, controls and amendments. Biodiversity was relatively low in all the sites and treatments, with the plots in Terrassa being the richest, with more than 50 species identified, mainly from Asteraceae, Fabaceae and Poaceae families. In control plots a relative dominance of leguminous was observed, mainly in very poor soils, such as those in Terrassa and Lloret de Mar. In DGT and CLO amended plots, a dominance of seeded grasses was observed during the first months after soil restoration (Tables S14, S16 and S17), although at the mid-term (ten months), ruderal vegetation increased, being dominant in some plots (Table S15 and S17). However, ruderal plants differed between treatments: in DGT plots, species typical from nitrogen rich environments (*Chenopodium sp., Cardus sp.*) were observed.

Discussion

Results relating to CLO and DGT characterization showed a high variability between products of different plants, and batches of the same plant. However, it is possible to clearly distinguish between DGT and CLO amendments, which have very different characteristics. Regarding DGT, a general pattern of relatively high moisture, low EC and homogeneous particle size distribution has been detected. Despite low EC and homogeneous particle size distribution being positive factors for the use of DGT, high moisture content (>80%) could cause technical problems such as difficulties for transporting and handling, and the production of leachates (Alcañiz *et al.* 2009).

Regarding CLO, impurities content seemed to be the main restricting factor for their application at soil. It is worth noticing that small impurities (below 2 mm) could be underestimated by the available current methodologies, so the total proportion of impurities could be even greater. Regarding the type of impurities, glass and plastics were those found in a greater proportion (in weight), and moreover, plastics represented a significant volume. These are relevant features in terms of public acceptance as those materials cause strong visual impacts, especially for glass particles due to its ability to reflect light. However, the most important issue is of the currently known negative (direct and indirect) effects of plastics (including microplastics) on trophic networks, and more specifically on soil organisms (Huerta Lwanga *et al.*, 2016; Zhu *et al.*, 2018). As expected, the CLO had higher concentrations of heavy metal content than the compost obtained from source-separated bio-waste in Spain (ESAB, 2005) and in European countries (Saveyn & Eder, 2014), however the analyzed CLO would still be able to be applied to agricultural soils according to Spanish law (BOE, 2013).

Despite EC values being higher than DGT, in general EC was not a critical factor for the use of CLO as an organic amendment, except for CLO-D, which had a conductivity above 9 $dS \cdot m^{-1}$. One hand, a germination stimulation was not observed in either treatment, despite some stimulation having been described previously after the application of mature compost to soil (Abdullahi *et al.*, 2008). On the other hand, an inhibition effect was observed for some CLO, but because all the treatments had a germination rate higher than 80% (compared to the control) any phytotoxic effect was discarded (McLachlan *et al.*, 2004). In any case, the use of CLO with conductivity higher than 9 $dS \cdot m^{-1}$ could cause germination inhibition in sensitive plants at high to medium application rates such as those used for Technosol construction in road slopes or other soil rehabilitation efforts (do Carmo *et al.*, 2016; Evenari, 1949). Moreover, after applying the amendments to soil, EC could increase due to mineralization of the organic matter, despite the fact that in the greenhouse experiment, soils treated with CLO showed an EC reduction. This is due to both an immobilization effect of soluble elements on minerals or on microbial biomass, and through losses by leaching or plant uptake (Raviv *et al.*, 2019). High sodium and chloride concentrations could be explained by the domestic origin of the organic wastes (Domene *et al.*, 2007).

The low soluble element contents in the CLO treatments can be explained by the immobilization effect, where nitrogen is immobilized as microbial biomass (Bastida *et al.*, 2008; Davidson *et al.*, 2013). The application of low stability wastes to soil increases microbial activity in the short-term (Bastida *et al.*, 2008; Meena *et al.*, 2016), but leads to sequestering (immobilizing) of nutrients, since the amount of plant-available N from municipal solid waste is closely related to the degree of compost maturity (Crecchio *et al.*, 2001) and stability (Davidson *et al.*, 2013; Tarrasón *et al.*, 2008). This trend was stronger at higher dosages of CLO that showed lower plant growth than at the lower dosages. This effect also explains the chlorosis and the poor plant development observed in some CLO treatments in the greenhouse experiment that increased at high dosages since more labile carbon was present and promoted this unintended effect on growth. This effect also affected spike biomass, which was taken as a proxy of the reproductive effort or seed output (Bazzaz *et al.*, 1992; Reekie and Bazzaz, 2002), but is also a key ecological parameter directly linked to seed bank establishment capacity in natural environments and indicative of the potential recovery of the herbaceous cover after a disturbance (Reekie and Bazzaz, 2005, 2002). Moreover, toxicity against *F. candida* observed in CLO-B mixtures when applied at high doses could be due to the release of decomposition products such as ammonium, phenols or organic acids (Domene *et al.*, 2007), or other toxic compounds (Andrés and Domene, 2005; Crouau *et al.*, 2002) inherited from the residue. Since the CLO-B treatment also showed the highest salinity contents in soil mixtures (see section 3.2), this parameter could also partially explain the negative effect observed (Domene *et al.*, 2007). However, in any case salinity was not the main reason because other treatments that were similar in terms of salinity did not show toxicity.

In the controls, soil chlorosis could be explained by the low concentration of plant-available N in the B horizon material. In C+DGT treatments the low mineral-N concentration was due to the double-stabilization process (anaerobic digestion and composting) carried out for this material, since anaerobic digestion mineralizes the more labile organic matter and releases ammonium-N, which can then be easily lost as ammonia in the composting phase (Himanen and Hänninen, 2011). In contrast, in the treatments with DGT (both high and low doses), the high concentration of nutrients (nitrates and phosphates, see section 3.2) boosted plant development. Biomass production did not differ significantly between the dosage range studied (20-80 $g \cdot kg^{-1}$), indicating that low dosages are also within the plant requirements and that higher rates might be unnecessary. Therefore, high dosages should be avoided in order to prevent the negative environmental effects of leaching of soluble elements provided by the organic amendments but not taken by plants (Logan and Visvanathan, 2019).

Despite all materials having organic matter with a high stability degree (>40%), as determined by acid hydrolysis, measuring stability with a single parameter gives limited information (Morel *et al.* 1985; Reinikainen and Herranen, 2001; Komilis and Tziouvaras, 2009) since different factors could affect this measure. The stability degree values were correlated with C:N ratio, with DGT-C being the one with the lowest value (<12). This might have modified the soil microbiological equilibrium, as C:N was below the 12-15 range (Bernal *et al.*, 1998), favoring N losses. However, some authors have criticized the use of the C:N ratio as an indicator parameter for compost stability and maturity (Wu *et al.*, 2010; Tiquia *et al.*, 2000). Maturity measured through the Rottegrade test is highly influenced by the sample heterogeneity

(Weppen, 2002) that may have been very high in some CLO samples in terms of composition and particle size distribution. Moreover, as stated by Huerta *et al.* (2010), the Rottegrade test is not a suitable method for evaluating maturity of DGT, and a combination of both stability degree and biological stability could give an idea about the potential for DGT to act as organic amendment (Tambone *et al.*, 2009). Regarding ammonia-N content, no correlation was observed between temperature (Rottegrade test), and neither with stability degree, despite ammonia-N content also being considered as a good indicator of low maturity of compost (Bernal *et al.*, 1998; Clemente *et al.*, 2006; Tiquia *et al.*, 2000).

Despite differences, both products were dominant in the fraction below 2 mm, which is of interest as it is this fraction that is mostly responsible for the organo-mineral interactions in soil, and consequently for the aggregate formation and preservation of organic carbon (Jones and Singh, 2015), a key issue in the improvement of soil fertility globally. Moreover, a relatively fast and direct effect of the application of organic amendments to soil was the improvement of structure, which enhanced the water retention capacity (Ojeda *et al.*, 2011, 2010). This effect could be observed in the highest application dosages, either in CLO and DGT, but not in low dosages, probably due to the clay loam texture of the experimental soil used. At the same time, DGT seemed to have a better capacity to improve soil structure, due to the deep effect on soil structure conferred by digestates (Ojeda *et al.*, 2011, 2010). Moreover, the slight acidification observed in both treatments could be considered positive, though soil remained basic due to its calcareous nature.

Differences between CLO and DGT observed at laboratory and greenhouse scales were also observed at the field plots and were even stronger when comparing the effects of those amendments in poor soils (CTR_{LM} and CTR_{TE}). Soils amended with CLO presented lower vegetation than soils amended with DGT. According to the results at greenhouse scale, these differences could be mainly explained by the high N content of DGT and the lower stability of CLO. Regarding biodiversity, despite the dominance of ruderal and sown species in amended plots, recruitment of weeds and shrubs was observed in all the sites, for which we should think, in accordance with other studies that the use of these organic amendments at moderate doses would not negatively affect plant recruitment and diversity in the mid-term (Carabassa *et al.*, 2018).

Conclusions

Low stability degree and high impurities content are the most restricting parameters for applying CLO to soils. Furthermore, there is a considerable heterogeneity of CLO produced by different plants and also between different batches of the same plant, in relation to the main physicochemical parameters considered, which is a constraint generalizing about CLO use. In contrast, DGT composition is more stable between plants and batches. DGT presents low impurities and high N content that makes it more suitable for applying to soil and promoting plant development and growth. Regarding dosage, doses higher than 20 $g \cdot kg^{-1}$ do not improve vegetation development and growth, neither in CLO or DGT, and increase environmental problems associated with the use of organic wastes, such as water pollution and toxicity to soil fauna.

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Supplementary materials

Table S1. Physicochemical and microbial characterization of selected CLO and DGT.

MBTP plant code	A	B	C	D	E	F
Material type	CLO	CLO	DGT	CLO	C+DGT	DGT
Dry matter (% fresh matter at 105 ºC)	66	70	25,4	65	68,1	32,3
pH	8,16	7,21	8,35	8	8,06	8,77
Electrical conductivity 25ºC ext 1:5 (dS/m)	1,66	6,75	3,88	9,2	6,85	3,26
Amonia-N (% dry matter)	0,29	0,25	1,29	0,48	0,39	0,95
Kjeldhal-N (% dry matter)	1,8	2,17	3,1	1,4	2,09	3,02
Organic matter (% dry matter)	64	45,5	64,6	44	55,7	50,7
Stability degree (% of OM)	52,8	46,5	44,5	47,8	60,1	57,4
C/N ratio	18	13,4	10,4	13	13,3	14,2
Rottegrade class	II	V	-	IV	V	-
P (% sms)	0,3	1,22	0,87	0,42	0,611	1,2
K (% sms)	0,59	1,07	0,77	0,63	1,26	0,78
Ca (% sms)	3,8	9,2	6,66	5,1	8,9	8,7
Mg (% sms)	0,49	1,05	0,825	0,561	0,795	1,43
Fe (% sms)	0,43	0,77	2,33	0,56	0,89	0,95
Cd(mg/kg)	1,1	1,13	1,98	1,9	0,94	1,05
Cu (mg/kg)	171	259	129	106	219	156
Ni (mg/kg)	35	23,7	33,4	182	19,2	27
Pb (mg/kg)	82	71,6	87,6	57	76,8	107,3
Zn (mg/kg)	302	426	372	190	415	463
Hg (mg/kg)	2	0,42	1,92	0,27	<0,40	1,03
Cr (mg/kg)	67	30,7	79	461	28,8	62
Salmonella (in 25g)	0	0	0	0	0	0

Table S2. MBTPs analytical information from representative samples of CLO and DGT obtained in different seasons and years. Excel file available in: https://ars.els-cdn.com/content/image/1-s2.0-S0301479719316275-mmc2.xlsx

Table S3. Organic waste types and doses tested on pilots trials (CLO= compost-like-output, DGT= digestate, C+DGT: compost-like-output from digestate). Olost road slope (OL), Terrassa road slope (TE) and Lloret de Mar landfill (LM). Available in: https://ars.els-cdn.com/content/image/1-s2.0-S0301479719316275-mmc1.docx

Table S4. Composition and proportion of the seed mixtures sowed on the pilot tests. Olost road slope (OL), Terrassa road slope (TE) and Lloret de Mar landfill (LM). Available in: https://ars.els-cdn.com/content/image/1-s2.0-S0301479719316275-mmc1.docx

Table S5. Particle–size distribution (% dry matter) of the CLO and C+CLO included in this study. Available in: https://ars.els-cdn.com/content/image/1-s2.0-S0301479719316275-mmc1.docx

Table S6. Impurities >2 mm (% dry matter) in the compost-like-outputs and digestates included on this study (A, B, D:CLO; C, F:DGT; E:C+CLO).

MBTP Plant code	A	B	C	D	E	F
Stones (%)	0,0	0,0	0,0	0,5	0,0	0,0
Glass (%)	0,5	0,3	0,0	3,0	0,4	0,0
Plastics (%)	1,1	0,2	0,4	0,8	0,9	0,3
Metals (%)	0,4	0,2	0,0	0,0	0,1	0,0
Total (%)	2,1	0,7	0,4	4,3	1,4	0,3

Table S7. Heavy metal contents of different compost-like-outputs (CLO), digestates (DGT) and compost-like-outputs from digestates (C+DGT) compared to those compost segregated at source produced in Spain (ESAB, 2005), the maximum values for EU EoW product quality criteria (IPTS, 2008) and for agricultural uses in Spain (BOE, 2013). *Source-Segregated.

MBTP plant code	Material type	Cd (mg/kg)	Cu (mg/kg)	Ni (mg/kg)	Pb (mg/kg)	Zn (mg/kg)	Hg (mg/kg)	Cr (mg/kg)
A	CLO	1,1	171	35	82	302	2	67
B	CLO	1,13	259	23,7	71,6	426	0,42	31
C	DGT	1,98	129	33,4	87,6	372	1,92	79
D	CLO	1,9	106	182	57	190	0,27	461
E	C+DGT	0,94	219	19,2	76,8	415	<0,4	29
F	DGT	1,05	156	27	107,3	463	1,03	62
ESAB, 2005	Spanish SS* compost	0,35	103	36	54	215	0,88	32
EU EoW quality criteria (IPTS, 2008)	Compost and digestate	1,5	200	50	125	600	1	100
Spanish límits for agricultural uses (BOE, 2013)	Compost	3	400	100	200	1.000	2,5	300

Table S8. Water holding capacity (WHC), increment of WHC relative to the control soil (ΔWHC), pH and EC evolution of the soil amended with two doses of DGT/CLO (CLO= compost-like-output, DGT= digestate, C+DGT: compost-like-output from digestate). EC_0= electrical conductivity just after applying the DGT/CLO; EC_3= electrical conductivity three months after applying the DGT/CLO. Available in: https://ars.els-cdn.com/content/image/1-s2.0-S0301479719316275-mmc1.docx

Table S9. Soluble elements concentration in a soil amended with compost-like-outputs and digestates just after mixing (0) and after three months of incubation in greenhouse conditions (3). Units: $g \cdot kg^{-1}$ (dose); $mg \cdot kg^{-1}$ (soluble elements). (A, B, D:CLO; C, F:DGT; E:C+CLO). Available in: https://ars.els-cdn.com/content/image/1-s2.0-S0301479719316275-mmc1.docx

Table S10. Respiration rate evolution (mg C-CO_2·day^{-1}) during the 12 weeks of incubation, according to the treatment. (A, B, D:CLO; C, F:DGT; E:C+CLO; HD, 80 g·kg^{-1}; LD, 20 g·kg^{-1}. Available in: https://ars.els-cdn.com/content/image/1-s2.0-S0301479719316275-mmc1.docx

Table S11. Correlation matrix for chemical and ecotoxicological parameters of CLO/DGT mixtures. Available in: https://ars.els-cdn.com/content/image/1-s2.0-S0301479719316275-mmc1.docx

Table S12. Anions content in soils amended with DGT/CLO in Olost (OL), just after amendment application (T0) and 4 months after (T1). Treatment codes indicate the waste tested (A, B, D:CLO; C, F:DGT; E:C+CLO) and dose (HD: 80 g·kg^{-1}; LD: 20 g·kg-1). Results are expressed in mg/kg, related to soil fine fraction (<2 mm). Available in: https://ars.els-cdn.com/content/image/1-s2.0-S0301479719316275-mmc1.docx

Table S13. Cations content in soils amended with DGT/CLO in Olost (OL), just after amendment application (T0) and 4 months after (T1). Treatment codes indicate the waste tested (A, B, D:CLO; C, F:DGT; E:C+CLO) and dose (HD: 80 g·kg^{-1}; LD: 20 g·kg^{-1}). Results are expressed in mg/kg, related to soil fine fraction (<2 mm). Available in: https://ars.els-cdn.com/content/image/1-s2.0-S0301479719316275-mmc1.docx

Table S14. Floristic inventory of OL plots during the first summer (six months after soil restoration) and abundance estimation (1=testimonial, 2=present, 3=abundant, 4=very abundant, 5=dominant). Available in: https://ars.els-cdn.com/content/image/1-s2.0-S0301479719316275-mmc1.docx

Table S15. Floristic inventory of OL plots during the first autumn (ten months after soil restoration) and abundance estimation (1=testimonial, 2=present, 3=abundant, 4=very abundant, 5=dominant). Available in: https://ars.els-cdn.com/content/image/1-s2.0-S0301479719316275-mmc1.docx

Table S16. Floristic inventory of TE plots during the first summer (six months after soil restoration) and abundance estimation (1=testimonial, 2=present, 3=abundant, 4=very abundant, 5=dominant). Available in: https://ars.els-cdn.com/content/image/1-s2.0-S0301479719316275-mmc1.docx

Table S17. Floristic inventory of dominant plants on LM plots during the first summer (six months after soil restoration) and autumn (ten months after soil restoration), and abundance estimation (1=testimonial, 2=present, 3=abundant, 4=very abundant, 5=dominant). Available in: https://ars.els-cdn.com/content/image/1-s2.0-S0301479719316275-mmc1.docx

7. RESTOQUARRY: indicators for self-evaluation of ecological restoration in open-pit mines

Des de diferents sectors vinculats a la restauració d'espais degradats s'ha demanat, i es continua demanant, un major seguiment dels projectes de restauració, així com l'establiment de criteris d'avaluació objectius, que siguin clars, generalitzables, amb base científica i aplicables a gran escala. Aquest capítol presenta un protocol dissenyat per avaluar la restauració d'activitats extractives mitjançant indicadors de fàcil mesura, que pot ser utilitzat pels propis treballadors de les activitats extractives i pels tècnics de l'Administració. L'aplicació del protocol a una mostra representativa d'activitats extractives de Catalunya ha permés implicar en major mesura el personal de les empreses en aquest seguiment, demostrant a la vegada que la majoria de les restauracions presenten punts febles que cal millorar, si bé en general podem dir que les restauracions efectuades progressen adequadament. Aquest capítol s'ha publicat a la revista Ecological Indicators (Carabassa, V., Ortiz, O., Alcañiz, J.M. (2019) RESTOQUARRY: Indicators for self-evaluation of ecological restoration in open-pit mines. Ecol. Indic. https://doi.org/10.1016/j.ecolind.2019.03.001).

Abstract

Several methods and criteria to evaluate and assess quarry restoration are available in the scientific literature, but they are very specialized and time consuming. Furthermore, there is a lack of evaluation tools appropriate for technicians involved in these types of activities, such as quarry engineers, restoration managers and quality control supervisors in public administration. The work presented attempts to bridge the gap between scientific knowledge and practical needs by proposing a simplified methodology (RESTOQUARRY protocol), which enables the non-scientific public to evaluate restored areas. This procedure focused on five groups of parameters for zone (homogeneous portions within the whole restored area) evaluation: geotechnical risk, drainage network, erosion and physical degradation, soil quality and vegetation status and functionality. Moreover, three groups of parameters are proposed for area (whole restoration) evaluation: landscape integration, ecological connectivity and fauna, and anthropic impacts. This protocol has been tested in 55 open-pit mines located throughout Catalonia (NE Iberian Peninsula), covering a wide range of Mediterranean climatic conditions and geological substrates. Results indicate that the proposed methodology is suitable for detecting critical parameters that can determine the success of the restoration.

Keywords: Open-pit mines reclamation; quarry rehabilitation; ecological restoration; restoration evaluation; ecological indicators

Highlights

- A new multicriterial procedure for integrated self-evaluation of mine restorations
- It includes ecological, technical and socio-cultural aspects
- It uses 34 evaluation parameters, selected and weighted by an expert panel
- The evaluation allows to score the whole restoration
- The score is accompanied by an interpretation of the monitoring values
- The evaluation allows to highlight critical factors for restoration success

Introduction

Ecological restoration is defined by The International Society for Ecological Restoration as the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed (Clewell *et al.*, 2004), in order to retrieve its environmental functions and ecosystem services. This institution provides a list of ecosystem attributes as a guideline for measuring restoration success after human-induced perturbations. However, what characterizes successful restoration and how best to measure it generates debate among members within the scientific community (Wortley *et al.*, 2013; Crouzeilles *et al.* 2016). Many methods to evaluate these attributes are available in the scientific literature and most studies are focused on vegetation composition and structure, biodiversity and ecological processes (Ruiz-Jaen & Aide, 2005; Wortley *et al.* 2013). In the present paper, the concept of restoration is used in a broad sense, including rehabilitation and other recovering alternatives of mined sites.

It is well known that advances in restoration ecology are intrinsically linked to advances in the ecological understanding of the ecosystems to be restored, and the knowledge of soil and vegetation properties is an appropriated way to guarantee restoration success (Prach, 2003; Temperton *et al.* 2004; Valladares and Gianoli, 2007). Moreover, geotechnical stability, runoff control, landscape integration, and ecological connectivity, among others, are basic site attributes to be considered for a good quality restoration, especially in mining activities. However, the choice of relevant evaluation attributes depends on the type of degradation processes that previously affected the restored zones. Specifically, sites affected by mining activities, such as quarries, are a paradigmatic case of drastic anthropic perturbation, as almost all the components and attributes of the original ecosystem have been destroyed and, therefore, must be restored.

Practitioners have asked researchers to provide potentially useful procedures based on objective indicators (Clewel & Rieger, 1997; Beier *et al.* 2017). On the other hand, researchers have appointed the need to improve the evaluation of restorations carried out in open-pit mines (Halldórsson *et al.* 2012, Hagen *et al.* 2013; Suding 2011), although the available information on the topic has increased in the last years (Wortley *et al.* 2013). Evaluation tends to be focused on the descriptive characterization of the restored areas, and restricted to a single or few checks after the restoration works (Suding 2011).

Nonetheless, a continuous monitoring during all the restoration process is necessary (Allen *et al.* 2002; Pander and Geist 2013) and should be coupled to the exploitation works. In any case, economic and ecological results of the restoration could be improved if clearer evaluation protocols exist, which also could facilitate the transfer of valuable information to other projects (Nilsson *et al.* 2015).

The present work attempts to satisfy these demands for evaluating restoration of mine sites, providing a scientifically based multifactorial methodology to be incorporated in the decision-making process. This will lead to regaining the restoration bonds (financial guarantee) that mine companies must deposit in many countries, in order to guarantee the correct restoration of the degraded land. This study aims to contribute to the generation of best available techniques in this field, filling the gap that already exists in the extractive activities sector with an innovative methodology that takes into account a wide range of geotechnical and ecological indicators. Some authors have proposed similar procedures for rangelands and mine sites (Courtney *et al.* 2010; Dzwonko and Loster 2007; Tongway and Hindley, 2004); however, these methodologies are rather inaccessible to the non-scientific public, as they assess excessively specific or technical indicators. In order to avoid these limitations, RESTOQUARRY protocol, a self-evaluation procedure of open-pit mines restoration, is proposed (Carabassa *et al.* 2010; Carabassa *et al.* 2015). This protocol is aimed to be useful for mining engineers and managers of environmental agencies, who can easily put it into practice without having to have much scientific knowledge about ecological restoration. If this goal is reached, better involvement by extractive companies in the restoration process would also be achieved and, therefore, the quality of the restorations carried out by these industries would rise. Moreover, the application of participatory methodologies such as the proposed in this work would aid the cooperation and communication between public administration and extractive industries, which is crucial for improving restoration and finding the most appropriate solution on a case by case basis.

Materials and methods

Selection of restoration indicators

A preliminary proposal of quality indicators/parameters of mining restoration success was subjected to a screening process by experts. This proposal has been based on the know-how generated in previous research projects and carried outwith the collaboration of engineers of mining industries, technicians of competent authorities, ecologists from NGOs, technicians from consulting companies and scientists with broad experience in mine restoration. These actors constituted an expert panel including 17 people/entities. After an independent review process, the first proposal of indicators was made. This proposal included specific indicators applicable to homogeneous zones within the whole area (zone indicators), and a set of more generalist indicators, applicable to the whole restored area (area indicators). This distinction between *area* and *zone* was made in order to correctly evaluate parameters that must be measured separately at slope, habitat or landscape level.

There are five groups of zone indicators: geotechnical risk, drainage network, erosion/degradation processes, soil and vegetation (Table 7.1). Some vegetation indicators (plant cover, woody species richness and density, or herbaceous species richness) are based on the comparison to a reference site, usually located in an undisturbed zone close to the mine. For geotechnical risk (area affected by landslides and fallen blocks) and erosion (area affected by rill erosion) indicators, the area influenced by instability processes could measured directly at the field or by photointerpretation, depending on the magnitude of the process. Soil bulk density is measured by the excavation method as coarse particles are often abundant in this kind of substrates. Soil sampling is performed using Edelman auger or similar tool to extract the first 20 cm of topsoil. The recommended sampling density is specified in the protocol (20 holes/ha). Vegetation measures are obtained on 10x10 m square plots, distributed along the evaluated zones, and on 10 m transects delimited by the sides of these plots (horizontal and perpendicular to the slope).

Indicators related to the area evaluation are mainly qualitative (see Table 7.2). This is especially true for the case of landscape integration, where the proposed indicators are based mainly on the perception of the evaluator. However, the protocol gives guidance in order to reduce the subjectivity of the observations, allowing the evaluator to classify landscape integration according to the similarity of the restored area to the surrounding natural landscape. All the methods for measuring the indicators are standardized and explained in Carabassa *et al.* (2015), including sampling density and recommendable sampling period.

Table 7.1. Pre-selection of restoration quality **zone indicators** included in the preliminary proposal of evaluation protocol. Zones are described in this work as homogeneous portions of the whole restored area.

Geotechnical risk	**Erosion/physical degradation**	**Drainage network**	**Soil quality**	**Vegetation status and functionality**
Maximum diameter of fallen blocks (m)	Area affected by rill erosion (% related to the total area)	Drainage channels broken (% of total channels)	Soil depth (m)	Plant cover (%) divided into: herbaceous cover and woody species (shrubs and trees) cover
Area affected by fallen blocks (% of the total area)	Estimated rill erosion rates ($Mg \cdot ha^{-1} \cdot year^{-1}$)	Drainage channels filling-in (% of total channels)	Particles <2 mm ($g \cdot kg^{-1}$)	Area occupied by exotic/invasive species (% of the total area)
Area affected by landslides (% of the total area)	Rain splash protection (% of the surface protected)	Drainage network functionality (% of damaged, stabilized and non-functional channels)	Clay content ($g \cdot kg^{-1}$)	Species with fruits (number of species)
Other signs of instability: cracks, subsidence, deformations, faults, fallen trees (qualitative)	Surface crusts presence (qualitative)		Organic matter ($g \cdot kg^{-1}$)	Mortality of planted woody species (%)
	Sheet erosion (qualitative)		Carbonates ($g \cdot kg^{-1}$)	Woody species richness (% related to richness on reference site)
	Piping or subsurface flows (qualitative)		Electrical conductivity, 1:5 extract, 25°C ($dS \cdot m^{-1}$)	Woody species density (% related to density in reference site, per species)
			Soil pH	Woody species recruited (number)
			Total nitrogen (%)	Herbaceous species richness (% related to richness on reference site)
			Available phosphorous ($mg \cdot kg^{-1}$)	
			Available potassium ($mg \cdot kg^{-1}$)	
			Physical contaminants presence (number of elements observed)	

Table 7.2. Pre-selection of restoration quality **area indicators** included in the preliminary proposal of evaluation protocol.

Landscape integration	**Ecological connectivity and fauna presence**	**Anthropic impacts**
Chromatic and textural integration (qualitative)	Ecological barriers (presence and type)	Uncontrolled vehicle circulation (qualitative)
Geomorphic integration (qualitative)	Woody plants with edible fruits (Species and density)	Waste dumping (type, magnitude and distribution)
Internal road networks (functionality, density and width)	Fauna refuges/supply structures (presence)	Grazing (presence and intensity)
	Fauna observations (number and species)	Abandoned constructions and facilities (presence, magnitude and height)
	Fauna paths (presence)	
	Fauna traces (presence)	
	Nests (presence)	
	Burrows (presence)	

Transformation of indicators to restoration quality indexes

In order to compare and integrate the evaluation data through a set of individual indicators, the use of functional curves is proposed (Figure 7.1). The objective is to obtain a global Restoration Quality Index (RQI) that summarizes the main factors influencing the restoration, using the proximity to target methodology (Rodríguez-Loinaz *et al.* 2015, Roces-Díaz *et al.* 2018). A functional curve for each parameter is proposed, according to previous works and the knowledge and expertise of the panel members (Cortina *et al.*, 2012; Deltoro *et al.*, 2012, Jorba *et al.*, 2010, Carabassa *et al.* 2010; Moreno-de las Heras *et al.*, 2008, Alcañiz *et al.*, 2008; Tongway and Hindley, 2004; Conesa, 2003, Forman, 2003). These functions transform each parameter value, measured in its own units, to its respective Restoration Quality units (RQ_x), which are standardized, dimensionless and fully comparable, where 1 represents the maximum quality for restoration and 0 the worst case.

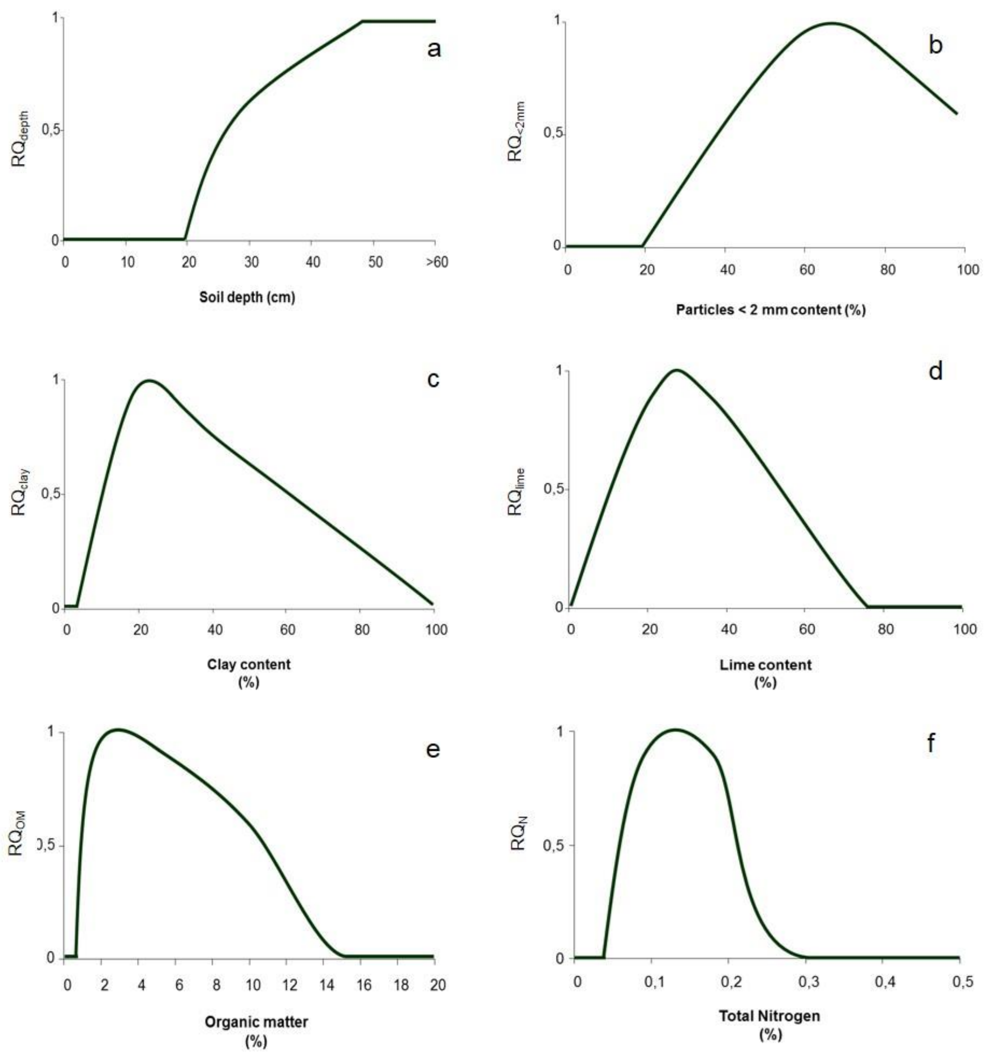

Figure 7.1. Functional curves for some soil parameters: (a) soil depth, (b) particles < 2 mm, (c) clay content, (d) lime content, (e) organic matter, (f) total nitrogen. RQ_x= restoration quality value for the respective parameter.

Indicators weighting

The expert panel was invited to weight the indicators in order of importance for the evaluation of the restoration success. Indicators were weighted using a pairwise comparison method through a Delphi process (Okoli and Pawlowski, 2004; Mukherjee et al 2015). The result of the ranking and pairwise successive comparisons gave a weight (W) for each indicator according to its importance for the whole restoration success. The global restoration quality index (RQI) was calculated as the sum of all the RQ_x multiplied by its respective W:

$$RQI = \sum_{x=m}^{n} (RQ_x \cdot W_x)$$

Study sites

The RESTOQUARRY protocol was assayed in a pilot test on 55 selected open-pit mines distributed along NE Iberian Peninsula (Catalonia), covering different climatic conditions, geological substrates, soil types and extracted materials (Figure 7.2, Table 7.3). A total of 106 restored zones were evaluated in these mines applying the proposed methodology.

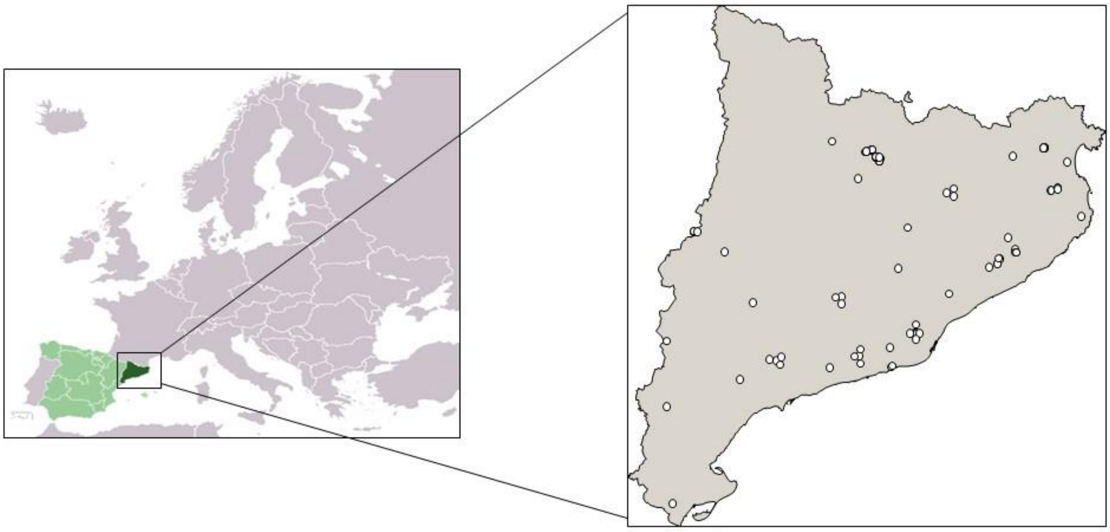

Figure 7.2. Geographical distribution of restored mining activities evaluated applying the RESTOQUARRY methodology, in the NE Iberian Peninsula.

Table 7.3. Geological substrates and ranges of precipitation and air temperature in a representative selection of the extractive activities included in the pilot test (n=55).

Dominant lithology (n=number of activities included)	Dominant mineralogy	Precipitation rank (mm/year)	Mean annual air temperature rank (°C)
Limestone (24)	Carbonatic	526-747	14.1-16.1
Gravel (9)	Mixed	416-799	13.1-15.2
Lignite (6)	Carbonatic	408-888	10.6-15.8
Sand and clay (6)	Siliceous and carbonatic	506-795	14.8-15.6
Evaporites (4)	Gypsic, saline and carbonatic	585-793	13.2-14.6
Basalt (2)	Siliceous	685-745	15.8-16.2
Weathered granite (2)	Siliceous	653-753	15.1-16.3
Granite (2)	Siliceous	599-613	13.8-15.3

The selected restored mine-zones of the pilot test included a broad range of restoration goals, landscape type and age. The main restoration goal in this selection was the ecological restoration, but also there were cases of conversion to agriculture or forestry plantations. The surface of the evaluated areas ranged between 0.8 and 165 ha. The trial areas had been restored between 4 and 21 years before the evaluation process, which allowed the comparison of old restorations with new ones.

Results

Zone evaluation

<u>Geotechnical risks</u>

Flat zones and steep slopes (30-37°) were the predominant geomorphologies in the selected restorations. The slope is an important factor that determines geotechnical risks, soil degradation processes, and vegetation establishment. In terms of geotechnical risk, fallen blocks were observed in 60% of the slopes. Fallen blocks represented big stones or boulders (> 20 cm diameter) that had fallen down from extremely steeped slopes (>45°) and/or vertical walls, representing a safety risk and compromising the vegetation located on the trajectory of this fall. Landslides are also related to slope, and a third of the slopes showed this type of geotechnical risk. Moreover, other geotechnical risks, such as subsidences or cracks were also detected, but they affected minor surfaces and in low grade.

<u>Erosion and physical degradation</u>

Regarding soil degradation processes observed, rill erosion was the most relevant. Rill erosion is a concentrated water erosion process that supposes an important soil loss and that could trigger the destabilization of the entire slope. Approximately half of the areas with slopes of more than 30° showed rills with a depth greater than 5 cm. Areas degraded by concentrated water erosion ranged between 1,053 to 40,700 m^2, which represents 4 to 100% of the surface of the restored zones. The calculated erosion rates ranged between 0.2 to 27 Mg ha^{-1} y^{-1} in the affected zones. The slope is also an important factor for sheet water flow as 61% of the zones with a slope greater than 30° were degraded by sheet erosion. Moreover, a quarter of the evaluated zones showed surface crusts as a consequence of splash. Soil compaction and subsurface erosion impacted 20% and 9% of the evaluated zones, respectively.

Soil quality

Organic matter content, electrical conductivity, available phosphorous (P), total nitrogen (N) content and soil depth seemed to be the most limiting factors in the evaluated soils (see Table 7.4). Poor organic matter contents (<0.8%) were detected in four of the analyzed soils, mainly in the sandy ones. Moderate to high conductivity was detected in some of the soils, but in most of the cases, this was not attributable to the mining activities. A quarter of the soils evaluated showed a low available P content while 12% of the soils showed high levels due to organic amendments (compost, sewage sludge, or pig slurry). This trend was similar to the observed for total N content. Zones with severe slope (>30°) showed an average soil depth of 0.2 m (due to the difficulty of stabilizing topsoil).

Table 7.4. Results for substrate quality indicators on the evaluated zones.

	Soil depth (m)	Particles <2 mm (%)	Clay content (%)*	EC, 1:5 extract ($dS·m^{-1}$)*	pH*
Average	22	44	24	0.4	8.0
Max.	50	94	50	2.2	8.8
Min.	0	19	6	0.1	6.5
Median	20	42	23	0.2	8.0
Standard deviation	22	19	10	0.5	0.3
	Carbonates (%)*	**Organic matter (%)***	**Total N (%)***	**Available P ($mg·kg^{-1}$)***	**Available K ($mg·kg^{-1}$)***
Average	22	2.6	0.14	33	217
Max.	58	12.4	0.57	199	972
Min.	0	0.2	0.02	2	38
Median	23	1.9	0.10	19	148
Standard deviation	15	2.2	0.11	42	184

*Data refer to <2mm soil fraction.

Vegetation status and functionality

The herbaceous cover was dominant in the evaluated zones with an average value of 55%, while mean total plant cover (including trees and bushes) was 73%. Plant cover is an important factor to prevent soil losses because erosion problems are mainly detected in zones with <40% of soil surface covered by plants. Bushy invasive species, such as *Arundo donax*, were present in 19% of the evaluated zones. However, these species were not extensively distributed and were found in small patches. In 81% of the evaluated zones, native bushy species were identified. Reproductively mature bushes were observed in 54% of the locations, and spontaneous reproduction of these species were observed in 45% of the cases, mainly corresponding to *Santolina chamaecyparissus* and *Dittrichia viscosa*. Regarding tree species, low canopy cover and diversity were observed as only 17% of the zones had more than three tree species. *Pinus halepensis,* which was widely planted for reforestation in the Mediterranean region due to its resistance to drought and soil deficiencies, was the dominant species. The mortality rate of planted trees was high for native *Quercus* species, reaching 100% in some cases. On the other hand, some of the evaluated zones were affected by grazing, which negatively strained vegetation development and soil quality (erosion) in the first steps of restoration.

Area evaluation

Landscape integration

Regarding chromatic and morphologic integration to the surrounding landscape, the majority of the evaluated restorations (93%) present good results. However, in some cases, the presence of artificial morphologies (cliffs in hilly landscapes, isolated tips, or repetitive and linear slope-berm morphology) and the dominance of herbaceous vegetation in a site surrounded by forests make this integration poor (Figure 7.3), at least in the first stages of restoration.

Figure 7.3. Differences in vegetation type between restored zones and surrounding areas (left), and the presence of artificial morphologies, like walls (cliffs) in flat/hilly landscapes (right), that make the integration of the restored areas to the landscape difficult.

Ecological connectivity and fauna presence

The presence of steep slopes or abrupt topographic changes is common on the boundaries of the quarries and could act as an ecological barrier for some animal species. Moreover, in the vast majority of the restored areas, structures for attracting fauna (refuges, drinking troughs or woody plants with edible fruits) are missing. Nevertheless, in most of the evaluated areas diverse fauna traces (mainly wild boar and rabbit traces) were observed. Burrows were observed in approximately one third of the evaluated areas, and nests were only observed in one quarry.

Anthropic impacts

Approximately 1/3 of the areas were affected by anthropogenic impacts of various types. The most common effects were related to dumping, mainly in quarries located near to urban areas, and to the presence of abandoned infrastructures and machinery (i.e. ruins of buildings, sheds, conveyor belts or old bulldozers and dumpers) from the previous mining activity (Figure 7.4).

Figure 7.4. The presence of abandoned facilities and machinery of the former extractive activity has a negative impact on the integration of the restored areas and represents a risk for people.

Indicators weight

As a result of the expert panel weighting process, a ranking of the indicators per group was made (Table 7.5). Zone indicators obtained greater weight than area indicators. Among the zone indicators, geotechnical risk was the most relevant since stability problems of the slopes compromise the success of the restoration. The presence of broken channels in the drainage network, directly related to geotechnical instabilities and erosion problems, was considered the second most important indicator. Geomorphologic integration was rated as the third due to its implications in geotechnical risks and soil degradation.

Table 7.5. Weight of the selected indicators according to their importance for restoration success measurement after pairwise comparison by experts panel members. *key indicators.

Group	Group weight (%)	Indicator	Indicator weight (%)
Geotechnical risk	18.0	Area affected by landslides*	9.9
		Area affected by fallen blocks*	4.7
		Other signs of instability*	3.4
Erosion and physical degradation	15.3	Rain splash protection*	4.5
		Area affected by concentrate erosion*	4.3
		Estimated erosion rates*	3.7
		Other degradation processes*	2.8
Drainage network	15.0	Drainage channels broken*	7.7
		Drainage channels filling*	3.9
		Drainage network functionality*	3.4
Soil quality	14.3	Soil depth*	2.4
		Particles <2 mm content*	2.5
		Texture	1.9
		Organic matter / Nitrogen*	2.4
		Electrical conductivity, 1:5 extract	2.0
		pH / Phosphorous / Potassium	2.0
		Impurities (glass, plastics, metals, etc.)	1.1
Vegetation status and functionality	12.7	Plant cover*	2.9
		Woody species richness*	2.6
		Woody species density	2.0
		Woody species recruitment	1.7
		Area occupied by exotic/invasive species	1.7
		Herbaceous species richness	1.8
Landscape integration	12.0	Chromatic and textural integration*	3.1
		Geomorphologic integration*	7,2
		Road network	1.7
Ecological connectivity and fauna presence	6.4	Ecological barriers*	2.1
		Woody plants with edible fruits	1.3
		Fauna refuges/supply structures	1.1
		Fauna observations	1.9
Anthropic impacts	6.3	Uncontrolled vehicle circulation	1.6
		Waste dumping*	2.4
		Grazing	1.0
		Abandoned constructions and facilities	1.3

According to the criteria of the expert panel and the field observations, evaluation parameters with a weight higher than 2% were considered key indicators for ecological restoration success and must be taken into special consideration when analyzing the results of the evaluations.

Restoration Quality Index (RQI) assessment

Using the results of the quality indicators per zone and area, the whole RQI was calculated. Most of the restorations evaluated had a global RQI >70 since the relatively high number of parameters considered make it difficult to have low RQI values. For this reason, a restoration with low values in a specific key indicator could obtain a relatively high global RQI value. In order to avoid that critical situations hidden by high RQI values and that could threaten the restoration, the adoption of corrective measures is recommended when:

- $RQ_x = 0$ for any indicator
- $RQ_x < 0.5$ for a key indicator

Usually, restorations with an RQI > 85 have partial $RQ_x > 0$ on all key indicators. In these situations it could be considered a good result, meaning that the restoration objective has been achieved. However, the adoption of corrective measures could not be excluded in some cases or may be recommended in order to improve some aspects to better guarantee that the ecosystem transition towards a more mature and resilient state occurs. According to this, we could consider that mining companies can regain the restoration bond when they have obtained an RQI > 85 and an $RQ_x > 0$ for all key indicators, and have adopted the recommended corrective measures. An example of the application of the RESTOQUARRY protocol is shown in Table 7.6. In this case, an RQI of 87 was achieved, but soil depth, woody species richness, chromatic and textural integration, woody plants with fruits, and grazing triggered warning alerts and improvement recommendations were needed. It can be seen that the use of this assessment procedure gives a detailed picture of the restoration status. The general overview of this example of evaluation can be that the restoration goals have been reached, although issues related to plant development should to be improved.

Table 7.6. Example of RQI index calculation for a quarry evaluated using the RESTOQUARRY protocol. Critical indicators warning: RQ_x < 0.5 for key indicators (weight more than 2%) or RQ_x = 0 for any indicator.

Group	Indicator	RQ_x	RQI_x	Critical indicators
Geotechnical risk	Area affected by landslides	1.0	9.9	
	Area affected by fallen blocks	1.0	4.7	
	Other signs of instability	1.0	3.4	
Erosion and physical degradation	Rain splash protection	1.0	4.5	
	Area affected by rill erosion	1.0	4.3	
	Estimated erosion rates	1.0	3.7	
	Other degradation processes	0.9	2.5	
Drainage network	Drainage channels broken	1.0	7.7	
	Drainage channels filling	1.0	3.9	
	Drainage network functionality	1.0	3.4	
Soil quality	Soil depth	0.2	0.5	WARNING
	Particles <2 mm content	1.0	2.5	
	Texture	1.0	1.9	
	Organic matter / Nitrogen	0.6	1.5	
	Electrical conductivity, 1:5 extract	1.0	2.0	
	pH / Phosphorous / Potassium	0.2	0.3	
	Physical pollutants	0.9	1.0	
Vegetation status and functionality	Plant cover	1.0	2.9	
	Woody species richness	0.2	0.5	WARNING
	Woody species density	0.9	1.9	
	Woody species recruitment	1.0	1.8	
	Area occupied by exotic/invasive species	1.0	1.7	
	Herbaceous species richness	1.0	1.7	
Landscape integration	Chromatic and textural integration	0.3	0.8	WARNING
	Geomorphologic integration	1.0	7.2	
	Road network	1.0	1.7	
Ecological connectivity and fauna presence	Ecological barriers	1.0	2.1	
	Woody plants with fruits	0.0	0.0	WARNING
	Fauna refuges/supply structures	1.0	1.1	
	Fauna observations	1.0	1.9	
Anthropic impacts	Uncontrolled vehicle circulation	1.0	1.6	
	Waste dumping	0.5	1.3	
	Grazing	0.0	0.0	WARNING
	Abandoned constructions and facilities	1.0	1.3	
		RQI = 87		Recommendation: bond return dependent on adoption of corrective measures

Discussion

The RESTOQUARRY protocol is a procedure that has been designed to help the evaluation of open mine restorations, using objective information obtained through simplified methodologies available for a non-specialized public. The protocol aims also to directly involve engineers of extractive companies in the design and monitoring process of the restoration of their mines, trying to respond to some demands from practitioners (Clewel & Rieger, 1997; Ockendon *et al.* 2018). Moreover, the RESTOQUARRY protocol provides a decision-making system useful for public administration bodies responsible for monitoring and evaluating mine restorations. This evaluation system is a very committed process, which must guarantee the correct evolution of the restorations towards the desired reference (eco)system, and which must maximize the provision of ecosystem services (Comín *et al.*, 2018). In addition, this evaluation process must ensure that the return of the restoration bonds deposited by extractive companies is decided on an objective and quantifiable basis, and made in the correct time, not unnecessary extending the guarantee time, neither shortening it.

The vast majority of the indicators proposed in the protocol indirectly evaluate (proxies) ecosystem services and/or ecosystem functions, allowing the quantification of some of them. For example, erosion control, soil fertility, nutrient recycling or nutrient retention are evaluated through soil quality, soil erosion, and vegetation indicators. Even the most general ones (area indicators), such as those related to anthropic impacts or landscape integration, could be considered proxies of ecosystem services linked to non-material benefits obtained through experiences (for example, cultural services).

The RESTOQUARRY protocol allows good quality restorations to be distinguished from those that need to take corrective measures (i.e. minor revision) and those that have critical failures that pose a risk to all the restoration efforts made (i.e. major revision). The simplicity of the protocol is not achieved at the expense of reliability or replicability since it is based on a wide literature review and the extensive knowledge of a panel of experts in the related fields (ecologists, quarry engineers, administration representatives). Moreover, this protocol has been tested in a wide representative sample of open-pit mines, with the direct participation of the end-users. One of the essential aspects of the protocol is that it does not evaluate the activities that have been carried out in the restoration, but rather its effective results. After applying the RESTOQUARRY protocol, we able to determine the whole restoration quality and to identify the critical features that need to be improved in the extractive activities assessed. Thereby most of the restorations evaluated in this work need the application of corrective measures in order to achieve the minimum standard quality. The RESTOQUARRY protocol also intends to be useful at the stage of restoration design, as it provides evaluation criteria that will be applied at the end of the restoration works. Engaging mine workers and engineers in the evaluation of restoration helps to improve the restoration works and their implication in the restoration process, which consequently could enhance the quality of the restorations carried out by these companies.

Similarities and differences with other evaluation procedures

Despite there being lots of studies evaluating ecological restorations (Ruiz-Jaen & Aide, 2005; Wortley *et al.* 2013), to our knowledge, there are not simplified methodologies readily available for practitioners, that give information about ecosystem services and assess the decision-making process. Landscape Function Analysis methodology (Tongway and Hindley, 2004) is a methodology that fits with these objectives; however it is impractical for a non-scientific public due to its complexity. Other studies also take a similar approach to RESTOQUARRY (Comín *et al.* 2018; Derhé *et al.* 2016; Lithgow *et al.* 2015; Bulloch *et al.* 2011; Birch *et al.* (2010)), taking into consideration the provision of ecosystem services and/or the ecosystem functions, but at a larger scale, with different target reference sites, and more focused on planning restorations than on evaluating the executed ones. While these other studies are focused on ecosystem services provided by ecological restoration in a general way (Comín *et al.* 2018; Bulloch *et al.* 2011; Birch *et al.* 2010), these works do not address the measure of some field parameters directly linked to the quantification of ecosystem services (i.e. carbon storage, nutrient cycling, water regulation, biomass production), as are made by RESTOQUARRY for evaluating restoration success. On the other hand, only a few studies are focused on the particular issue of the evaluation of mine restorations (Courtney *et al.* 2010; Dzwonko and Loster 2007), and they mainly assess very specific indicators related to soil rehabilitation or vegetation recover. Another differential characteristic of RESTOQUARRY is that includes zone specific indicators (geotechnical risk, drainage network, soil quality and degradation, vegetation structure and diversity) adapted to the specificities of mine restoration, such as the need of constructing a drainage network or creating a new soil layer (technosol).

Links between the current procedure of quarries control and the RESTOQUARRY protocol

Mine restoration evaluation tests should assure the correct restoration of mine sites and the recovery of the financial guarantees posted by mine companies conditioned to obtaining satisfactory results in these tests. This evaluation scheme is adopted in some countries like Canada (Mining Act), USA (Surface Mining Act), or the European Union (Directive 2006/21CE). In Spain, for example, the transposition of the EU Directive 2006/21/CE (RD 975/2009) established the need to monitor restorations works each year, until the end of the guarantee period. According to this law, the monitoring process could be done directly by competent administration officers or by accredited external companies. Currently, since the evaluation protocols, indicators, and reference values are not provided, the assessment result depends on the criteria of the evaluator, which sometimes varies according to its background. In this context, RESTOQUARRY protocol is a more accessible tool for a non-scientific public that could help to objectify and standardize the evaluation process, enhancing its transparency for administration bodies, companies, and citizens.

Methodological limitations

A limitation of the global RQI could be that it is confusing if it is not accompanied by an interpretation of the RQ_x partial values. The fact that a wide range of indicators is considered makes it difficult to obtain low RQI values despite the fact that some RQ_x could be very low or even 0, leading to relatively high global RQI values for restorations even though they may have critical faults. We propose the consideration of key indicators in order to decide the adoption of corrective measures could help to solve this problem. Other methodologies for evaluation (Lithgow *et al.* 2015) have used a similar approximation (hierarchical grouping) to prioritize among indicators, obtaining satisfactory results. However, by using the current criterion for key indicators definition (weight higher than 2%), more than a half of them are considered key indicators, which may be excessive. This criterion could be redefined in order to reduce the number of key indicators; however this will increase the chances that some poor quality restorations pass the assessment.

The RESTOQUARRY protocol has been designed and tested mainly in Mediterranean quarries, therefore its application in other climes or mine types could present mismatches in some indicators and reference values. Moreover, this protocol is not a suitable tool for evaluating very case-specific restorations, targeting singular habitats or species (endangered and/or protected) where an expert knowledge is needed. In these cases, some indicators and reference values included in the RESTOQUARRY could not be appropriate, or, alternative indicators should be evaluated. For the same reason, RESTOQUARRY may not be appropriate for evaluating agricultural restorations, but in these cases, the protocol could be easily adapted to specific goals by changing the set of indicators while preserving the general scheme.

Conclusions

The RESTOQUARRY protocol was designed to help mine companies, competent administration and accredited monitoring consultancies in the process of evaluating ecological restoration of mine sites. It consists of a multifactorial procedure, including selected expert-weighted indicators, that allows its large-scale application in the context of ecological restoration of Mediterranean quarries. The protocol could support mine companies in the decision-making process to select corrective measures for improving and optimizing the restoration process. At the same time, it could be useful for competent administration bodies to approve the return of restoration financial guarantees. In summary, RESTOQUARRY is a tool that can contribute to improve the practice and the monitoring of ecological restoration of mine sites.

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8. Consideracions finals i propostes de futur

Sobre la qualitat del sòl

Encara que sembli obvi, m'agradaria començar aquest darrer apartat de la tesi remarcant la importància del control i seguiment de la qualitat del sòl per tal de poder avaluar correctament les accions de restauració, o de manera inversa, els impactes de les pertorbacions associades a les activitats antròpiques, incloent també l'ús d'esmenes orgàniques o els materials que s'utilitzen en treballs de restauració. Aquesta reflexió és fruit de l'experiència guanyada durant tots aquests anys fent recerca aplicada vinculada directament a projectes de restauració reals, executats per empreses i administracions, en els que moltes vegades s'obvia la importància del sòl en el procés de restauració. Per exemple, pocs mesos abans d'escriure aquestes línies encara em trobo amb projectes de restauració executats, amb despeses importants per part de l'Administració, en els que no s'havia previst cap partida relativa a la caracterització de sòls ni s'ha tingut en compte durant l'execució dels treballs. Penso que abans de fer reflexions més concretes cal insistir en aquesta idea, tot i que pugui semblar molt òbvia i bàsica, ja que malauradament aquesta encara és la realitat de molts projectes de restauració a casa nostra. Per tant, vull insistir en què la caracterització inicial del sòl o dels substrats de restauració, ja siguin minerals o orgànics, és bàsica per a l'execució de qualsevol acció que n'impliqui la seva gestió.

Una altra reflexió prèvia és en relació al propi procés o a les metodologies de caracterització del sòl i dels substrats orgànics i minerals. Estem més acostumats a la caracterització basada en paràmetres químics, que ens donen informació valuosa per a la presa de decisions durant el procés de restauració i en la que es basen la majoria de normatives. No obstant, i especialment quan parlem de contaminació, els indicadors biològics són determinants i molt més informatius que els fisicoquímics, ja que els efectes adversos dels contaminants sobre els organismes del sòl són molt difícils d'estimar si només disposem d'informació sobre els indicadors fisicoquímics. Per això, l'ús d'assaigs ecotoxicològics hauria de ser un requisit per avaluar el risc dels contaminants al sòl ja que les proves ecotoxicològiques proporcionen un vincle entre la caracterització química i els efectes de l'exposició sobre humans i biota. En aquest context, l'estudi de poblacions microbianes en condicions d'estrès químic constitueix un tema important de la investigació en ecotoxicologia. Per aquesta raó, hi ha un interès evident en determinar la relació entre les concentracions de productes químics i els efectes sobre el microbioma. A més, els assaigs ecotoxicològics ens permeten detectar efectes sinèrgics o antagònics entre contaminants, situació freqüent en sòls contaminats per un còctel de productes. Fins i tot, els assaigs ecotoxicològics ens permeten establir les dosis hormètiques que molts residus orgànics aplicats al sòl poden generar, malgrat contenir contaminants.

En aquest sentit, el protocol OCDE-217 és una metodologia vàlida per a la detecció d'efectes adversos (i també beneficiosos) dels tres metalls estudiats, Cr, Zn i Cu, sobre la respiració del sòl induïda per substrat. No obstant, les toxicitats avaluades per la prova de respiració del sòl OCDE-217 van resultar baixes en el sòl sorrenc lleugerament àcid escollit, malgrat la seva baixa capacitat d'adsorció i de precipitació de metalls pesants que, per tant, romanien potencialment biodisponibles. No obstant, els valors EC_{50} per Cr, Zn i Cu obtinguts són comparables als presentats per altres autors que han treballat amb sòls de característiques similars. La magnitud dels efectes inhibidors observats també és coherent amb les referències consultades, essent el Cu el metall més tòxic. Ara bé, les respostes microbianes a la contaminació per metalls produïdes en assajos a curt termini, com és el protocol OCDE-217, poden tenir poca semblança a les observades a llarg termini, i encara menys si les comparem amb les obtingudes en assajos de camp, on entren en joc els mecanismes de selecció i adaptació de determinats grups microbians o (sub)poblacions tolerants o resistents. Malgrat aquesta limitació, els assaigs ecotoxicològics en general, i el protocol OCDE-217 en particular, són una bona eina per a la predicció del risc de contaminació dels sòls.

Sobre les esmenes orgàniques

Em pogut comprovar que l'adició de matèria orgànica fàcilment biodegradable emmascara l'efecte dels metalls pesants sobre la respiració del sòl ja que incrementa la biomassa microbiana, així com l'adsorció i la immobilització dels metalls. Per tant, des d'aquest punt de vista de la contaminació per metalls és favorable. No obstant, l'addició de matèria orgànica fresca també pot tenir efectes negatius des del punt de vista ecotoxicològic, especialment si es fa a dosis excessives. Per exemple, la mineralització de la materia orgánica pot provocar una acidificació del sòl, la qual pot ajudar a mobilitzar els metalls pesants, especialment en sòls àcids, on la utilització d'esmenes calcàries pot ser recomanable. Per contra, en ambients calcícoles aquest problema es minimitza per l'efecte tampó dels carbonats.

S'ha demostrat que l'ús d'esmenes orgàniques permet millorar els resultats de la rehabilitació de sòls degradats, a la vegada que es valoritzen residus orgànics i minerals. No obstant, cal valorar l'aptitud de les esmenes cas a cas, a l'existir una gran diversitat de productes i residus orgànics per a ser utilitzats en aquests treballs. Fins i tot, utilitzant un mateix tipus d'esmenes, com per exemple compost, els resultats poden ser molt diferents en funció del seu origen i composició.

Aquesta tesi pretén ajudar a fer un ús més adequat de les esmenes orgàniques en els treballs de rehabilitació de sòls, especialment en activitats extractives, però també en altres terrenys degradats. En aquest context s'ha de veure la utilització de fangs de depuradora urbana com a esmena orgànica en la preparació de tecnosols, com una via més de valorització d'aquests residus, que pot complementar l'aplicació en sòls agrícoles com a fertilitzants. A més, quan s'usen en la restauració d'activitats extractives, permeten valoritzar també els materials de rebuig miner, com són els materials d'extracció sense ús comercial o que no tenen qualitat suficient per a ser comercialitzats, per exemple fraccions terrígenes, estrats de margues o argiles intercalats a les calcàries, cues de procés (p. e. tot-u) o fangs minerals de rentat d'àrids, entre d'altres. També es poden utilitzar els fangs de depuradora per esmenar terres de decapatge que requereixin d'una activació biològica o que s'hagin de mesclar amb materials de rebuig, o per esmenar terres procedents d'excavacions, especialment quan aquestes són deficitàries en matèria orgànica.

L'esmena de tecnosols amb fangs de depuradora facilita una ràpida revegetació, especialment pel que fa a espècies herbàcies, fet que permet l'estabilització ràpida de talussos i redueix l'escorrentiu, la qual cosa redueix també l'erosió superficial, que és un objectiu prioritari en talussos d'infraestructures viàries i activitats extractives, on habitualment es treballa amb talussos de fort pendent. No obstant, la mineralització del nitrogen contingut en els fangs pot suposar un risc per a les aigües subterrànies, degut a l'exportació d'elements solubles com el N dels nitrats, si es fa de manera extensa i a dosis altes. La seva aplicació en dosis excessives també pot limitar el reclutament d'espècies arbustives i arbòries, degut a la selecció d'espècies nitròfiles i a la competència amb l'espai ocupat per les herbàcies.

Per tal de minimitzar aquest risc cal aplicar-los en dosis moderades i garantir l'ús de fangs de depuradora degudament digerits mitjançant processos aerobis o anaerobis. Els fangs han de tenir un grau d'estabilitat superior al 30%, si bé és recomanable que aquest sigui major. També cal garantir que els fangs no contenen una excessiva càrrega contaminant. Així, el protocol d'ús de fangs proposat inclou el control dels metalls pesants, dels quals es disposa d'informació continua i fiable ja que s'inclouen en les anàlisis rutinàries que realitza habitualment l'Administració competent i les empreses explotadores de les depuradores. En els darrers 20 anys s'ha notat una millora general en la qualitat del fangs de depuradora, resultat de la implantació de normatives ambientals europees més exigents que controlen les activitats industrials i la gestió dels residus, de manera que més del 90% dels fangs EDAR que es produeixen a Catalunya compleixen les condicions per a ser aplicats a sòls agrícoles, i per tant també per a la rehabilitació de sòls. No obstant, els fangs també contenen altres contaminants, com els orgànics inclosos els microplàstics, dels quals es disposa de molta menys informació, si bé es considera que l'aplicació de fangs de depuradora al sòl és una de les vies principals d'entrada de micro(nano)plàstics al sòl. És certament un tema que mereix ser investigat, juntament amb els metabòlits de medicaments i d'altres contaminants anomenats "emergents".

Per altra banda també cal garantir una qualitat mínima dels substrats minerals usats en la fabricació de tecnosols, que han de tenir un contingut suficient de terra fina, una granulometria equilibrada, així com un contingut de carbonats i un pH que no limitin el desenvolupament de la vegetació. També cal controlar la presència d'impropis, especialment quan s'utilitzen terres procedents d'excavacions o obres, o la salinitat dels substrats.

A mig termini (una dècada), els sòls esmenats amb fangs de depuradora afavoreixen l'enriquiment del sòl en matèria orgànica i el segrest de carboni, ja que hem comprovat que contenen tres vegades més carboni orgànic que els sòls no esmenats. Aquest enriquiment és el resultat de l'augment de la producció primària a causa dels nutrients aportats pels fangs, que encara és evident després de deu anys. Aquest augment de la producció primària es tradueix també en un major recobriment vegetal que protegeix el sòl de l'erosió. En general podem dir que l'aplicació de fangs a dosis moderades, d'acord amb el protocol proposat (conegut com a RESTOFANGS), afavoreix un major desenvolupament de la vegetació, especialment de l'estrat herbaci en els primers anys, però també progressivament de l'arbustiu i l'arbori, sense limitar el reclutament de les plantes pròpies del sistema natural de la zona.

Pel que fa als bioestabilitzats, hem constatat que existeix una important heterogeneïtat en relació als principals paràmetres estudiats. Aquesta heterogeneïtat es dona entre diferents plantes de tractament de residus, però també entre diferents lots d'una mateixa planta, cosa que suposa una restricció a l'hora de fer generalitzacions en relació a l'ús de bioestabilitzats.

Els digests procedents de la digestió anaeròbica de la matèria orgànica obtinguda per la separació mecànica de la fracció resta dels residus urbans, presenten un contingut baix d'impureses i una major concentració de N que els bioestabilitzats, la qual cosa els fa més adequats per a ser aplicats al sòl. L'elevat contingut en N dels digests afavoreix el desenvolupament i el creixement de les plantes, la qual cosa el fa especialment interessants en aquells casos en que cal garantir una coberta herbàcia que protegeixi el sòl de l'erosió. Podem dir que els digests s'assemblen més als fangs EDAR que als bioestabilitzats, malgrat l'origen comú amb els darrers. A més, la composició dels digests és més estable entre plantes i lots.

Pel que fa a la dosificació, hem pogut comprovar que les dosis superiors a 20 $g{\cdot}kg^{-1}$ no milloren el desenvolupament i el creixement de la vegetació, tant en bioestabilitzats com en digests, i també augmenten el risc de contaminació de les aigües subterrànies i la toxicitat sobre la fauna del sòl. El missatge és clar, les esmenes orgàniques generen beneficis als sòls degradats o deficitaris en matèria orgànica, però no té sentit aplicar-les en dosis superiors a les establertes en els protocols que hem publicat o les indicades en d'altres codis de bones pràctiques agràries.

Sobre el seguiment de la restauració

Diferents actors vinculats a la pràctica de la restauració ecològica apunten a la necessitat d'establir protocols de seguiment de la restauració, que han de ser fàcils d'aplicar i aplicables a un ampli ventall de situacions o tipologies de restauració. El protocol RESTOQUARRY, basat en bona part en resultats d'aquesta tesi, pretén contribuir a donar-hi resposta al ser una eina adreçada al personal de les empreses extractives i de l'Administració, que ajuda a incrementar el compromís i la implicació dels enginyers de mines i el personal encarregat d'executar la restauració, la qual cosa es tradueix sens dubte en una major qualitat de les restauracions.

La major part dels indicadors proposats en el protocol avaluen de manera indirecta serveis i/o funcions ecosistèmiques, permetent en alguns casos la seva quantificació. Fins i tot alguns dels indicadors més generals (indicadors del conjunt de l'àrea restaurada), com són els relacionats amb impactes o integració visual, poden ser considerats aproximacions a la valoració de serveis ecosistèmics culturals.

El protocol RESTOQUARRY (o RESTOCAT en la seva versió en català i castellà) permet distingir restauracions de qualitat (*accepted*), d'aquelles que necessiten l'adopció de mesures correctores (*minor* *revisions*), o que presenten problemes crítics que posen en risc l'èxit de la restauració (*major revisions*), o que fins i tot posen en entredit tot el procés restaurador i que requereixen de canvis profunds en l'execució i el disseny dels treballs (*rejected*). No obstant, una crítica que es pot fer al protocol és que, al considerar molts indicadors, és difícil obtenir puntuacions molt baixes. Per això és important el fet que fixi indicadors clau o excloents, que cal assegurar per tal de garantir que la restauració no té limitacions greus. La selecció actual d'aquests indicadors, establerta mitjançant la seva ponderació per part d'un panell d'experts, per una part pot semblar excessivament àmplia de manera que caldria ser més restrictiu, però per altra part, la reducció del nombre d'aquests indicadors clau podria fer que algunes restauracions amb falles crítiques passessin el procés d'avaluació. Si bé és difícil obtenir puntuacions molt baixes, el fet de considerar molts paràmetres d'avaluació també fa difícil obtenir puntuacions molt altes (*accepted*). Això comporta que la major part de les restauracions avaluades amb el protocol requereixen de mesures correctores per tal de millorar algun aspecte. Per tant, es tracta d'una eina que serveix per pujar el llindar de qualitat de la restauració, posant a l'abast indicadors objectius que cal tenir en compte en totes les fases del procés restaurador, des del disseny dels treballs fins a la seva execució, acabant pel seu manteniment.

El protocol ha estat dissenyat i testat específicament en restauracions forestals, o amb vegetació natural, d'activitats extractives en clima mediterrani. Per tant, en altres climes o tipologies d'activitats podria presentar desajustos en alguns indicadors o valors de referència, si bé es podria adaptar fàcilment a tipologies de restauracions que tenen punts en comú amb les de les activitats extractives, com és el cas de la restauració d'abocadors o infraestructures de transport. També, seleccionant alguns indicadors i donant-los un pes diferent, podria ser aplicat a altres models de restauració, com per exemple la restauració com a camps agrícoles. Per altra part, el protocol no és apropiat per avaluar restauracions molt específiques, que busquen restaurar hàbitats molt singulars (protegits o amenaçats). En aquests casos, alguns indicadors o valors de referència poden no ser apropiats, requerint de coneixement expert, de manera que queden fora de l'abast del protocol.

L'aplicació del protocol pot presentar també limitacions en restauracions de zones molt extenses o amb accessos complicats que dificultin la presa de dades. En aquests casos, l'ús de sistemes de seguiment remot pot ser la solució per disposar d'informació fiable i de qualitat, com a mínim sobre uns quants paràmetres clau com poden ser les cobertes del sòl, el desenvolupament i vigor de la vegetació, l'afectació per processos de degradació del sòl o per deposició de pols, entre d'altres. En aquest sentit s'ha treballat en un protocol de seguiment de les restauracions mitjançant drons, que permet quantificar de manera semi-automàtica aquests paràmetres, si bé segurament es podria obtenir encara molta més informació gràcies a la molt alta resolució espacial d'aquestes imatges (en alguns casos poden arribar a píxels de menys de 5 cm) i a la utilització de diferents tipus de sensors, com per exemple combinant càmeres hiperespectrals amb sensors LIDAR. A més, la utilització de les ortofotoimatges disponibles al nostre país de manera gratuïta i amb resolucions espacials relativament altes (25 cm), permet també fer un seguiment de la restauració mitjançant fotointerpretació, després d'un procés d'entrenament i estandardització, que si bé no exclou la necessitat de verificació al camp, suposa una gran ajuda.

Un aspecte que ha quedat fora del protocol, degut a què té implicacions que van més enllà de la restauració d'activitats extractives, és la gestió posterior dels espais restaurats, que esdevé un punt clau per garantir la conversió real d'aquests espais en infraestructures verdes al servei de la societat. Qualsevol projecte de restauració hauria d'incorporar de manera explícita aquest apartat, especificant les mesures de gestió de l'espai i les vies de finançament, ja que esdevé un punt clau per garantir la sostenibilitat d'alguns tipus de restauracions. No obstant, aquest pas no està recollit en la llei de restauració, i per tant, ha de ser abordat mitjançant acords voluntaris, la qual cosa en alguns casos ja s'està fomentant, o fins i tot ja existeix. Aquest aspecte és especialment rellevant, per exemple, en els projectes de restauració que preveuen deixar fronts o parets verticals per tal d'afavorir determinades espècies d'aus, com per exemple l'àguila cuabarrada o el còlit negre, i que preveuen també la restauració d'algunes parts de l'activitat com a àrees de campejat. Aquest model és sens dubte molt interessant, tant per l'ajuda que pot suposar per la recuperació d'aquestes espècies, com pels avantatges que suposa per l'empresa a nivell de reducció de costos de restauració. No obstant, si no es garanteix la correcta gestió d'aquests espais un cop finalitzi l'activitat, el seu valor de cara al foment d'aquestes espècies pot quedar molt minvat, arribant a constituir de nou un passiu ambiental, i per tant fent fracassar tot el procés de restauració.

Un altre aspecte que ha quedat al marge del protocol és la implicació dels agents locals en el procés de disseny i avaluació, així com en l'ús i/o gaudi posterior de l'espai restaurat. S'ha de tenir en compte que, segons la concepció actual de la restauració ecològica, la implicació del públic d'interès (*stakeholders*) és un dels pilars bàsics que cal tenir en compte a l'hora de dissenyar i executar un projecte de restauració. Per altra banda, és una eina molt valuosa d'educació ambiental que pot contribuir a ajudar a conscienciar a la població de la necessitat de disposar de sistemes naturals de qualitat i de restaurar els que estiguin degradats. En aquesta línia, un altre dels aspectes que pot ajudar és la quantificació dels beneficis econòmics i de creació de llocs de treball que pot generar el propi procés de restauració i la gestió posterior de l'espai, al marge de la monetització dels serveis ecosistèmics. Tots aquests aspectes són molt rellevants i cal abordar-los d'una manera àmplia, ja que han d'ajudar també a legitimar l'acció política i mobilitzar les inversions que es requereixen per afrontar la restauració dels espais degradats de manera massiva, tal com preveu i pretén la declaració de les Nacions Unides de la dècada 2020-2030 com la de la Restauració dels Ecosistemes, o el *Green Deal* de la UE.

Altres actuacions en curs o a desenvolupar

Precisament la implicació de la societat i dels agents locals en la restauració és i ha estat una de les preocupacions del Grup de Recerca de Protecció de Sòls del CREAF. Aquest grup s'ha caracteritzat per dur a terme una recerca aplicada, útil per la societat, que resolgui problemes reals, sense deixar de banda qüestions de recerca fonamental. Els treballs presentats en aquesta tesi sempre han pretès implicar el màxim d'actors en el disseny i validació de metodologies, intentant respondre a les seves demandes i suggeriments. Un exemple n'és el protocol RESTOQUARRY, que va implicar actors de diferents disciplines professionals en un panell d'experts, i que posteriorment va ser validat directament al camp amb els usuaris finals, que van poder dir-hi la seva, per exemple proposant simplificacions de les metodologies o fins i tot altres maneres d'aplicar-les. No obstant, és evident que calia anar més enllà, i fomentar molt més la implicació de la població en els treballs de restauració, també els que s'executen amb objectius científics. En aquests sentit, durant els darrers anys he intentat involucrar molt més a la població local, habitants dels municipis, propietaris forestals, en els projectes de restauració. Un exemple d'això és el recent acabat projecte LIFE The Green Link, que he coordinat, i en el que hem provat nous mètode de restauració en diferents indrets de la conca mediterrània, sempre intentant implicar en totes les fases del projecte (disseny, execució, seguiment) el màxim d'*stakeholders* possibles. Un dels aspectes que no hem pogut implementar en aquest projecte ha estat la implicació de col·lectius desafavorits, vulnerables o amb dificultats especials d'inserció laboral. És per això que en una nova proposta que està al forn, i que pretén contribuir als programes i objectius de restauració abans esmentats, estem intentant involucrar també aquests col·lectius mitjançant empreses del tercer sector ambiental, a banda d'altres tipus de professionals que també es poden vincular a la restauració d'espais degradats o abandonats.

Un altre dels aspectes que estem treballant des del Grup, i en el que estic especialment involucrat, és en l'ús de noves tecnologies i sensors remots en el seguiment d'espais degradats i la seva restauració. Ja des de fa anys hem fet alguns treballs explorant les possibilitats d'utilització de diferents fonts d'informació obtingudes per teledetecció (ortofotoimatges RGB, imatges satèl·lit hiperespectrals, dades LIDAR) en el seguiment dels treballs i els resultats de l'explotació i la restauració d'activitats extractives (detecció d'afectacions, mapes de cobertes, recobriment per estrats, desenvolupament de la vegetació arbustiva i arbòria), amb resultats modestos. No obstant, més recentment estem treballant en l'ús de drons en el seguiment de la restauració i l'explotació d'activitats extractives, on hem obtingut resultats molt interessants gràcies a les prestacions d'aquesta tecnologia que s'ajusten molt bé a les característiques de les activitats extractives a casa nostra. En aquest sentit, s'han publicat dos articles sobre el tema (*Monitoring opencast mine restorations using Unmanned Aerial System (UAS) imagery,* Padró *et al.* (2019); *UAS remote sensing protocol for restoration and extraction monitoring in quarries,* Carabassa *et al.* (2020)) i fins i tot s'ha publicat un protocol tècnic adreçat als professionals del sector (Especificacions tècniques per a l'ús de drons en el seguiment d'activitats extractives, CREAF-DTES, en fase de maquetació en el moment d'escriure aquestes línies) per tal de fomentar l'ús d'aquesta tecnologia amb multitud aplicacions en l'àmbit de la mineria i la restauració ecològica, entre molts d'altres.

La llista de temes interessants a desenvolupar podria ser més llarga però per acabar aquesta secció volia esmentar un projecte que fa temps que volem encetar des del Grup, i que esperem poder iniciar aviat. És en relació a l'estudi de la càrrega de grups de contaminants poc estudiats a les esmenes orgàniques, especialment als fangs de depuradora, bioestabilitzats i composts. Com s'ha comentat anteriorment i en diferents capítols, aquestes esmenes tenen una càrrega contaminant que va més enllà del seu contingut en metalls pesants, i sobre la que darrerament alguns autors estan començant a alertar. És el cas per exemple dels plàstics, en totes les seves mides (macro, micro i nano), que trobem presents a vegades en proporcions massa grans, en algunes esmenes utilitzades en restauració i fins i tot en sòls agrícoles. Com ja he comentat anteriorment, cada cop hi ha més veus que alerten que l'aplicació d'algunes esmenes orgàniques, especialment els fangs de depuradora, podria ser una de les principals vies d'entrada de plàstics al sòl. Per tant, per tal d'assegurar una gestió ambientalment segura d'aquests residus és molt important aprofundir en l'estudi de la seva càrrega contaminant..

9. Conclusions

Seguidament es recullen les principals conclusions d'aquesta tesi:

Determinació de concentracions efectives de metalls pesants (Cu, Zn, Cr) sobre l'activitat microbiana del sòl

- S'ha comprovat que el protocol OCDE-217 és una metodologia vàlida per a la detecció d'efectes adversos de la contaminació del sòl per tres metalls, Cu, Zn i Cr, ja que afecten a la respiració del sòl quan s'afegeix un substrat biodegradable.
- La magnitud dels efectes inhibidors que s'han observat pels tres metalls estudiats (Cu> Zn>Cr) és coherent amb els treballs publicats per altres autors que han treballat amb sòls de característiques similars, essent el Cu el metall més tòxic després de 28 dies d'exposició.
- L'adició de matèria orgànica fàcilment biodegradable emmascara l'efecte dels metalls pesants sobre la respiració induïda pel substrat ja que incrementa la respiració del sòl.

Utilització de fangs de depuradora com a esmena orgànica en rehabilitació de sòls

- L'esmena de sòls o substrats amb fangs de depuradora facilita una ràpida revegetació, especialment pel que fa a espècies herbàcies, que permet l'estabilització ràpida de talussos, redueix l'escorrentiu, i la converteix en una pràctica molt efectiva pel control de l'erosió hídrica.
- Cal, però, que els fangs compleixin unes condicions de qualitat, com unes concentracions màximes de metalls pesants, que estiguin degudament estabilitzats per digestió aeròbica o anaeròbica, i siguin aplicats en dosis moderades.
- Cal garantir una qualitat mínima dels substrats minerals receptors de fangs de depuradora, que han de tenir un contingut suficient de terra fina, una granulometria equilibrada, així com un contingut de carbonats i un pH que no limitin el desenvolupament de la vegetació. També cal controlar la salinitat i la presència d'impropis, especialment quan s'utilitzen terres procedents d'obres.
- A mitjà termini, els sòls esmenats amb fangs de depuradora afavoreixen l'enriquiment del sòl en matèria orgànica i el segrest de carboni, ja que s'ha constatat que poden contenir tres vegades més carboni orgànic que els sòls no esmenats.
- En general, l'aplicació de fangs a dosis moderades afavoreix un major desenvolupament de la vegetació, especialment de l'estrat herbaci en els primers anys, però també de l'arbustiu i l'arbori a més llarg termini, sense limitar el reclutament d'espècies natives.

Potencialitats d'ús de bioestabilitzats i digests de FORM com a esmena orgànica de sòls

- Els bioestabilitzats presenten una composició heterogènia en relació als principals paràmetres estudiats. Aquesta heterogeneïtat es dona entre plantes de tractament diferents, però també entre diferents lots d'una mateixa planta, cosa que suposa una restricció molt important per poder fer generalitzacions en relació a l'ús de bioestabilitzats. En canvi, la composició dels digests és més estable entre plantes i lots.
- Els digests procedents de la digestió anaeròbica de la matèria orgànica obtinguda per la separació mecànica de la fracció resta dels residus municipals presenten un contingut baix d'impureses i un elevat contingut en N, la qual cosa els fa adequats per a ser aplicats al sòl.
- L'elevat contingut en N i altres nutrients dels digests afavoreix el desenvolupament de la vegetació, la qual cosa els fa especialment interessants en aquells casos en que cal garantir una coberta herbàcia que protegeixi el sòl de l'erosió.
- Pel que fa a la dosificació, les dosis superiors a 20 $g \cdot kg^{-1}$ no milloren el desenvolupament de la vegetació, tant de bioestabilitzats com de digests, i augmenten el risc de contaminació de les aigües i la toxicitat sobre la fauna del sòl.

Protocol d'avaluació i seguiment de la restauració

- S'ha demostrat que el protocol RESTOQUARRY ajuda a incrementar el compromís i la implicació dels enginyers de mines i el personal encarregat d'executar la restauració, la qual cosa es tradueix en una major qualitat de les restauracions efectuades.
- El protocol RESTOQUARRY permet identificar els factors limitants d'una restauració. Així, permet distingir restauracions de qualitat d'aquelles que necessiten l'adopció de mesures correctores, les que presenten problemes crítics que posen en risc l'èxit de la restauració, o que fins i tot les que posen en entredit tot el procés restaurador i que requereixen de canvis profunds en l'execució i el disseny dels treballs.
- El protocol RESTOQUARRY estableix indicadors clau que cal assegurar per tal de garantir que la restauració no té limitacions greus. S'ha observat, però, que la selecció actual d'aquests indicadors, establerta mitjançant ponderació per part d'un panell d'experts, pot ser excessivament àmplia i caldria ser més restrictiva. No obstant, la reducció d'aquests indicadors clau pot fer que algunes restauracions amb falles crítiques passessin el procés d'avaluació. Per tant, caldrà revisar el protocol quan es disposi de l'experiència d'aplicació a més casos i més llarg termini.
- La major part de les restauracions avaluades amb RESTOQUARRY requereixen de mesures correctores per tal de millorar algun aspecte de la restauració, fet que demostra l'eficàcia del protocol.
- El protocol està dissenyat i testat específicament per a restauracions d'activitats extractives en escenaris de vegetació natural, forestal i altres, en clima mediterrani. Per tant, en d'altres climes

o tipus d'hàbitats podria presentar desajustos en alguns indicadors o valors de referència, si bé es podria adaptar fàcilment a restauracions agrícoles i també a altres tipus d'activitats relacionades.

- El protocol RESTOQUARRY no és apropiat per avaluar restauracions molt específiques, que busquen recuperar hàbitats molt singulars o adreçades a espècies que presenten requeriments especials.
- La major part dels indicadors proposats en el protocol RESTOQUARRY avaluen de manera indirecta serveis i/o funcions ecosistèmiques, permetent en alguns casos la seva quantificació. Fins i tot alguns dels indicadors més generals del conjunt de l'àrea restaurada, com són els relacionats amb la integració paisatgística, poden ser considerats aproximacions a la valoració de serveis ecosistèmics culturals.
- Elprotocol RESTOQUARRY és una eina que serveix per pujar el llindar de qualitat de la restauració d'activitats extractives, posant a l'abast indicadors objectius de la restauració, que cal tenir en compte en totes les fases del procés restaurador, des del disseny dels treballs fins a la seva execució, continuant pel seu manteniment.