GOLD

Growing energy crops on contaminated land for biofuels and soil remediation

Ref. Ares(2023)3491187 - 18/05/2023

D3.1

Extent, location and contaminated land status potentially suitable for phytoremediation 2022

Document Summary

Deliverable Number: 3.1

Version: 1

Due date: 31.10.2022

Actual submission date: 14.05.2023

Work Package: 3 – Integrated sustainability assessment for bridging the gap

Task: 3.1. – Mapping selected contaminated lands and phytoremediation scenarios

Lead beneficiary: Wageningen Research

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Dissemination level: Public

Document history

Version	Date	Beneficiary	Reviewer
1.0 send for internal review	06-02-2023	WR	Giacomo Talluri

Horizon 2020 Grant Agreement Number: 101006873

Project Start Date: 15 April 2021

Duration: 48 months

Project coordinator: CRES

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Executive Summary

This report presents the results of the first part of task 3.1 in GOLD which aims at mapping in detail the contaminated sites in the EU and their characteristics. A distinction is made between sites/or rather areas affected by diffuse and by point source pollution.

1) **Diffuse pollution** (def. EEA: ***Pollution*** *from widespread activities with no one discrete source, e.g. acid rain, pesticides, urban run-off, etc*.)

2) **Point source pollution** (def. EPA: *Pollution from any single identifiable source (e.g. landfill, mine, industrial site)*

Diffuse pollution

Diffuse pollution is ongoing in the EU and has already caused widespread emission of a range in pollutants including nutrients, organic pollutants and metals. Effects of diffuse pollution on water quality are well documented and effects of proximity pollution are known in various member states, however for soil pollution this is different. Current soil screening values (SSV's) are targeting point source pollution mostly whereas soils affected by diffuse pollution often do not exceed such SSVs. This does not imply that diffuse soil pollution poses no risk to the soil ecosystem or quality of food and fodder. A direct assessment of the current soil quality as affected by diffuse pollution is however not possible since SSVs currently in use are specific for individual member states. At EU level there is currently no agreed uniform screening level that can be used as a first approximation to allocate areas that need remediation. Therefore, in this project we propose a risk assessment model instead that is applied based on specific risks in view of ecosystem health, food quality and water quality. This approach assumes that there is a connection between soil quality as expressed by relevant soil properties (for metals based on pH, organic matter, and clay) and the acceptable pollutant concentration at which the risk for either food, water or ecosystem is avoided. The resulting regional critical concentrations in soil can be compared with actual concentrations to detect areas at risk.

Critical concentrations of pollutants in soil can be related to critical concentrations in three environmental compartments: water, food and soil dwelling organisms. For each of these, three critical concentrations are available. For food, critical concentrations are based on WHO food quality criteria, for water critical concentrations based on drinking water criteria or aquatic organisms are available. For soil dwelling organisms critical concentrations in solution have been derived from laboratory studies for a large number of species. All of these can be converted to a corresponding critical concentration in soil that can be compared to current, measured concentrations in soil. For food and ecotoxicology the results are realistic in that the pollutant in the soil is in direct contact with either plant roots (uptake) or the soil dwelling organisms. For water quality the calculation is a worst-case approach since it would assume that water leaving the topsoil is in equilibrium with the groundwater. An alternative approach for water is available but requires a substantial amount of both soil chemical and hydrological data both of which are not available at EU level.

A major advantage of the risk based approach as outlined in this chapter is that metal concentrations across member states can be compared using the same criteria considering specific risks for humans and the environment. Here risks are expressed in calculated critical concentrations in soil as related to the quality of food, drinking water and ecotoxicology.

Maps of heavy metals are available for Pb. For Cu and Zn and have been used to construct spatially explicit maps at EU level. The calculation of critical concentrations of metals in soil beyond which the critical

concentration in water or food is exceeded requires additional information on soil properties. Key properties include soil organic carbon, pH and clay content. Here we use the two largest databases currently available (LUCAS and SoilGrids) that do contain all required soil properties but do reveal however substantial differences in the spatial pattern and absolute level of soil carbon.

The differences in organic carbon led to markedly different critical concentrations for Cd, Cu and Pb. Most noticeable are the lower critical concentrations calculated based on the LUCAS database in Poland, Spain and part of Portugal and Italy. This also leads to differences in the level of exceedance at country level. In general, however, the exceedance risk of Cd critical concentrations appears to be limited as is the exceedance risk in view of ecotoxicology for Pb. For Cu and Zn the exceedance of the ecotoxicological critical concentrations is larger. This is partly related to higher concentrations of Cu in areas in the Mediterranean countries and, for Zn, related to a combination of low pH and low soil carbon concentrations in among others Poland, parts of Spain and Portugal.

However, the difference in the exceedance when comparing results based on LUCAS data versus those based on SoilGrids suggests that these results need to be used with care. Both uncertainty related to differences in basic soil properties as well as model uncertainty (not addressed further in this study) can lead to a substantial range in both the actual concentration of metals and soil carbon and also in the absolute level of the critical concentration.

Despite these shortcomings, the approach outlined here is a promising way to identify areas that are or can be at risk of pollution by the metals addressed in this study. It is however recommended to critically evaluate current soil databases to establish the reliability of maps derived from these databases. In addition, model uncertainty in many of the models used here can be reduced when more data become available. This specifically relates to models used to predict the concentration of metals in food. In contrast to data on soil, data on crop (product) quality and soils where these crops are grown are scarce. This is even more of an issue when considering many of the emerging contaminants that are or will become an issue in view of food safety.

Mapping contaminated sites

Enquiry at JRC-ESDAC and consultation of the websites of EEA and Eurostat revealed that at present, there is no database of contaminated sites for Europe that carries spatially referenced information on area and contaminants. Because of lack of EU wide spatially explicit sources on contaminated sites another approach to mapping these contaminated sites was developed. For this reason, in this study we have taken another approach to mapping contaminated sites, i.e. to identify potentially contaminated areas from Open Street Map based on properties of geographical objects, and to cross-check these areas with information on land cover and with recordings of contaminated sites in the literature and the internet. In addition, national registers of contaminated sites were also consulted for several countries in 2023, but with relatively little success.

In addition polluted areas using other data than OpenStreetMap (OSM) was also applied because not all types of pollution are covered using OSM alone. This applies to land currently in use as agricultural land, that was previously used for irrigation with or treatment of wastewater, or for the disposal of sewage sludge.

The results of the contaminated sites identification show that the total area estimated in potentially contaminated sites due to military training activities, industrial activities, mining and landfills, of which less than 40% is sealed, amounts to 2,013,722 ha. This corresponds to 0.5% of the total area of the countries considered. In individual countries, the area of potentially contaminated sites identified on Open Street Map is at most 1% of the total surface area of the country. France, Germany and Spain have the largest total areas of all types of potentially contaminated sites, amounting to more than 150,000 ha in each of the countries.

The largest areas of potentially contaminated sites are in areas tagged on OSM as military sites (41%), industrial sites and brownfields (29%), quarries (25%) and landfills (4%). The land cover from CLC2018 in the considered OSM sites corresponded to the expected land cover for the larger part, i.e. forest and other semi-natural vegetation for military sites, industrial or commercial units for industrial sites and brownfields, mineral extraction sites for quarries and dump sites for landfills. This supports the correct selection of the sites in OSM.

In sites where pollutants may occur, land cover consisting of densely built-up area, forest or other natural vegetation is considered unsuitable for phytoremediation as these types of land cover areas are either already vegetated by trees & shrubs or sealed by buildings and roads. This also applies to other land cover types unsuitable for cropping, such as beaches and dunes, bare rocks and water bodies. Land cover types in potentially contaminated sites with discontinuous urban fabric (e.g. mineral extraction sites) and with some form of agricultural land use are considered suitable for phytoremediation, provided that less than 40% of the area is artificially sealed (impervious). The total area of potentially contaminated sites with land cover types suitable for phytoremediation, and with less than 40% of the area sealed (impervious), amounts to 2,013,722 ha in the EU27 and UK. This area corresponds to 0.5% of the total surface area of these countries.

France, Germany, Spain and UK have the largest total areas of all types of potentially contaminated sites, amounting to more than 150,000 ha in each of the countries.

Land currently in use for agriculture covers between 7% (in military sites) and 20% (in landfills) of the area in potentially contaminated sites identified in OSM. These areas offer opportunities for phytoremediation through biomass cropping, because less effort is required for conversion of the land use than if the area would be covered by constructions or natural areas.

The Minerals4EU database features 42,731 mines in 22 EU Member States in 8 commodity groups considered of interest for phytoremediation. Of these, only 738 were found in proximity of potentially contaminated sites identified in Open Street Map. A large number of mines in the Minerals4EU database (20,137) was not identified in OSM, and of this number, only 204 are indicated as mines in the land cover class ‘mineral extraction sites’ in the Corine Land Cover database (class nr 7). These findings show that the databases with European coverage OSM and CLC2018 represent only a small part of the potentially contaminated sites, and that dedicated databases with spatial information on geographical objects associated with local contamination are required to map contaminated sites.

Commodities produced in mines, as specified per mine in the Minerals4EU database, were ranked according to the risk for human health and the possibility to reduce the risk in the site with biomass crops, and the likeliness of three modes of phytoremediation to manage the commodity. In 57% of the mines, commodities pose a high risk to human health and there is a need to remediate the contamination. For the commodities in this group phytoremediation might be possible to reduce the risk. In 40% of the mines, commodities do not pose a high risk for human health and the need to apply remediation is low.

Of the total of 20,708 mines observed in land cover classes considered relevant for phytoremediation, almost half (10,206) are located in areas with agricultural land use. These findings suggest a potential for options to use existing agricultural land in (former) mine areas for biomass crop production.

21% of the areas indicated as landfill in OSM is covered with some form of agricultural land, mainly by non-irrigated arable land and pastures, which may be relevant for phytoremediation using bioenergy crops, in case soil pollution is present. This requires an assessment at the level of these sites.

The total area of landfills in EU27 and UK on Open Street Map is 99,992 ha, overlapping with 88% of the total area of dump sites on CLC2018 (113,763 ha). This might suggest that not all landfills are identified in Open Street Map. However, there are also countries where the total area of polygons tagged as 'landfill' in Open Street Map is larger than the total area covered by dump sites on the CLC2018 map. Again it confirms to the need to consult multiple spatial datasets for the purpose of mapping potentially polluted areas in or around landfills.

Brownfields may be considered a sub-set of industrial areas. In Open Street Map, 66,048 ha was tagged as both types of land use in the EU27 and UK, corresponding to 94% of the total area of brownfields. For the generation of a map of potentially contaminated sites, the polygons tagged as industrial areas and brownfields on Open Street Map were therefore merged. This results in a total of 2,725,502 ha of industrial sites and brownfields, occurring in the EU27 plus the UK. Of this area 167,877 ha is in use by some form of agriculture (according to the overlay with CLC2018), which may be relevant for phytoremediation using bioenergy crops, in case soil pollution is present.

In the category of industrial sites, steel production sites with blast furnaces may deliver pollution risks through the emission of fine particles, but pollution of soils has not been demonstrated. It is however conceivable that vegetation might be used to stabilize particulate matter in the vicinity of the steel production sites and to prevent transport to other areas. 27 steel production sites with blast furnaces were mapped in the EU, with land cover in an area of 5 km around these sites. Considering only land cover types suitable for phytoremediation with <40% imperviousness, 60% of the area currently has land cover reflecting agricultural use. This might offer potential to deploy the area for stabilization of fine particulate matter by biomass crops.

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1 Introduction

1.1 Aim

This report presents the results of the first part of task 3.1 in GOLD that aims at mapping in detail the contaminated sites in the EU and their characteristics. A distinction is made between sites/or rather areas affected by diffuse and by point source pollution:

1) Diffuse pollution (def. EEA: ***Pollution*** *from widespread activities with no one discrete source, e.g. acid rain, pesticides, urban run-off, etc*.)

2) Point source pollution (def. EPA: *Pollution from any single identifiable source (e.g. landfill, mine, industrial site)*

In the following we first present some more background to the issue of contaminated sites/areas and why and with what purpose these are to be identified spatially in the GOLD project. The last section elaborates further on the organisation of this report is chapters.

1.2 Context

When considering soil contamination one has to distinguish between point source pollution usually occurring at or near contaminated, active or abandoned industrial sites versus diffuse pollution usually affecting larger areas of land, including agricultural land. Point source pollution usually affects a limited surface area such as former industrial, mining or land fill sites (**Error! Reference source not found.**). The type of pollution present in soil usually is directly related to a specific active or historic source and often reaches high concentrations in excess of soil standards aimed at the protection of human or animal health as well as ecosystem health.

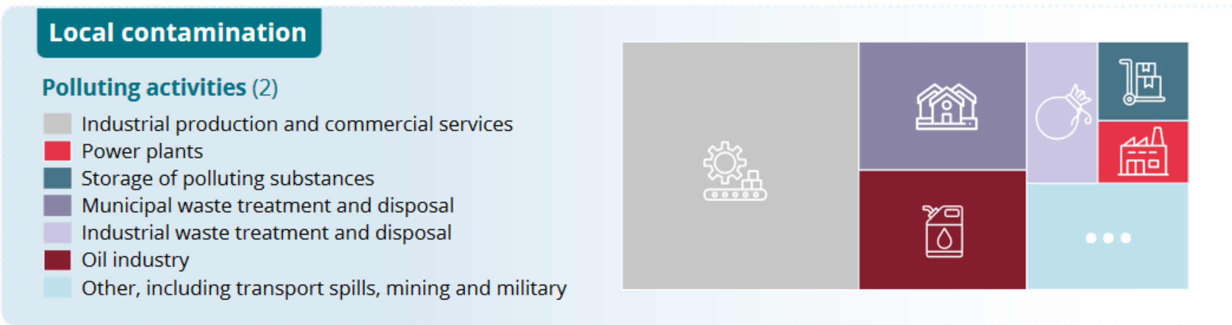

Figure 1 Activities leading to local contamination. Source: EEA (https://www.eea.europa.eu/themes/soil/soil-threats).

Areas affected by diffuse pollution not only cover a far greater area, often also the source of the pollution is not clear and includes sources like traffic, industry (both leading to deposition via air) as well as agriculture. An additional difference between point source and diffuse pollution is the type and level of pollution. Point source pollutants often have a direct link to a specific industry which sometimes leads to a limited number of pollutants present but at high levels. Diffuse source pollution via air or because of agricultural soil management can be characterized by a wide range of substances present in soil ranging from metals (in fertilizers and manure) to nutrients (N and P), biocides, persistent organic pollutants present in sludge applied to land as well as soil acidifying substances like ammonia emitted from nearby intensive animal husbandry farms (Huber et al., 2008).

Most European countries by now made good general inventories of contaminated sites, the type of pollutions and the status of remediation. Some of this information is collected and European wide level through a joint effort of the EC-JRC and the EIONET of the EEA through a survey with National Reference Centres (NRCs) in 39 European countries. The results of the survey have been reported in a JRC publication (Payá Pérez & Rodriguez Eugenio, 2018) and the EEA Land and soil indicator 'Progress in management of contaminated sites'. It shows that that there are approximately 2.5 million sites in the EEA-39 countries which are potentially contaminated. National and regional inventories of replying countries reveal that out of these 2.5 million sites 650,000 are registered sites where polluting activities took or are still taking place. Out of these more than 65,500 sites have been remediated. Contaminants most frequently encountered include mineral oil and heavy metals. The most used remediation technology applied, notably to point source polluted areas is "dig-and-dump", which involves excavation and -off-site- disposal of contaminated soil.

Diffuse pollution of top soils has also been systematically assessed across countries in different projects. Three main studies have been done that provide a quite detailed overview, particularly on metals. A study by the JRC (Toth et al., 2017) based on the LUCAS 2009 top soil survey[1] showed the distribution of concentration of metal(loid)s. In the large majority of agricultural lands the concentration of metals is very low and far below levels that pose a threat to human health. Still Toth et al. (2017) estimated that in 6 % of the agricultural surface of the EU (approx. 137 000 km^2) there are elevated levels that need local assessment and potentially remediation action. This is also the land on which the new LUCAS topsoil survey 2018 concentrated and most probably the new ongoing LUCAS 2022 survey.

A previous study by the GEMAS project (Reiman et al., 2014) which was done with the Euro GeoSurveys Geochemistry Expert Group and Eurometaux, analysed samples of arable and grassland soils sampled in approximately 2000 points used as arable land (Ap-horizon, 0-20 cm, regularly ploughed fields) and another 2000 sampling points in permanent grasslands (grazing land soil, 0-10 cm) across Europe. The conclusion from GEMAS was that in the very vast majority of agricultural land contamination with metals is very low. Later this was also found by Toth et al. (2017). Reiman et al. (2014) even concluded that the impact of diffuse pollution on quality of agricultural soils was 'vastly overestimated' but that pollutions by anthropogenic sources plays an important role at a much more local level, particularly around large cities (e.g. mercury concentrations). In addition, Reimann et all. (2014) pointed out that for metals like cobalt, copper and zinc

[1] Topsoil Survey. Arsenic (As), cadmium (Cd), cobalt (Co), chromium (Cr), copper (Cu), iron (Fe), mercury (Hg), nickel (Ni), magnesium (Mg), manganese (Mn), phosphorous (P), lead (Pb), vanadium (V) and zinc (Zn) were analysed in about 22,000 topsoil points, with a sampling density of 200 km2. The analysis of heavy metals will be repeated again in the LUCAS 2018 Topsoil Survey. On this occasion, the analysis for metals will be repeated only in topsoil samples collected from locations where the concentration of the metal was above a certain threshold value. (Liederke et al., 2018)

regional deficiencies may occur due to low native levels in soil, prevailing geochemical conditions (e.g. high soil pH) or low supply to the soil.

An older study by FORESG project (Lado et al., 2008) modelled the spatial distribution of eight critical heavy metals (arsenic, cadmium, chromium, copper, mercury, nickel, lead and zinc) in topsoils in Europe. It was based on 1588 georeferenced samples from the Forum of European Geological Surveys Geochemical database (26 European countries). The results showed the highest concentrations of heavy metals, although not per se exceeding critical human health threat levels, in (1) Liege (Arrondissement) (BE), Attiki (GR), Darlington (UK), Coventry (UK), Sunderland (UK), Kozani (GR), Grevena (GR), Hartlepool & Stockton (UK), Huy (BE), Aachen (DE) (As, Cd,Hg and Pb) and (2) central Greece and Liguria region in Italy (Cr, Cu and Ni). It also concluded that not all elements could be mapped satisfactory particularly for Cr, Cu, Hg and Zn (36–41%) and most unreliable was Cd. Elements that could be mapped best were As, Ni and Pb.

In addition to the dig-and-dump numerous other soil clean-up in-situ technologies have been developed including electro reclamation, microbially enhanced degradation and many others (Ok et al., 2020; Ossai et al., 2020). For most persistent pollutants however, such technologies either are too expensive to be applied at large scale or are too ineffective to obtain the desired target levels within a specific amount of time or budget. Hence removal of the polluted soil still is effectively the only way to decrease the pressure to the environment of human health resulting from the presence of pollutants. This approach (dig and dump) as well as most other soil clean up technologies is not very suitable to deal with diffuse polluted areas due to the sheer volume of soil to be treated. This calls for alternative methods to be applied.

One such technology is phytoremediation of soil. Originally phytoremediation was quickly promoted as a ‘green, low cost’ technique that was able to remove contaminants from soil via plant uptake. Despite numerous reported ‘success’ stories of phytoremediation, largely confined however to small scale (pot)trials, it became clear that even though plants are able to remove contaminants from soil, the ultimate effectiveness under field conditions depends on the type and level of pollution. Nevertheless, phytoremediation can be a successful strategy to be used in those cases where the pressure on land to be used for other purposes or pollution levels are low to moderate but still in excess of for example agricultural advisory levels. In addition, rather than aiming to clean the soil, crops grown on moderately or highly polluted soils also can prevent deleterious effects of the pollutants present on nearby water systems or humans living in the vicinity of such sites. Crops not only reduce leaching rates due to evaporation; they also can reduce emission of dust which can affect nearby residents. In addition, plants can increase soil health by providing nutrients and organic matter to the rhizosphere (Peco et al., 2021).

The merits of growing crops can be further enhanced if crops are used that can be (partly or in its entirety) used as biomass feedstock to produce biofuels from. The key advantage of this combined strategies therefore is that it tackles both the extent and impact of pollution on health and the environment and in addition uses marginally suitable land, now often left bare, for energy production. The biomass from these type of lands, which can be potentially categorized as ‘degraded’ can then also be certified a low ILUC biomass under the current EU’s Renewable Energy Directive II (RED II) mandates.

Clearly such a combined strategy, and the potential for upscaling across the EU requires several technical issues to be resolved. In this report we present the approach to map contaminated sites/areas in the EU further in order to understand their real extend and the type of polluted lands that are suitable for bioremediation in combination with biofuel production. There are several mapping challenges to be overcome to derive acceptable spatial estimates, location and classification of contaminated lands in the EU in relation to best strategies to stabilize and clean the soils.

1.3 Structure of this report

This report consists of 6 chapters. The second chapter addresses the issue of diffuse pollution. First it is discussed how the thresholds are to be determined for identifying areas where contamination levels exceed critical limits and which are candidates for phytoremediation. Here the work presented aims to identify EU wide risk based approach in which one or more critical limits in so-called end-points are used as target not to be exceeded. The approach of mapping areas where these end-points for diffuse pollution are exceeded is then further worked out in chapter 3 and in chapter 4 the results of the implementation of this approach, including in relation to data availability, to derive the spatial estimates of areas where critical end-points of diffuse pollutions are exceeded and which are potential candidates for bioremediation. In Chapter 5 the approach to mapping contaminated sites suitable for bioremediation in combination with biomass production for biofuels is addressed. It will first elaborate on the availability of data and then propose an approach on how to derive spatially explicit estimates of these contaminated sites given data availability and the objective of GOLD. The report finishes with conclusions and a description of further work planned in the project within WP3.

2 Diffuse Pollution in the EU

2.1 Introduction

Despite the, on average low to normal concentrations in most topsoils in the EU, inputs of contaminants to soil are still ongoing and diffuse pollution of soils in the EU is a still widespread phenomenon. During the last decades however, the relevance and contribution of common sources of diffuse pollution have changed. Emission by industry and traffic have been greatly reduced (a.o. emission of Cd and Pb) whereas emission via agriculture (manure, sludge, compost) has remained virtually unchanged. In contrast to areas affected by what is called point source pollution, there is, in case of diffuse pollution no direct connection to a specific source. In the literature diffuse pollution therefore has been defined as 'Pollution from widespread activities with no one discrete source, e.g. acid rain, pesticides, urban run-off, etc.' (EEA, 2022). Previously alternative definitions have been proposed especially targeting the interaction between the terrestrial and aquatic ecosystems: "the release of potential pollutants from a range of activities that individually may have no effect on the water environment, but at the scale of a catchment can have a significant impact (i.e. reduction in water quality, decrease in wildlife, ..." SEPA (2013). In the latter definition, which also has been included in the Water Framework Directive (EC2000/60), a key aspect is introduced which relates to the pathways that connect emission of pollutants to the effect they resort. In view of the WFD 'effects' are primarily targeting the quality of surface waters. A recent update of the impact of diffuse pollution related to agriculture on groundwater quality reveals that substantial areas in the EU are affected by diffuse pollution (EEA, 2022) (Figure 2). Note however that this may relate to various kinds of 'pollutants' including nutrients like N and P which are essential in soil but can pose a threat to ground- and surface waters at the same time.

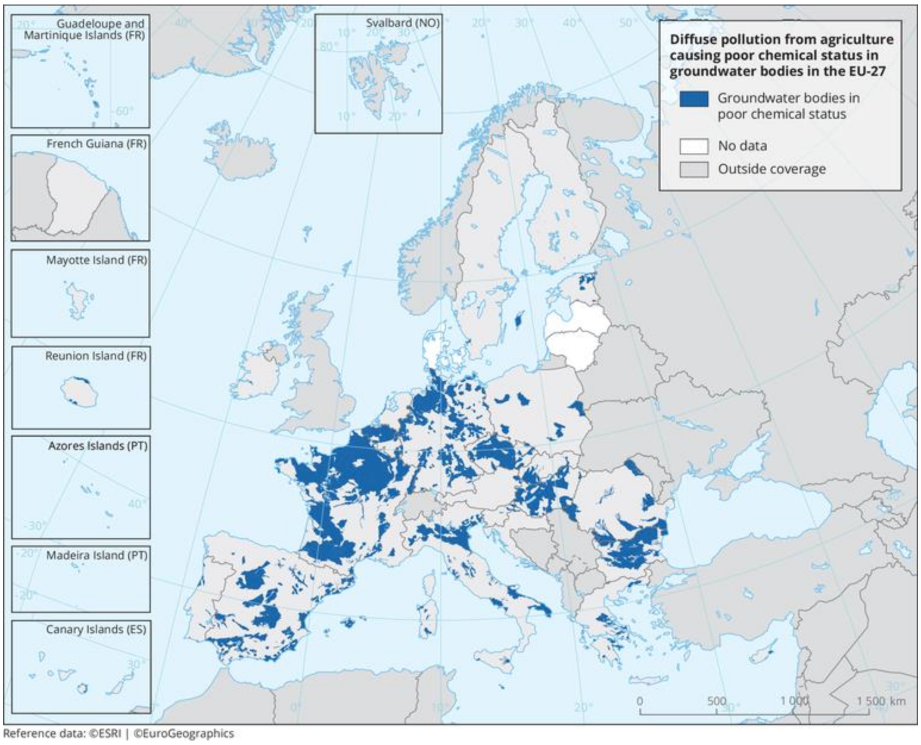

Figure 2 Impact of diffuse pollution on groundwater quality in the EU-27 Member States (EEA, 2022)

In view of a more holistic approach to soil pollution other effects need to be considered as well. Relevant effects include among others the protection of the soil ecosystem and food quality intended for human and animal consumption to mention two relevant aspects.

Typically for diffuse pollution is a slow but ongoing build-up of levels of pollutants in soil. At present, levels of most pollutants in soil have not yet reached critical levels beyond which effects in soil become noticeable. Exceptions to this are areas that have been subjected to higher inputs of pollutants due to for example the presence and emission by industry. Examples of such industrial emission include those by smelters of Cd and Zn ores which have caused widespread pollution in for example the Dutch-Belgian border area 'de Kempen'. This in fact is a special case of diffuse pollution, often called proximity pollution (van Camp et al., 2004), since there is one specific source that has caused the majority of the emission (and subsequent deposition) in a specific area.

An example of the assessment of the impact of agriculture on the quality of arable soils related to diffuse inputs also revealed that for metals of concern such as Cd, predicted changes of Cd concentrations in soil are slow and small compared to current levels in soils (Römkens and Smolders, 2018). Nevertheless current model approaches allowed to assess, at country level whether or not accumulation actually occurs. In most, if not all countries however, predicted levels remained well below actual risk levels or soil screening values (SSV's) used at member state level.

This is not surprising since that in most cases of diffuse pollution actual concentrations of pollutants of concern are well below current soil screening values (SSV's) that have been derived in most EU member states to detect soil pollution. The detection of areas affected by diffuse pollution is more complicated than that of areas affected by point source pollution. The three main reasons for this are that it typically would require a much larger area to investigate (monitoring); it requires much more data to actually determine the speed of accumulation in soil and last but not least the current system of SSV's has paid limited attention to the quantification of risks related to diffuse pollution. In most countries SSV's are basically suitable to detect areas where risks are such that effects in soil (or adjacent water bodies) are imminent and require action to prevent risks for human beings or the soil ecosystem.

A complicating factor also is that SSV's developed so far in EU member states vary widely (see also paragraph 2.2 Soil pollution and the need for SSVs in the EU: an overview). This is because countries used their own risk assessment approach and a range of acceptable risks. At EU level there is, at present no harmonized approach to detect risks beyond that at individual member states. This implies that risk levels, and the need to remediate soils, depends on the country itself.

In this chapter we will first discuss current SSV's used at MS level (2.2 Soil pollution and the need for SSVs) followed by a discussion on the background of risk assessment approaches underlying the derivation of risk-based SSV's (2.3 The principle of harmonized risk-based critical levels of contaminants in soil) and the application thereof at EU level. The final paragraph (2.4 Calculation of time required to remediate the soil where critical limits are exceeded) discusses the potential of phytoremediation considering factors that affect the duration of the process.

2.2 Soil pollution and the need for SSVs in the EU: an overview

A recent overview of contaminated sites in Europe was presented by JRC (Pérez & Eugenio, 2018). This assessment revealed that 65000 sites that have been remediated or are under aftercare, but an additional 650000 registered sites exist where polluting activities took or take place. For these corrective actions, or the need thereof has not yet been established (Pérez & Eugenio, 2018).

The awareness within Member States that soil pollution poses a potential threat to the environment has triggered the development of soil quality guidelines in many EU Member States. The general idea behind many of these quality standards is that when exceeded, the soil or the use of the soil (for agriculture, water quality etc.) is at risk. However, the approach and criteria used to derive soil quality standards varies strongly between countries. In addition, soil monitoring studies revealed that even without anthropogenic influence soil quality (i.e. levels of most heavy metals in soil) vary strongly within the EU depending on among others the parent material of soil. Most countries therefore identify soil quality standards at different levels ranging from what commonly is mentioned as 'background' value up to levels beyond which soils are considered polluted to such an extent that they cannot be used for normal purposes or pose a threat to human health. Such levels often are called intervention values which states that some action is needed to protect the ecosystem and humans from toxic effects metals and other contaminants pose.

A previous overview of such heavy metal thresholds in EU countries has been made with the goal of mapping polluted soils (Hirschmugl & Sobe, 2020). Used were the so-called threshold values which upon reviewing are soil screening values or background values cited in Amlinger et al (2004) and Carlon et al (2007). As discussed by Hirschmugl & Sobe (2020) the result of the match between current levels of contaminants in soil and such critical values is that large areas of Europe exceed these threshold values. This clearly leads to an incorrect estimate of the area of land that is needed for remediation either via phytoremediation or otherwise.

This is due to the concept and often confusing nomenclature of screening value or background values, which has been discussed in various reviews (Rodrigues *et al.*, 2009; Pinedo *et al.*, 2013; Antoniadis *et al.*, 2019). This concept is explained here in short.

In a number of EU member states various screening values are used which are used in a specific regulatory framework. For example in the Netherlands there a relative low soil quality standard has been derived (SQS1) below which levels in soil are not assumed to pose any risk to either man or the environment. This low soil screening value, SQS1, is, in the Netherlands based on the 95th percentile value of soil samples collected from 100 sites that are assumed not to be affected by specific anthropogenic activities. Due to the nature of these samples -i.e. not affected by specific forms of pollution- often the term 'background value' is used. This approach has been used in other countries as well but the name given to the resulting soil quality level varies among countries. Where it is called background value in the Netherlands, it is called Reference Generic Level (RGL) in Spain (xx), or Threshold Value in Italy (xx) and Finland (). Also the approach used to derive this quality standard referring to the 'non-risk level' of background values differs slightly between EU member states. (Reimann *et al.*, 2018).

At present such SQS1 values are used in most EU member states as summarized in a report from the BIOPLAT-EU project[2]. However, in view of risk assessment the use of such background or reference levels to identify soils that are at risk is questionable since the SQS1 commonly represents a level of contaminants in soil below which risks are absent. This, however, does not imply that soil where contaminants exceed such SQS1 levels automatically are at risk. This is partly due to the derivation of background values which are often not risk-based but rather represent levels of metals that are expected to be found in non- affected soils. In most cases soils where the background is exceeded are not at risk at all. Hence using a background (or reference value) as indicative for soils in need of remediation would lead to a gross overestimation of the area where action is required and above all would result in a selection of areas where the soil de facto is not at risk at all.

To overcome this and to identify levels of contaminants in soil beyond which these metals can pose a risk additional soil quality standard (here identified as SQS2) were derived above which risks are assumed and actions are necessary (Pinedo et al., 2013). As with background values, the nomenclature and approach used to derive SQS2 levels for contaminants differs per country. The first such comparison was compiled by Carlon (2007) and results for EU Member States are listed in Table for metals and in Table **Error! Reference source not found.** for selected organic micropollutants. For metals, an update was prepared recently by Baritz et al. (2021; Table) showing not only that some SQS2 values have changed during the last two decades but also that in many countries the level of SQS2 itself depends on land use and soil type.

[2] https://bioplat.eu/

Table 2-1 SQS2 soil quality standards for selected heavy metals (Carlon, 2007 all data in mg/kg).

Legend: Austria (AUT); Belgium Flanders (BE(F)); Belgium Bruxelles (BE(B)), Belgium Walloon (BE(W)); Czech Republic (CZE); Finland (FIN); Italy (ITA); Lithuania (LTU); Netherlands (NLD); Poland (POL); Slovakia (SVK); United Kingdom (UK); Denmark (DNK)

	AUT	BE(F)*	BE(B)	BE(W)	CZE	FIN	ITA	LTU	NLD	POL	SVK	UK	DNK
As	50	110	110	300	70	50	20	10	55	22.5	50	20	20
Ba					1000			600	625	285	2000		
Be					20		2	10	30		30		
Cd	10	6	6	30	20	10	2	3	12	5.5	20	2	5
Co					300	100	20	30	240	45	300		
Cr	250		300	520	500	200	150	100	380	170	800	130	1000
Cu	600	400	400	290	600	150	120	100	190	100	500		1000
Hg	10	15	15	56	10	2	1	1.5	10	4	10	8	3
Pb	500	700	700	700	300	200	100	100	530	150	600	450	400
Mo					100			5	200	25	200		
Ni	140	470	470	300	250	100	120	75	210	75	500		30
Sb	5				40	10	10	10	15				
Se							3	5	100		20	35	
Sn					300		1	10	900	40	300		
Te									600				
Tl	10						1		15				
V					450	150	90	150	250		500		
Zn		1000	1000	710	2500	250	150	300	720	325	3000		1000

*For new contaminants only

Table 2-2 SQS2 soil quality standards for selected organic micropollutants (Carlon, 2007; all data in mg/kg unless specified otherwise).

Legend: Austria (AUT); Belgium Flanders (BE(F)); Belgium Bruxelles (BE(B)), Belgium Walloon (BE(W)); Czech Republic (CZE); Finland (FIN); Italy (ITA); Lithuania (LTU); Netherlands (NLD); Poland (POL); Spain (ESP); Slovakia (SVK); Sweden (SWE); United Kingdom (UK) for human health; Denmark (DNK)

	AUT	BE(F)*	BE(B)	BE(W)	CZE	FIN	ITA	LTU	NLD	POL	ESP	UK	DNK
Benzene		0.5	0.5	0.4	0.8	0.2	0.1	0.5	1	12.6	1		
Ethylbenzene		5	5	28	50	10	0.5	5	50	38	20	41	
Toluene		15	15	33	100	5	0.5	0.1	130	38	30		
Xylene		15	15	10	30	10	0.5	0.1	25	18	100		
Naphtalene		5	5		60	5	5	5		12.5	8		
Anthracene		70	70		60	5	5	5		12.5	100		
Benzo(a)anthracene		10.5	10.5	5	5	5	0.5			12.5	2		
Benzo(g,h,i)perylene		3920	3920	15	30		0.1			10			
PAHs (total)	50					30	10	5	40	30			40
Dichloromethane		0.35	0.35			1	0.1	2	10		6		
Trichloroethylene		1.4	1.4			1	1	2	60		7		
Tetrachloromethane		0.02	0.02				0.1	1	1		0.5		
Hexachlorobenzene		0.1	0.1			0.05	0.05	0.5			0.1		
Phenol							1	10	40	10.25	70	280	
Cresols (sum)							0.1		5	10.25	40		
Atrazine (p)						1	0.01		6	3			
DDT (sum DDT, DDE & DDD)						1			4	2.01			
PCB	1		0.9		5	0.5	0	0.1	1	0.55	0.08		
methyl t-butyl ether		9	9			5	10		100				
1,1,1-trichloroethane		13	13				0.5		15				
Chlorobenzenes (total)									30	1.05			
Benzo(a)pyrene	5	1.5	1.5	4.4	2	2	0.1	0.1		7.5	0.2		
PCDD/PCDF (in ng I-TEQ TeCdd/g)	100				0.5	0.0001	1.00E-05		0.001				

*For new contaminants only

Table 2-3 Overview of SQS2 values for Cd, Cu, Pb and Zn (mg/kg).

Geographical region	Cadmium (Cd)				Copper (Cu)			
	Intermediate risk		Critical risk		Intermediate risk		Critical risk	
	Stratification	SV	Stratification	SV	Stratification	SV	Stratification	SV
Albania_Tirana		0,7				36,3		
Austria	*Land use*	1 - 40		10	*Land use*	100-1500		600
Belgium_Brus		1	*Land use*	2-30		40	*Land use*	145-800
Belgium_Fland			*Land use*	2-30			*Land use*	200-800
Brussels_Wall.	*Land use*	1-10	*Land use*	10-50	*Land use*	40-120	*Land use*	80-500
Bulgaria	*pH*	0.04 - 3			*pH*	15-280		
Czech Republic		10	*Land use and Texture*	0.4-30		500	*Land use and Texture*	60-1500
Denmark		5		5		500		1000
Finland		1	*Land use*	10-20		100	*Land use*	150-200
Germany		20						
Hungary		1		10		75		1000
Ireland				1				
Italy			*Land use*	1.5-15			*Land use*	100-600
Lithuania				3				100
Netherlands	*Land use and Texture*	1-10	*Land use*	12-13	*Land use and Texture*	40-200		190
Poland			*Land use, Saturated hydraulic conductivity and Soil depth*	1-20			*Land use, Saturated hydraulic conductivity and Soil depth*	30-1000
Slovakia	*Land use*	0.1 - 5		20	*Land use*	1-100		500
Slovenia		2		12		100		300
Sweden	*Land use*	0.4 - 12		4	*Land use*	100-300		1000
United Kingdom			*Land use*	2-1400				500

Geographical region	Lead (Pb)				Zinc (Zn)			
	Intermediate risk		Critical risk		Intermediate risk		Critical risk	
	Stratification	SV	Stratification	SV	Stratification	SV	Stratification	SV
Albania_Tirana		85,5				151		
Austria	*Land use*	100-300		500		300		
Belgium_Brus		120	*Land use*	200-2500		120	*Land use*	300-3000
Belgium_Fland			*Land use*	200-2500			*Land use*	600-3000
Brussels_Wall.	*Land use*	80-385	*Land use*	170-360	*Land use*	120-320	*Land use*	215-1300
Bulgaria	*pH*	20-80			*pH*	20-370		
Czech Republic		250	*Land use and Texture*	100-800		1500	*Land use and Texture*	130-5000
Denmark		40		400		500		1000
Finland		60	*Land use*	200-750		200	*Land use*	250-400
Germany		400						
Hungary		100		750		200		2500
Ireland								
Italy			*Land use*	100-1000			*Land use*	150-1500
Lithuania				100				300
Netherlands	*Land use and Texture*	15-590		530	*Land use and Texture*	150-720		720
Poland			*Land use, Saturated hydraulic conductivity and Soil depth*	50-1000			*Land use, Saturated hydraulic conductivity and Soil depth*	100-3000
Slovakia		150		600	*Land use*	2-500		3000
Slovenia		100		530		300		720
Sweden	*Land use*	80-300		800	*Land use*	350-1050		3500
United Kingdom			*Land use*	450-750				

The concept of the second soil quality standard, SQS2, or various standards, is used in a limited number of European countries (Austria, Belgium, Germany, Denmark, Spain, Finland, Italy, Lithuania, Netherlands, Sweden, United Kingdom), and is often based on some form of model approach to identify the critical level in soil beyond which the target to be protected is at risk. (Carlon, 2007; Swartjes *et al.*, 2009).

In other countries the concept of SQS2 is not used (France, Ireland). The SQS2 is potentially relevant to for sanitation by phytoremediation as this value declares a soil as contaminated and actions to be taken.

In most if not all cases, the SQS2 level exceeds the SQS1 level and this often can lead to confusion as to how to assess the impact of degrees of contamination in soils between the SQS1 and SQS2 level. Whether or not soils that contain contaminants between the SQS1 and SQS 2 level need further remediation depends among others on the land use, the soil type and the risk considered.

Often in-between target values have been derived that identify levels below which a specific function is believed to be feasible, i.e. there is no risk anymore. These SQS levels are identified here as SQS_{LU} (LU: Land Use) which indicates that the level of acceptable risk is linked to the land use that is foreseen in a specific area that needs to be treated. Typical for SQSLU levels is that these become stricter (i.e. the value of the SQS decreases) when contact with soil increases. Typically, SQS-LU for industry are highest (least strict) based on the assumption that contact between the polluted soil and humans is limited. Also, in case of industry the impact on the soil ecosystem is allowed to be larger than for example in case of other forms of land use such as areas intended for housing, agriculture or recreation. In the NL the order of SQS-LU typically increases from SQS-LU levels for agriculture and nature (one SQS-LU), housing, recreational areas to industry.

Such SQS-LU therefore appear suitable as criteria to identify areas that need remediation. At present the system of SQS_{LU} are used in the Netherlands, Germany, United Kingdom, and Belgium. An overview is given in Table . In Figure 3 the conceptual relation between the metal concentration in soil and the level of SQS1, 2 and LU is shown.

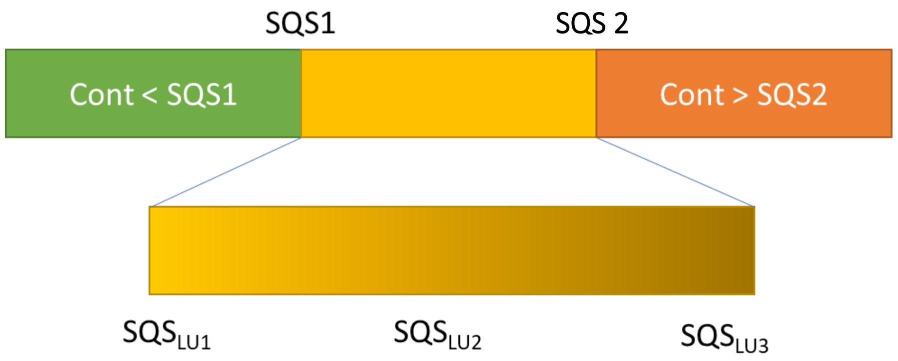

Figure 3 Conceptual relation between the contaminant concentration in soil and corresponding soil quality standards.

According to this scheme, soils with contaminations below the level of SQS1 are considered to pose no risk and are suitable to be used for any kind of land use. On the other hand, soils with levels in excess of SQS2 are in need of treatment to reduce a specified risk (which can differ per country) for man or the environment. The target level for this remediation (SQSLU) depends on the soil type and land use. In many cases the SQSLU for an area to be used for agriculture or nature will be close to the level of SQS1 to avoid any risk (here marked as SQSLU1. On the other hand, the SQSLU for an area designated to become an industrial area can be close to (but not exceed) the SQS2 level (here marked as SQSLU3). In between levels of SQSLU can refer to other forms of land use (e.g. recreational areas or housing). Between Member States both the number of and the absolute levels of such SQS-LU standards are variable. In the NL for example three levels of SQSLU are derived, one for allotments, nature, and arable land (SQSLU1), the level of which is equal to the background value. The second level SQSLU is derived for housing areas, recreational areas, and other 'green' spaces with designated natural values (SQSLU2) whereas the SQSLU3 has been derived for industry, sealed land with construction and other green areas where contact between humans and soil is limited. Boundaries for SQSLU1 is the background value, SQSLU1 cannot be lower than the background value where on the other hand the SQSLU3 cannot exceed SQS2 since this value is the upper limit of acceptable soil quality.

Depending on the choices made in Member State, the number and level of SQS_{LU} is even more variable than those for SQS 1 or SQS2 standards. This is partly due to differences in soil properties that are considered in the derivation of SQS_{LU} standards, ranging from no correction to correction for soil pH, organic matter and/or texture.

2.3 The principle of harmonized risk-based critical levels of contaminants in soil

As shown in the previous section, criteria to derive SQS in soil differ between countries. As a result, corresponding SQS levels also vary widely between countries. To decide which areas are in need of remediation it can, therefore, be advantageous to use a single risk-based approach. This would avoid having to use country specific risk limits. The rationale behind the risk based approach is that one or more critical limits in so-called end-points are used as target not to be exceeded.

Here, end-points refer to the target that needs to be protected. Such targets include the quality of food, the quality of soil itself, e.g. in view of ecosystem health, or the quality of ground- or surface water that are affected by soil. For several of these end-points EU-wide legal limits are in place. This is the case for example for Cd (EU2021/1323[3]), or Pb (EU2021/1317[4]) in food or surface water concentrations for Zn and Cu (EC 2000/60). For others, like ecosystem health, critical limits in soil solution for metals have been derived that can be converted to corresponding critical limits in soil (Lofts et al., 2004).

To apply this approach, it is imperative that there is a relation (model) between the level of contaminant in the target (food, water, soil solution) and the concentration in soil itself. Often this requires a number of selected soil properties (e.g. pH) to translate the critical limit in a specific endpoint to a level in soil. This relation is then to be used to convert the critical limit in the target to a corresponding critical limit in soil as shown by de Vries et al. (2007). This approach is schematically illustrated in Figure 4.

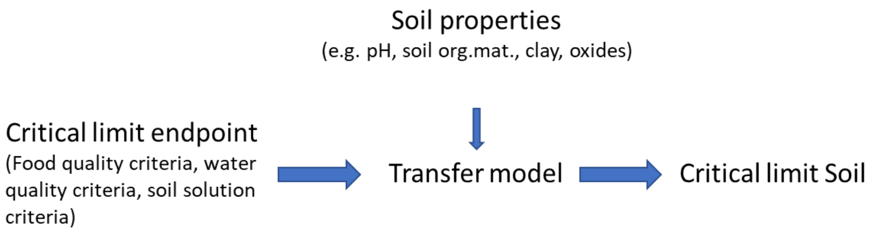

Figure 4 Schematic representation of the derivation of risk-based critical limits in soil using specific critical limits in endpoints.

3 Commission Regulation (EU) 2021/1323 of 10 August 2021 amending Regulation (EC) No 1881/2006 as regards maximum levels of cadmium in certain foodstuffs

4 Commission Regulation (EU) 2021/1317 of 9 August 2021 amending Regulation (EC) No 1881/2006 as regards maximum levels of lead in certain foodstuffs

Once a critical limit for a specific area is calculated, the comparison with the actual concentration in soil reveals whether the endpoint chosen is at risk. To apply this approach on a regional scale the following data and models are needed:

1. Critical limits in endpoints. At present a limited number of critical limits in endpoints (food, water, ecosystem) have been derived or are actually in use in current legislation. Food quality criteria as well as water quality standards are already implemented in several EU Directives such as Commission Regulation (EU) 2021/1323 that regulates levels of Cd in food products (EU, 2021) or the Water Framework Directive (WFD; 2000/60/EC) that includes limits for Cu and Zn (EC, 2000). For other endpoints, such as the protection of soil micro-organisms, no-effect levels in soil solution have been derived from data (e.g. Lofts et al., 2004).
2. All relevant soil properties used in the transfer model to relate the critical endpoint to a corresponding level in soil. At present soil data collected at EU level, e.g. as present in the LUCAS database (Orgiazzi et al., 2018) include a number of essential properties such as pH, organic matter, clay and/or oxide content of the soil.
3. The transfer model itself. At present, transfer models that can relate a critical concentration in food, water or soil solution to soil are scarce. Existing models include those to predict the Cd and Zn concentration in several arable crops and models to predict soil solution concentrations. The latter can be used to convert critical concentrations in solution that have been derived to protect soil micro-organisms to corresponding soil concentrations. For many contaminants, notably organic pollutants like PAH's or emerging pollutants like PFOS and related substances, such models have not been derived or have been derived for a limited number of soils itself.
4. To derive a SQS related to ground water or surface water quality, additional hydrological models as well as information on the composition of the soil profile below the surface layer are needed. During the flow from topsoil to deeper soil layers and water bodies concentrations usually decrease substantially due to sorption and it is therefore not realistic to relate the quality of the topsoil and the water quality of this layer directly to surface or groundwater quality standards.

So far, the approach as outlined above has been applied at EU level only for food quality criteria for Cd and ecosystem health for Cd, Cu, Pb and Zn. For food quality, both soil pH, organic matter and clay are used whereas the model to derive the critical soil concentration for ecotoxicity considers pH and organic matter only (Figure 5 from Trombetti et al., 2022).

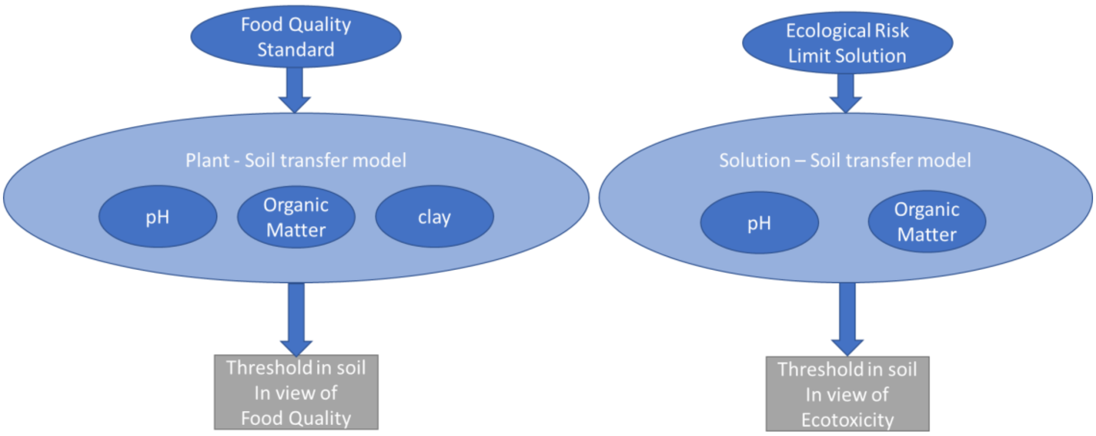

Figure 5 Approach to calculate risk limits in soil in view of food safety (left) or ecotoxicity (right).

Once the critical limit in soil has been established, a comparison with the actual concentration in soil will reveal whether an area is at risk. If the current concentration exceeds the calculated critical concentration in soil, it is likely that the endpoint considered is not sufficiently protected. This could mean that Cd in food products will exceed the legal limit or that the functioning of soil organisms is affected.

One of the key aspects of this approach is that differences in soil properties will lead to a variation in the critical metal concentration. This means that the metal concentration in soil alone is not the only parameter of relevance. For most risks considered here for example, the critical limit decreases (i.e. changes towards lower acceptable levels) with a decrease in pH. This can lead to a situation where, in areas with low pH and/or organic matter levels in soil, the critical limit can be exceeded even at a low current level in soil.

Details of the approach used here to delineate areas where critical limits are exceeded are given in chapter 3.

2.4 Calculation of time required to remediate the soil where critical limits are exceeded

In those areas where the current concentration in soil exceeds the critical levels, the amount of pollutants that needs to be removed from soil can be calculated by subtracting the critical level from the current level. Based on experimental data on the annual removal rate of metals via phytoremediation, the time required to achieve the minimum quality (i.e. Cd concentration in soil) then can be calculated.

A crucial aspect to establish the applicability of phytoremediation is the time required to remediate the soil to a target concentration. Other than the typical soil remediation approach where polluted soil is replaced by clean soil, phytoremediation will cause a gradual decrease of the pollutant level in soil. This removal rate depends on several factors including the uptake rate of the crop, the biomass production, the availability of the contaminant in the soil itself and the, often, non-linear response in crop uptake to lower pollutant levels in soil. This will be explained in more detail below.

Impact of soil properties on availability of pollutants in the soil

Since uptake of contaminants like metals occurs via roots, metals need to be in the soil solution before they can be transferred to the root and into the shoot. The concentration in the soil solution heavily depends on soil properties like pH and organic matter and solution concentrations for most cationic metals tend to decrease strongly with an increase in both pH, soil organic matter and clay content. This already suggests that metal concentration in soil solution tend to be very low in soils with a neutral to alkaline pH and high clay content like the ones that are commonly found in Mediterranean countries. On the other hand, the solution concentration can be high in acidic sandy soils that occur in Central and Northern parts of the EU.

Impact of crop type and biomass used to extract pollutants from soil

To reduce the time required to remove pollutants from soil, crops with a high biomass production ánd high uptake rates are preferred. Unfortunately, this combination is rather rare and crops that tend to accumulate large amounts of, for example, metals (hyperaccumulators) tend to have a low (typically less that 1 -5 tons per hectare per year, e.g. brassica species) biomass production or are relatively slow growing (e.g. willows). Further understanding of this will of course be generated in the other activities in this GOLD project.

Non-linear response of uptake at lower pollutant concentrations

For pollutants like metals a clear non-linear relationship has been established in the relation between concentration in soil and crops. Often non-linear Freundlich equations are used to relate the concentration in crops or water to that in soil. This means that the efficiency of the annual removal rate from soil decreases upon a reduction of the metal concentration in soil as was documented by Koopmans et al., 2008).

Considering the three aspects mentioned above (availability, biomass production and non-linear response) a model was developed to predict phytoremediation times for well-known sites affected by proximity pollution (Koopmans et al., 2008) for a soil located in the Netherlands as well as by Liang et al. (2009) for a site in Taiwan. In both cases a strong increase of the remediation time was calculated based on difference between crops and degree of pollution (Figure 6) or the biomass production (Figure 7).

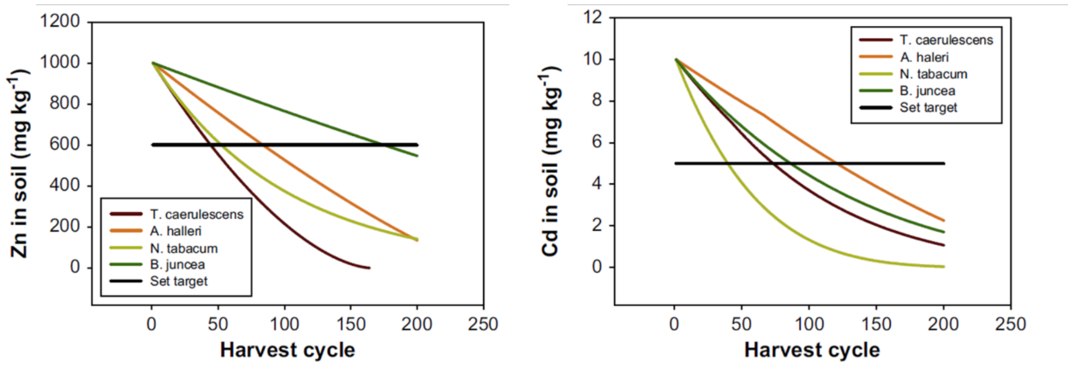

Figure 6 Effect of crop type and degree of pollution on the number of harvest cycles needed to reach the target concentration in soil (from: Liang et al., 2009)

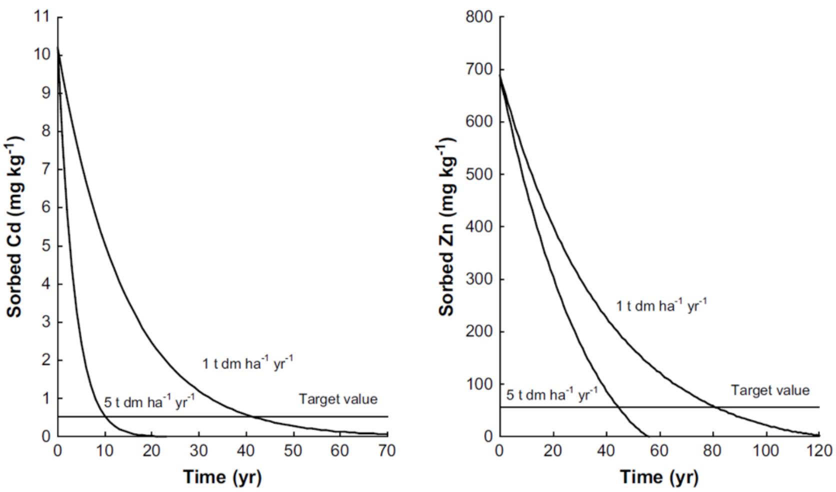

Figure 7 Effect of biomass production (Thlaspi caerulescens) on the time needed to reduce the level of Cd (left) or Zn (right) to the level of the Dutch target value for arable soils.

All results show that remediations times to clean soils to a selected safe (target) value are long and range from, in this cases 10 years to 80 years for the Dutch soil and between 50 and 150 harvest cycles for the Taiwan soil. It is interesting to note that even the target value for Cd ranged from 0.5 mg kg^{-1} for the Dutch soil to 5 mg kg^{-1} for the Taiwan soil. Both examples show nevertheless that extraction efficiency decreases with time which is largely due to the non-linear response of crops to a decrease in the availability in soil.

2.5 In summary

Diffuse pollution is ongoing in the EU and has already caused widespread emission of a range in pollutants including nutrients, organic pollutants and metals. Effects of diffuse pollution on water quality are well documented and effects of proximity pollution are known in areas in various member states. At present however current SSVs are targeting point source pollution mostly whereas soils affected by diffuse pollution often do not exceed such SSVs. This does not imply that diffuse soil pollution poses no risk to the soil ecosystem or quality of food and fodder. A direct assessment of the current soil quality as affected by diffuse pollution is however not possible since SSVs currently in use are specific for individual member states. At EU level there is currently no agreed uniform screening level that can be used as a first approximation to allocate areas that need remediation.

Therefore, in this project we propose a risk assessment model instead that is applied based on specific risks in view of ecosystem health, food quality and water quality. This approach assumes that there is a connection between soil quality as expressed by relevant soil properties (for metals based on pH, organic matter, and clay) and the acceptable pollutant concentration at which the risk for either food, water or ecosystem is avoided. The resulting regional critical concentrations in soil can be compared with actual concentrations to detect areas at risk. The feasibility of remediation using phytoremediation then can be assessed considering the time frame that is considered acceptable. Remediation times, however, strongly depend on both initial pollutant levels, soil properties of the targeted area, biomass production. Also, the efficiency of phytoremediation typically decreases with time and expected remediation duration times can vary from 10 years to several decades for the examples tested so far.

3 Calculation of risk-based soil screening values to detect areas at risk of diffuse pollution

3.1 Introduction

To determine where, at present, soil quality is such that effects on either food quality, water quality or ecosystem health can be expected a risk assessment approach has been developed (de Vries et al., 2022). This was done since SSVs by individual MSs cannot be applied beyond the country itself due to country specific assumptions related to soil type, land use or risk considered.

The basic principle (see also section 2.3 The principle of harmonized risk-based critical levels of contaminants in soil*)*

Models are used to connect critical limits in endpoints (food, water or soil dwelling organisms) to a critical metal concentration in soil (Figure 8). Here endpoints are those environmental compartments that need to be protected. This means that concentrations of pollutants in food, water or soil organisms should remain below an agreed upon critical limit. Examples of such critical limits in endpoints include food quality criteria, water quality standards or critical solution concentrations related to toxicity. Food quality criteria for example are set for a range in pollutants including metals such as cadmium (Cd), lead (Pb), arsenic (As) and mercury (Hg). For As and Hg these refer largely to fish and other food products from aquatic environments but for Pb and Cd these include a large range of food products such as wheat, vegetables, fruit etc. (EU 2021/1323) Water quality criteria are in place to protect drinking water (Council Directive 98/83 EC) as well as surface water quality in view of aquatic ecosystem protection (WFD EC2000/60). For soil organisms, critical concentrations in solution have been derived below which the risk of adverse impacts of pollutants on microorganism functioning is deemed minimal (Lofts et al., 2004). Here we show how each of these three risk limit (for food, water and ecosystem health) can be used to derive critical limits in soil. For both food, water and soil solution, soil properties including pH, organic matter or clay are relevant since they affect the transfer from soil into water and food.

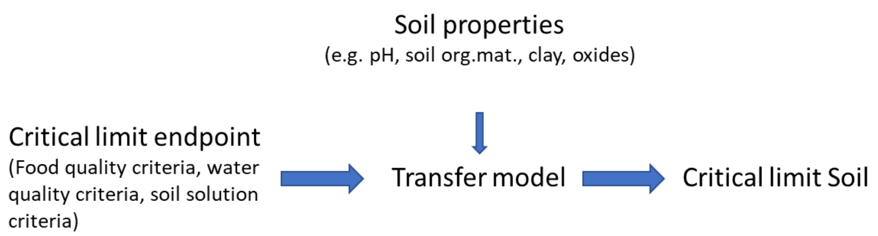

Figure 8 Schematic representation of the derivation of risk-based critical limits for metals in soil using specific critical limits in endpoints

Once a critical concentration in soil has been derived at the appropriate scale level using representative soil properties for that area, a match between the actual (measured) concentrations and the critical concentrations then reveals whether an areas is at risk. Once it has been established that the actual concentration exceeds the critical concentration the amount of pollutants that has to be removed can be calculated. This amount is equal to the difference between the actual and the critical concentration multiplied by the depth of the layer and bulk density. This yields the amount in gram/ha to be remediated. This pool to be removed then can be used as input for the phytoremediation models to calculate how long removal of the excess amount of pollutants would take. As will be illustrated in the examples, this time required also depends on soil type and crop type (biomass/uptake rates).

In order to spatially map areas of actual, critical and exceedance maps (i.e. maps showing only these areas where actual concentration exceed critical concentrations), several databases and models are needed. Here we briefly summarize those applied here. In this chapter we illustrate this approach for metals which require the following data and models:

1. Database with current metal concentrations. At the moment only one large representative database with measurements at point level is available. This is the GEMAS database (n=approx. 4000, Reimann et al., 2014). The model currently used is therefor based on the GEMAS data and point data are upscaled to spatial grids of 1x 1 km. The point data in the LUCAS database (n= approx.22000; Orgiazzi et al., 2018) are, at the time of writing, not publicly available and will not be used in this study other than the maps prepared by JRC from these data.
2. Database with soil properties including pH, organic carbon and texture (clay content) available in both the LUCAS (Aksoy et al., 2016; Orgiazzi et al., 2018, Panagos et al., 2022) and SOILGRIDS database.
3. Regression relationships to relate soil solution concentrations for metals to a corresponding concentration in the solid phase (available for most metals) (see Table).
4. Critical limits in food (for Cd and Pb) or fodder (for Zn and Cu), water (drinking water criteria) or toxicity (note: the toxic concentration in soil solution can be calculated using pH and SOM based on Lofts et al., 2004; this concentration then can be recalculated to a corresponding level in the soil as summarized in section 2.3, based on de Vries et al., 2007).

This will yield the areas where the critical concentration exceeds the actual one as well as the total pool of metals to be removed to reduce the metal concentration to the critical one (or below).

To derive approximations of the remediation time, models are needed (to be supplied by other GOLD project partners) that describe the uptake of metals by the crops tested as well as expected biomass production. These parameters then are used to calculate the time needed to reduce the metal concentration in soil. To do this we also need information on water dynamics (how much water is leached each year) to account for leaching losses. Ideally you need all inputs and outputs to the soil (inputs via fertilizer, manure etc.). This can be done in INTEGRATOR since this information is all in there.

Now this approach is limited to a few selected metals for which we have the requested information (see Table).

For food this approach is, for now, limited to Cd since, for Pb the relation between soil and crops is poor so we cannot predict at what levels in soil food quality criteria are exceeded (see Table). For other metals specific health-based quality criteria only exist for mercury (Hg) and arsenic (As) but this refers almost exclusively to products of marine or fresh water organisms (fish, mollusks etc.). Also, there is growing concern about potential food safety issues related to organic emerging pollutants including pesticides, antibiotics and flame retardants (a.o. PFAS). For most of these substances (with the exception of plant protection chemicals or metabolites thereof) food quality criteria are not available. Also, there is a lack of reliable soil to crop transfer models (see Table). Finally, also spatial data to prepare maps at the desired scale level are lacking since, as of now, there is no systematic monitoring of most of these substances.

Table 3-1 Overview of availability of data, transfer models and critical limits for different pollutant groups

Compound	Data availability EU level (soil)	Availability transfer models	Relevant critical limit	Application level
Metals	Moderate to good (GEMAS, 4000 samples) plus raster maps derived from LUCAS data	Sufficient (Cd, Zn) Moderate (Cu, Pb)	Food (Cd, Pb) Water (Cu, Zn) Ecotox (Cd, Cu, Pb, Zn)	Regional
Metals	Moderate to good (GEMAS, 4000 samples) plus raster maps derived from LUCAS data	Moderate-Limited: Limited for food Moderate for water, Limited for ecotox	Water (other metals: Sb, As, Co, Cr, Ni, V)	Regional
Organic chemicals (PAH's, pesticides)	Poor, no national coverage	Very limited (water) to absent (poor relationships)	Limited in soil for ecotox. Generic standards soil protection; ecotox in water	Local
Upcoming chemicals (PFAS, nanoparticles)	Very poor, initial stage	Under development based on laboratory experiments	Limited (water, generic standards soil protection)	Local

For ecology the approach is now applicable for Cd, Cu, Pb and Zn (Lofts et al., 2004; de Vries et al., 2007). The principles applied can be used for other metals of interest as well since for most metals the dose-response curves have been derived for a large range of organisms. However, data at EU level to prepare maps for other metals (a.o. As, and metals like Co, Mo, Se, U or V) are not yet publicly available.

For water we can do this for all relevant metals based on including drinking water criteria. However, this is a worst-case calculation since it assumes that the water concentration leaving the topsoil is the same as the concentration in groundwater. This is almost never the case since metals will be retained during vertical transport through the soil column which means that the concentration that reaches the groundwater invariably will be (much) lower than that in the topsoil.

3.2 Critical soil concentrations in view of Food Safety

Here we use wheat as the key crop to be considered. This is because wheat is a staple crop that is grown in large parts of the EU. In addition, Cd is taken up rather easily by wheat (next to leafy vegetables) compared to most other crops which means it is a suitable crop to be used as indicator. The critical limit in wheat, here we use the WHO food quality criteria, is converted directly to critical concentration in soil. The resulting calculated value is expressed in mg/kg soil and directly comparable to measured values in Aqua Regia. The relationship between soil properties and Cd in wheat is based on a database used by Römkens et al. (2007) and contains field data only (measured concentrations in soil and corresponding Cd concentrations in grains grown at these soils)

Model assumptions:

Cd limit in food	0.10 mg/kg fresh weight for wheat (note: critical values range from 0.05 for rye and barley to 0.2 for wheat germ ((EU) 2021/1323)
Dry weight wheat:	0.85
$Cd_{wheat\text{-}crit}$:	0.12 mg/kg dw

Relation Cd in wheat and soil:

$\log(Cd_{wheat\text{-}grain}) =$ $0.22 - 0.12 \cdot pH_{KCL} - 0.33 \cdot \log(SOM) - 0.04 \log(clay) + 0.62 \cdot \log(Cd_{soil})$

Using the critical concentration of 0.12 mg/kg a corresponding critical Cd concentration in soil can be calculated:

$\mathrm{Log}(Cd_{crit\ food\ safety\text{-}soil})$: $\{\log(Cd_{wheat\text{-}crit}) - (0.22 - 0.12 \cdot pH_{KCL} - 0.33 \cdot \log(SOM) - 0.04 \log(clay))\}/0.62$

Where Cdsoil is expressed in mg/kg dw, SOM and clay in % and Cd wheat in mg/kg dw.

3.3 Critical soil concentrations in view Water Quality

For the protection of water quality relevant threshold criteria for a range of heavy metals including Cd are in place. These include EU-wide standards for both drinking water and ecology. For drinking water criteria are set both by the EU (EU2020/2184) as well as the WHO. These range from 3 µg L^{-1} (WHO) or 5 µg L^{-1} (EU). Assuming that these concentrations are not to be exceeded in water that leaches from the soil, a corresponding critical concentration for soil can be calculated. However, values of 3 to 5 µg L^{-1} for Cd in soil solutions are rather high and such values would therefore lead to rather higher corresponding critical limits in soil. In addition, the pathway from soil to groundwater used for drinking water extraction usually is long and concentrations in the upper part of the soil are not representative for the final concentration in groundwater.

Next to criteria for drinking water, also critical concentrations in surface water related to ecological risks can be used to calculate corresponding critical concentrations in soil. For Cd such criteria have been derived for surface water bodies. For inland surface waters Cd is considered a priority toxic substance and AA (Annual Average) and MAC (Maximum Allowable Concentrations) values have been derived (EC2008/105). Depending on the class (1 to 5) AA values range from < 0.08 µg/L tot 0.25 µg/L whereas MAC values range from < 0.45 to 1.5 µg/L. Other than for groundwater the pathway from soils to surface waters can be rather short, especially in areas with high groundwater tables and or surface run-off. Using critical concentrations in surface waters to derive corresponding critical levels in soil therefor seems more plausible than using critical concentrations based on drinking water standards.

The critical concentration in soil as related to a critical concentration in water is derived directly using the relation between Cd in solution and soil (Römkens and Smolders, 2018):

$\text{Log(Cd-solution)} = 3.655 - 0.713*\log(\text{SOM}) - 0.48*\text{pH}_{\text{CaCL2}} + 1.116\log(\text{Cd}_{\text{soil-total}})$

With Cd-solution in µg L^{-1}; SOM in % and $\text{Cd}_{\text{soil-total}}$ in mg/kg

Hence the critical Cd concentration in soil ($\text{Me}_{\text{soil-crit-W}}$) based on the water criteria applied ($\text{Cd}_{\text{crit. W}}$) equals:

$\text{Log}(\text{Me}_{\text{soil-crit-W}}) = \{\log(\text{Cd}_{\text{crit. W}}) - (3.655 - 0.713*\log(\text{SOM}) - 0.48*\text{pH}_{\text{CaCL2}})\}/1.116$

Where, for $\text{Cd}_{\text{crit. W}}$ either the critical concentration in surface waters or the drinking water standard can be used.

NOTE: *the critical soil concentration thus calculated is a worst case since it implies that water from the topsoil is in equilibrium with that of groundwater or surface water. Especially in case of groundwater, retention of pollutants in deeper soil layers will cause a substantial removal from solution thus leading to much lower concentrations in groundwater. Critical concentrations in soil derived via this approach therefore tend to be overprotective compared to those derived in view of food safety (par 3.2) and ecology (par 3.4).*

An approach that would consider the actual displacement of chemicals through soil, requires, however, much more information (data) as well as models (hydrology) to accurately predict the vertical displacement of pollutants through soil. This requires not only detailed profile description of both pollutants and soil properties but advanced hydrological input as well to characterize the flow of water through soil towards ground- and surface water systems. At present this information is not available at the European level but has been developed and applied at country level for the Netherlands (Van der Bolt and Römkens, 2022). Calculations for 14 metals reveal that despite considerable inputs to the soil via manure and other sources, drinking water criteria in upper groundwater are, at present, not exceeded. Due to the relatively short pathways in specific areas with high groundwater tables, ecological risk limits in surface waters can be exceeded at a regional level. This is because especially in the Netherlands part of the water present in soil is in short contact to surface and groundwater.

For Cd the results of the combined geochemical-hydrological model are shown in Figure 9**Error! Reference source not found.**; here average concentrations are shown.

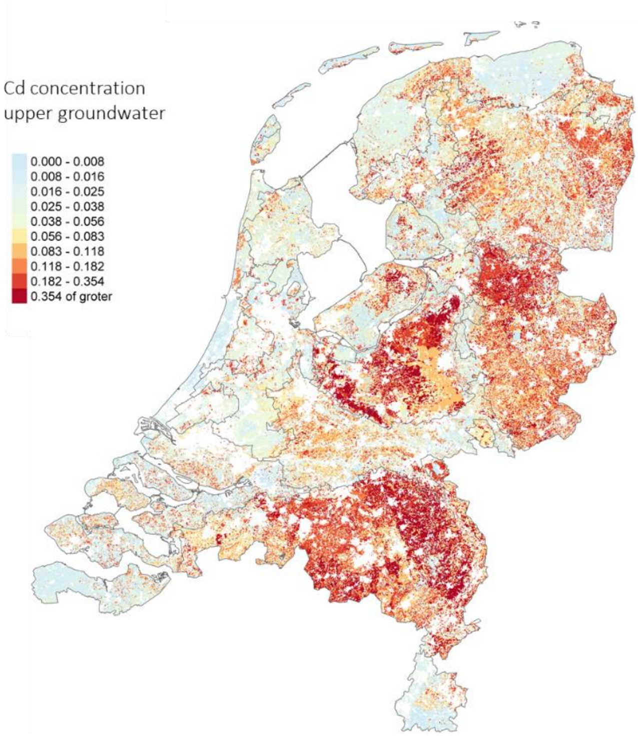

Figure 9 Predicted Cd concentrations in upper groundwater (Van der Bolt and Römkens, 2022).

Results in Error! Reference source not found. clearly illustrate the dominant role of soil properties, in this case pH and, to a lesser extent organic matter. Especially in soils from natural areas (forest) and arable sandy soils, characterized by low pH levels, predicted Cd concentrations exceed the AA concentrations at a large scale.

Other than for food safety and ecology, back-calculation to critical Cd concentrations in soil is difficult if not impossible since the resulting concentration in groundwater is a combination of the impact of soil properties, inputs to soil and hydrology. This means there is a number of variables that affects the relationship between pollutants predicted in groundwater and corresponding levels in the topsoil. The approach used here therefore only can be used in a forward calculation mode, i.e. based on current conditions, expected concentrations in water can be calculated and compared with critical limits in groundwater. This does not however automatically imply that the topsoil concentration of Cd is the main cause of the exceedance of groundwater concentrations.

3.4 Critical soil concentrations in view of potential effects on the soil ecosystem

Since soil dwelling organisms are in direct contact with the soil solution in the topsoil, it is possible to calculate a critical concentration in the soil based on critical concentrations in the soil solution in view of ecotoxicity. This approach involves three consecutive calculations as summarized below. More details can be found in Lofts et al. (2004) and de Vries et al. (2007). The main assumption is that critical concentrations in soil are based on an equivalent toxic amount that is added to the soil to provoke an effect. First the critical concentrations in solution (NOEC levels) are converted to a corresponding adsorbed concentration. These NOEC levels are based on laboratory experiments for many test organisms and the 5 percentile of all test results is chosen as the relevant protection level which implies that below this concentration, 95% of all species are protected. The conversion of the $NOEC_5$ concentration to a corresponding adsorbed concentration in soil depends on pH and organic matter and is based on a Freundlich type equilibrium between solution and soil solid phase. The outcome of this step is the amount of the critical added amount of Cd adsorbed in soil. This needs to be corrected for the presence of non-reactive Cd and the sum of reactive and non-reactive is called the total-added amount (step 2). To compare this value to real field soil samples a background concentration needs to be added to obtain the total critical Cd concentration in soil (step 3)

This yields the total critical metal concentration in soil that can be compared to the total measured concentration at grid cell level (1x1 km upscaled value). Here we briefly summarize the models used in step 1, 2 and 3.

Step 1: calculation of critical reactive added concentration in soil.

Critical concentrations in soil added to soil at which the critical $NOEC_5$ concentration is exceeded can de directly calculated from organic matter and pH.

Log $M_{re,add(crit)} = b_0 + b_1 \cdot \log SOM + b_2 \cdot pH\text{-}H_2O$

With Me in mg/kg, SOM in % and pH expressed as pH H_2O; values of b_0, b_1, and b_2 for Cd, Cu, Pb and Zn are shown in

Table (from de Vries et al., 2007)

Table 3-2 Coefficients used to derive the critical added metal concentration in soil (from de Vries et al., 2007)

Metal	b_0 (-)	b_1 SOM (%)	b_2 pH-H_2O (-)
Cadmium	-2.27	1.00	0.33
Lead	0.58	0.66	0.11
Copper	0.26	0.68	0.02
Zinc	-0.74	1.07	0.14

Step 2: Conversion of critical reactive added to critical total added

For most if not all metals, part of the pool in the soil solid phase is considered geochemically inert. This includes metals present in clay or oxide minerals that do not take part in the geochemical equilibrium between the solution and the solid phase (sorption). This fraction has been experimentally measured in a reference database (Römkens et al., 2004; Groenenberg et al., 2017) and the amount in the inert fraction correlates well to the measured reactive fraction (as predicted in step 1 here) and soil properties according to:

log $M_{tot,add(crit)}$ = (log $M_{re,add(crit}$ $- c_0 - c_2 \cdot \log SOM - c_3 \cdot \log clay)/c_1$

here

with Mre,add and Mtot in mg kg^{-1}, $c_0 - c_3$ regression coefficients (listed in Table)

Table 3-3 Coefficients to calculate the total added metal concentration in soil from the reactive added concentration

Metal	c_0 (-)	c_1 M_{tot} ($mg.kg^{-1}$)	c_2 SOM (%)	c_3 Clay (%)	R^2_{adj}	se-y_{est} [2)]
Cadmium	-0.089	1.075	0.022	-0.062	0.96	0.11
Lead	-0.263	1.089	0.031	-0.112	0.92	0.16
Copper	-0.331	1.152	0.023	-0.171	0.93	0.13
Zinc	-0.703	1.235	0.183	-0.298	0.96	0.16

Step 3: calculation of background total concentration

The amount calculated up to step 2 still only refers to the total added amount of metals in soils. To convert this amount to a corresponding level in field soils a background concentration has to be added. Here we use the results from the Geochemical Atlas of Europe (GEMAS) that was used to derive a relationship between soil properties and background levels in soil (Reimann and Garrett, 2005). The background concentration was found to be related to a combination of pH (KCl), organic matter and clay content. For most soils pH KCl is not available but is closely correlated to pH $CaCl_2$ that is present in the database:

pH-KCl = 0.88*pH-$CaCl_2$ + 0.17

The total background concentration ($Me_{tot\text{-}BG}$) then is calculated as:

Log $M_{tot\text{-}BG} = d_0 + d_1 \cdot \log SOM + d_2 \cdot \log clay + d_3 \cdot$pH-KCl

With $M_{tot\text{-}BG}$ in mg/kg and $d_0 - d_3$ regression coefficients (listed in Table).

Table 3-4 Coefficients to calculate background concentrations for Cd, Cu, Pb and Zn based on pH, organics matter and clay content (from: Reimann and Garrett, 2005)

Metal	d_0 (-)	d_1 SOM (%)	d_2 Clay (%)	d_3 pH-KCl (-)
Cadmium	-1.919	0.418	0.186	0.059
Lead	0.443	0.469	0.267	-
Copper	-0.142	0.481	0.594	-
Zinc	0.330	0.402	0.425	0.076

The final critical concentration in soil (here: M-$Cd_{crit\text{-}total}$) that can be directly compared to the actual measured values is calculated as the sum of the background concentration and the total added concentration from step 2.

M-$Cd_{crit\text{-}total} = M_{tot\text{-}BG} + M_{tot,add(crit)}$

3.5 In summary

Critical concentrations of pollutants in soil can be related to critical concentrations in three environmental compartments: water, food and soil dwelling organisms. For each of these, three critical concentrations are available. For food, critical concentrations are based on WHO food quality criteria, for water critical concentrations based on drinking water criteria or aquatic organisms are available. For soil dwelling organisms critical concentrations in solution have been derived from laboratory studies for a large number of species. All of these can be converted to a corresponding critical concentration in soil that can be compared to current, measured concentrations in soil. For food and ecotoxicology the results are realistic in that the pollutant in the soil is in direct contact with either plant roots (uptake) or the soil dwelling organisms. For water quality the calculation is a worst-case approach since it would assume that water leaving the topsoil is in equilibrium with the groundwater. An alternative approach for water is available but requires a substantial amount of both soil chemical and hydrological data both of which are not available at EU level.

4 Mapped Results for diffuse pollution

4.1 Introduction

As described in Chapter 3, critical limits for metals (here: Cd, Pb, Cu and Zn) depend on the relationships between a critical concentration in food, water or soil solution on one hand and corresponding concentrations in soil. Such relationships depend on key soil properties. Here we use pH, organic matter and clay as model parameters to predict the concentration in either food, water or soil solution. To map critical limits in soil at EU level therefor it is essential to have base maps of such key properties.

All calculations and mapping actions are prepared using the QUICKScan tool. This will be briefly illustrated in next section)

Maps of underlying soil properties used to calculate the critical concentrations are described in section 4.3. For all three key properties (pH, clay and organic matter) two widely used databases (SoilGrids and LUCAS) are used and will be included. By using these different, but widely published databases, we also illustrate the effect of using different input data for the same soil factors (pH, clay and organic matter).

The actual risk is calculated as the difference between the actual metal concentration and the calculated critical concentration. For metals the following maps are available to be used as base maps:

Cadmium: maps based on either GEMAS data (Reimann et al., 2014) after upscaling to a 1 x 1 km grid raster or the map prepared by JRC based on the LUCAS 2009 data (Toth et al., 2016; Ballabio et al., 2023). IN the assessment included in this study we use the maps prepared from the GEMAS data only[5].

Copper: map prepared by JRC based on the LUCAS 2015 database (500 x 500 m grid raster, Ballabio et al., 2018)

Lead: map prepared by JRC based on the LUCAS 2009 data (Toth et al., 2016)

Zinc: map based on the GEMAS (Reimann et al., 2014) data. Maps based on LUCAS data are, at the time of writing not available.

The resulting maps for the three risks considered for the four metals are discussed both in view of the spatial pattern observed across the EU as well as differences between risk maps considering the 2 available databases for soil properties (Section 4.3).

The approach used here allows for the objective comparison of soils across countries more so than using soil quality criteria used by countries. This is not only because countries use different soil quality standards but more so since the underlying concepts used to derive such standards are highly variable. However, the approach applied and illustrated here is a first attempt to predict critical concentrations in soil using selected soil properties, selected risks to be considered and specific transfer models to be used to relate the quality of soil to a specific endpoint. Both data and models are however not flawless and model uncertainty and or the use of soil data from different databases will result in a, possible considerable, uncertainty of the predicted

[5] A comparison (data not shown here) between maps based on GEMAS and LUCAS2009 data revealed that LUCAS, on average has slightly lower concentrations for most of the EU. Since point data from the LUCAS database are not publicly available the reason for this cannot be assessed.

critical concentration. This obviously affects the area where critical concentrations are exceeded as well since this area is calculated as the total area where current concentrations in soil exceed predicted critical concentrations. This is illustrated by the results from the two databases used here. As such this approach is therefore under development. To avoid ample discussion on differences between the maps based on either LUCAS data or SoilGrids data, we do not include exact data on the areas where metal concentrations in soil exceed critical limits. This will be included in an updated version following a more thorough evaluation of the selection of input data to be used (both maps of current metal concentrations in soil as well as soil properties). The results presented here merely are to be considered an illustration of the approach.

4.2 Mapping tool Quickscan

The QUICKScan tool is used to calculate maps of critical metal concentrations from the base maps of key soil properties, using the equations presented in Sections 3.2, 3.3 and 3.4 for food, water and ecotoxicity, next to calculate maps of actual risk for the endpoints of food, water and ecotoxicity using maps of actual metal concentrations in soil. QUICKScan (Verweij et al 2016) is a software tool and a spatial modelling environment to combine expert knowledge with spatial and statistical data. In GOLD, QUICKScan is used to make spatial integrated assessments (SIA) for the whole EU. In QUICKScan, results are visualized in interactive maps, summary charts and trade-off diagrams. Results on any location in the maps of critical metal concentrations can be traced back to the underlying key soil properties. Results in the risk maps can be traced back to actual metal concentrations and critical limits.

4.3 Base maps (pH, clay, organic carbon) used to calculate critical limits for metals in soil

For all three risks considered pH, organic matter and clay soil factors are needed to calculate critical limits for metals in soil (note: current concentrations of metals in soil are not included in this step). In Figure 10, Figure 11 and Figure 12 the base maps for each of these three soil maps are presented derived from LUCAS and from SoilGrids as prepared using QUICKScan. The left side of the figure always refers to the map based on LUCAS data (Aksoy et al., 2016; Orgiazzi et al., 2018), whereas the right-hand side shows the maps based on SOILGRIDS (https://soilgrids.org/). The LUCAS data used are based on the maps provided by JRC (1 x 1 km grid) and converted to a corresponding map in QUICKSCAN without further processing

Table 4-1. Summary of data used in the risk assessment

Element	Metal data to prepare maps of current concentrations	Type of data for metals	soil data (pH, SOM, clay)
Cadmium	GEMAS (Reimann et al., 2014)	Point data, upscaled to 1x1 km	LUCAS & SOILGRIDS
Copper	LUCAS (Ballabio et al., 2018)	Maps as distributed by ESDAC	LUCAS & SOILGRIDS
Lead	LUCAS (Toth et al., 2016)	Maps as distributed by ESDAC	LUCAS & SOILGRIDS
Zinc	GEMAS (Reimann et al., 2014)	Point data, upscaled to 1x1 km	LUCAS & SOILGRIDS

In the results section below critical concentrations of metals are presented on top of the exceedance maps showing areas where the actual concentration exceeds the critical concentration.

Here we calculate critical concentrations based on either the LUCAS soil properties (pH, SOM and clay) as shown in the upper left corner for all four metals. The corresponding critical limit based on soil properties in the SOILGRIDS database are always shown in the upper right corner.

To calculate the exceedance map, the actual concentration for each grid cell (1x1 km) is compared to the corresponding critical concentration for that grid cell. Whenever the actual concentration exceeds the current concentration, the area is considered at risk for the specific risk considered (water quality, food quality or ecosystem health). For Cu and Pb this is done using the maps from the LUCAS database (as listed in Table 4-1). For Cd and Zn this is done using the maps based on GEMAS data (Reimann et al., 2014). The lower left figure for all metals shows the exceedance thus calculated when using the critical concentration based on the LUCAS database. The lower right figure shows the exceedance derived from the critical concentration based on the SOILGRIDS maps for soil properties.

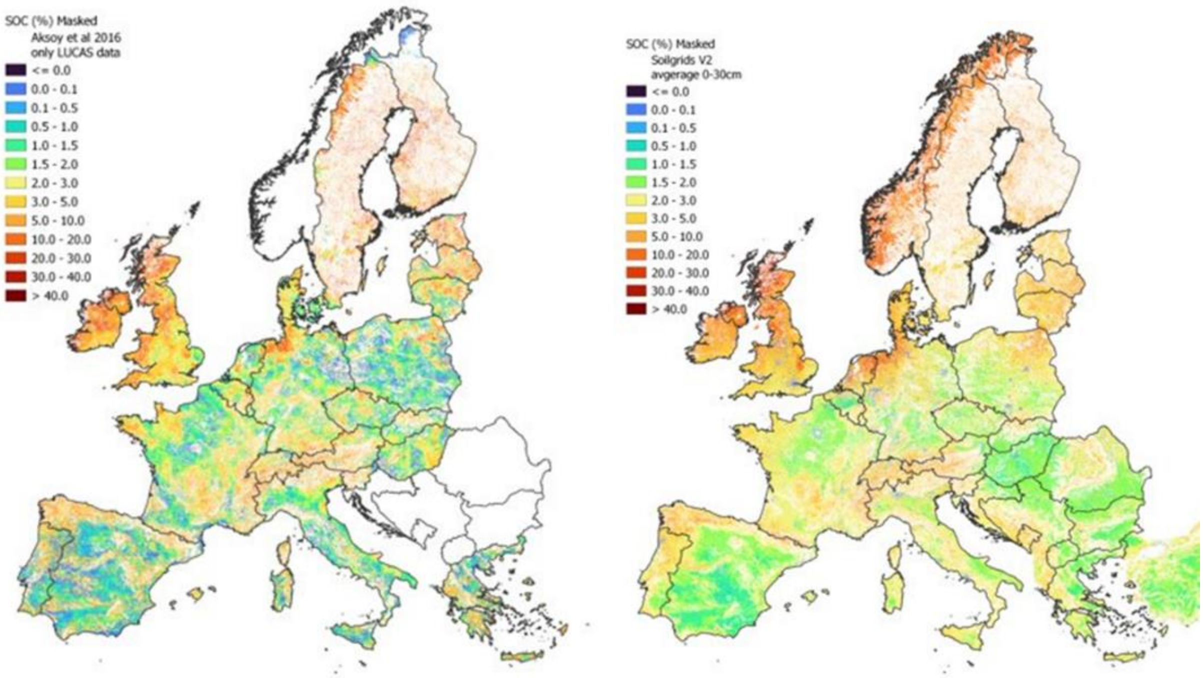

Figure 10 Soil Organic Carbon based on LUCAS database (left) and SoilGrids data (right).

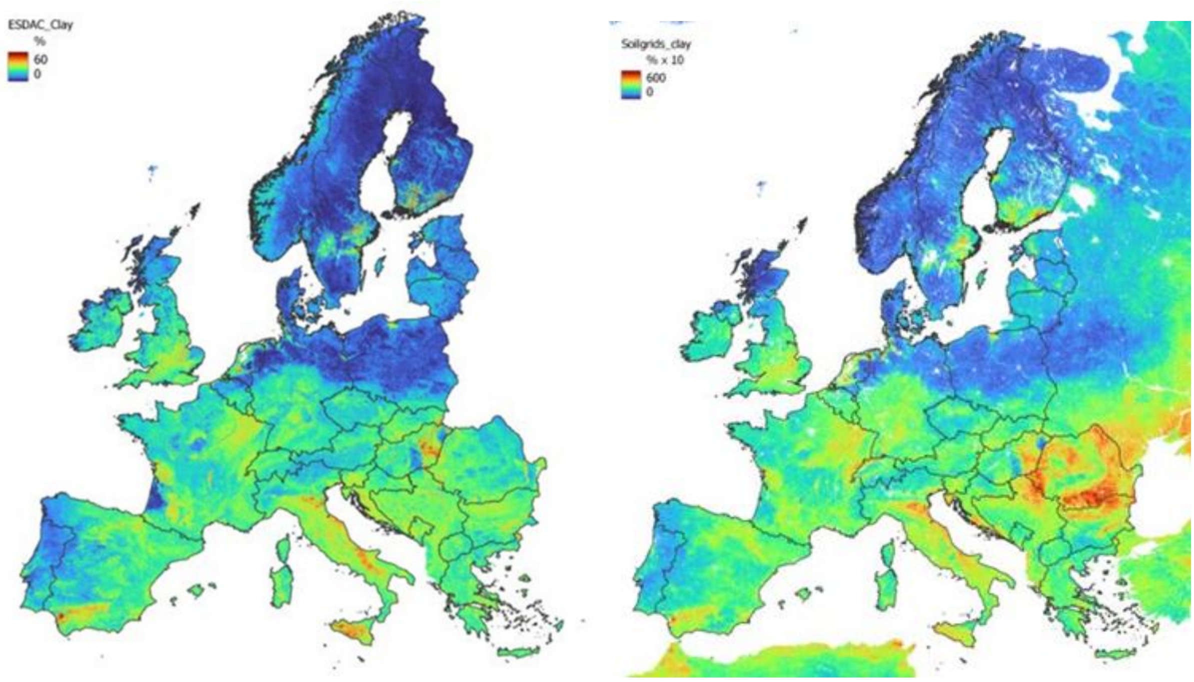

Figure 11 Clay content based on LUCAS database (Left) and SoilGrids (right).

Soil carbon concentrations vary widely between regions in the EU, with, on average higher soil carbon concentrations in the Northwestern part of the EU and lower levels in central and southern areas (Figure 10).

This is largely related to climate; both temperature and rainfall favor the build-up of soil carbon in cool climates compared to warmer climates. Differences between the map based on LUCAS (left) and SoilGrids (right) are clearly visible with lower values of soil carbon in the LUCAS database compared to SoilGrids. Also the spatial patterns of LUCAS seem more variable as can be seen in Spain, France and Poland for example.

Maps of the soil clay content based on both databases are fairly comparable (Figure 11), with the exception of areas in Balkan, notably in Romania, where predicted percentages of clay based on SoilGrids (right) appear to be lower compared to the ones based on LUCAS (Left). The overall pattern observed for clay is that concentrations are low to moderate (sandy, loamy soils) in the North-western part of the EU and moderate to high (loam and clay soils) in most of the southern part of the EU with the exception of Portugal. The main reason for the low clay content in the soils in the northern parts of the EU is the fact that these soils have largely been removed by glacial action during the last ice ages.

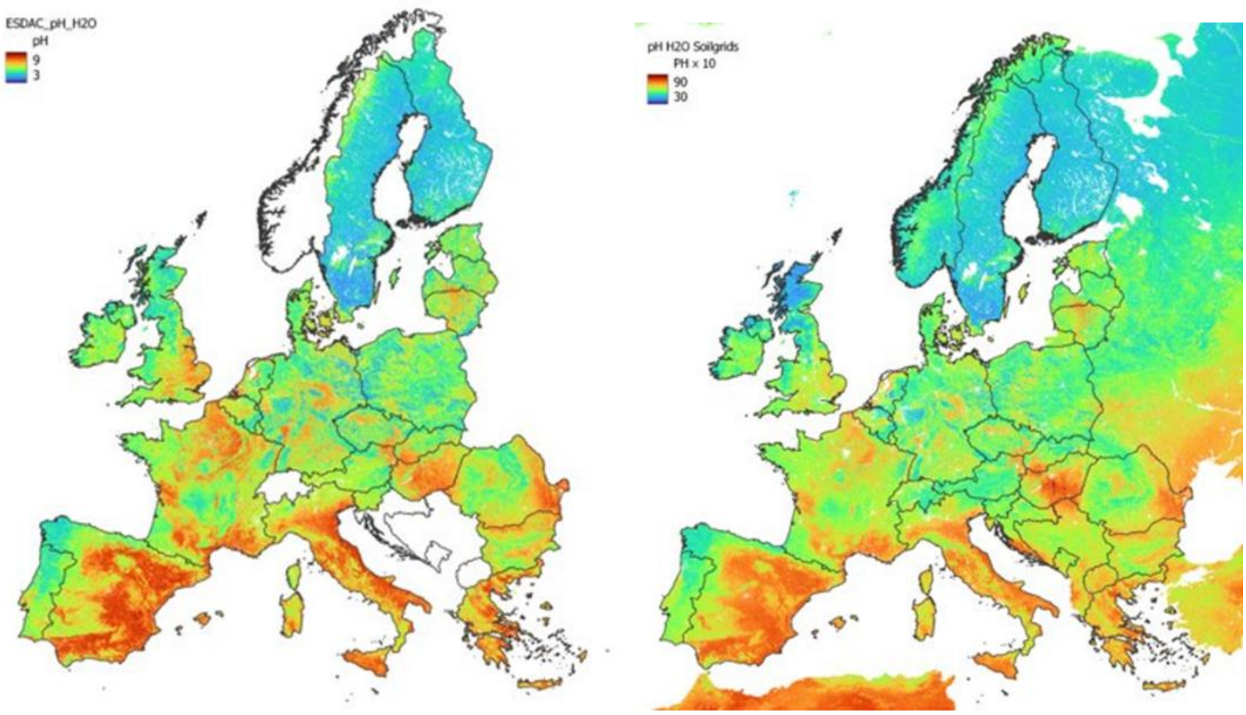

Figure 12 pH-H2O based on LUCAS database (Left) and SoilGrids (right).

The maps of pH are fairly comparable for both databases and show the general pattern that was observed for clay as well (Figure 12). Low pH soils dominate in the North-western parts of the EU whereas calcareous, high pH soils dominate in the southern parts of the EU. This is related to the parent material of the soils which, in most of the Northwest of the EU consists of sandy parent material without any lime present. Calcareous clayey sediments in the South on the other hand result in high pH soils (pH > 7). In some areas, e.g. Hungary, the high pH is also related to saline soils.

4.4 Cadmium

The distribution of Cd follows the pattern as observed for pH and clay, with low Cd soils dominating in the Scandinavian area and, on average, lower Cd concentrations in Mediterranean soils (Figure 13). Regional impact of industrialization and aerial deposition have resulted in higher Cd concentrations in the Netherlands and Belgium, as well as in southern Poland. Also accumulation of Cd in organic rich soils such as in Ireland have resulted in higher Cd concentrations compared to those in most mineral soils. Also in some calcareous sediments in the Balkan higher Cd concentrations prevail, these are also mostly of natural origin and are part of the parent material from which the soils have developed.

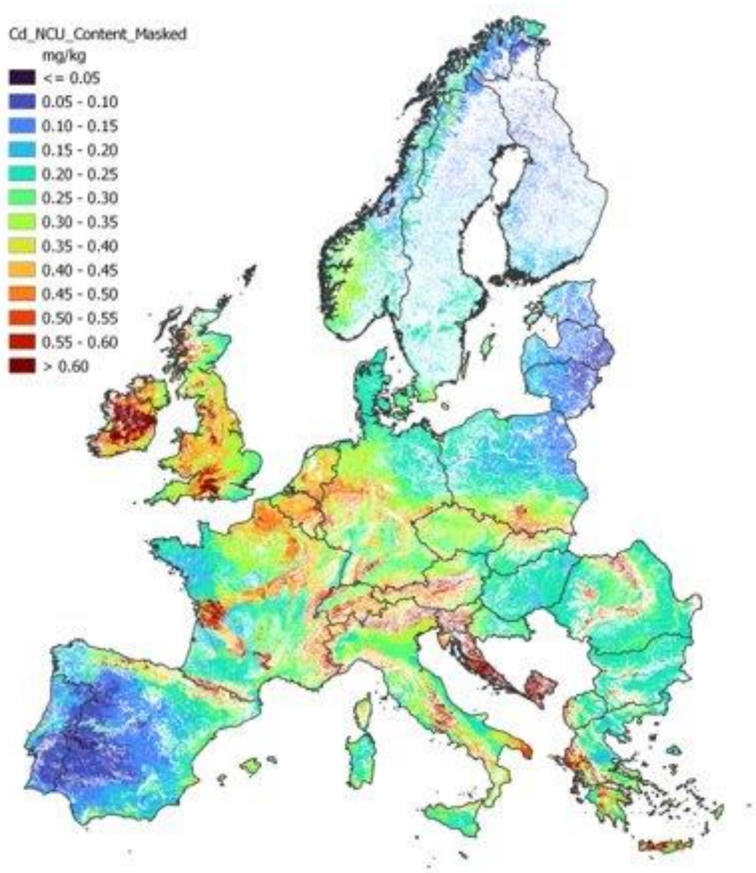

Figure 13 Soil Cd concentration (source data: GEMAS, Reimann et al., 2014).

Critical concentrations for Cd in soil (Figure 14) are close to current concentrations observed in soil and are usually in the range from 0.5 to 1.0 mg/kg. The spatial pattern (top graphs) of the critical concentrations shows that these are lowest in Central Europe and parts of Portugal. This is largely due to a combination of low levels of soil carbon and low pH values present in these areas. Soils rich in soil carbon (such as in Ireland) or high pH soils (Spain, Italy) have higher critical concentrations compared to most mineral soils. The maps based on the LUCAS database tend to yield lower critical concentrations compared to the map based on SoilGrids. Especially in Poland and Portugal this difference is obvious (Figure 14).

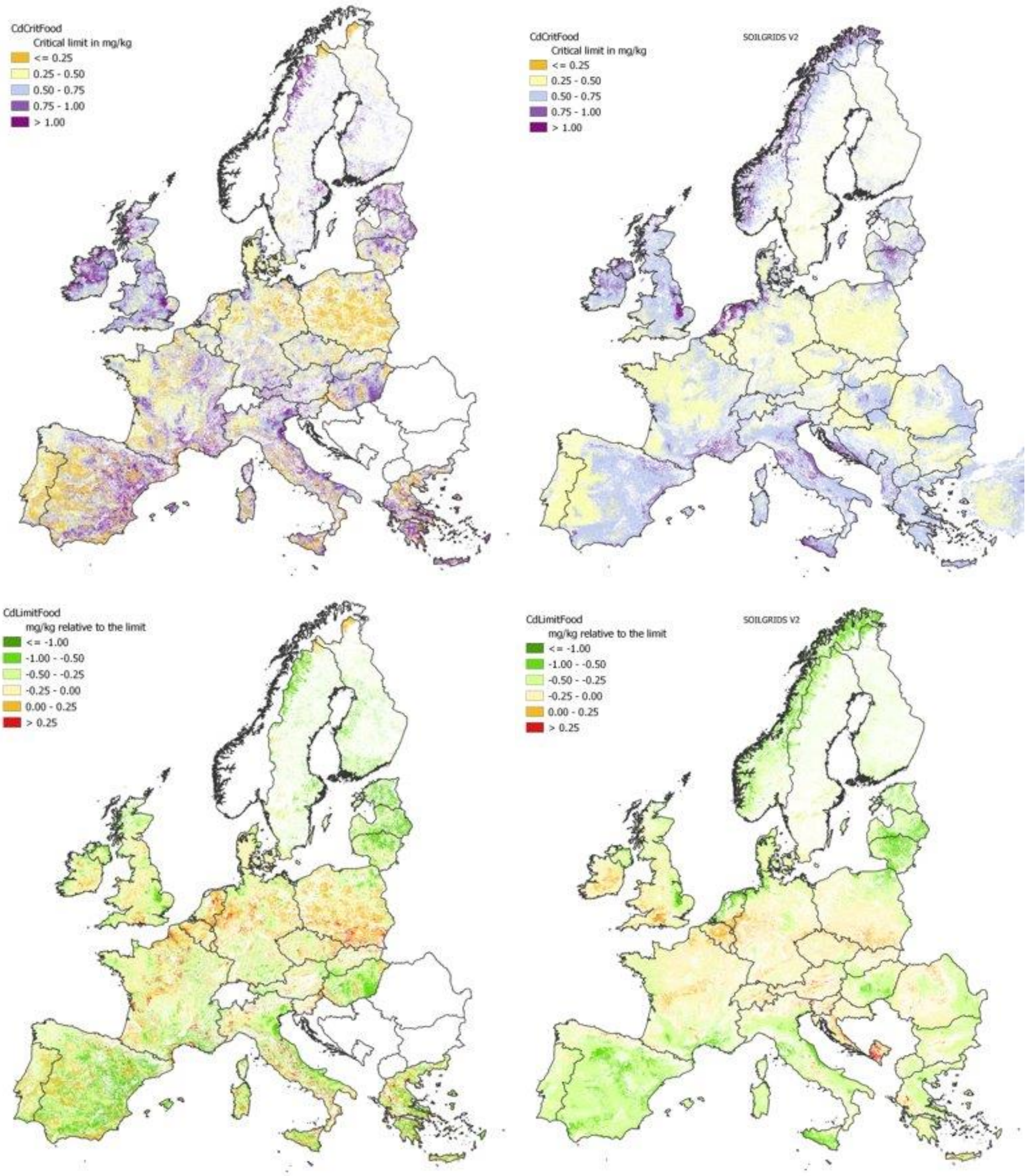

Figure 14 Critical concentrations and exceedance thereof for Cd in view of Food safety. Upper and lower left figures are based on LUCAS data and maps; upper and lower right figures are derived from data from SoilGrids.

Despite the relatively low critical concentrations for Cd, the exceedance rate is still, at EU level, low. Differences between the maps based on LUCAS and SoilGrids however are clear.

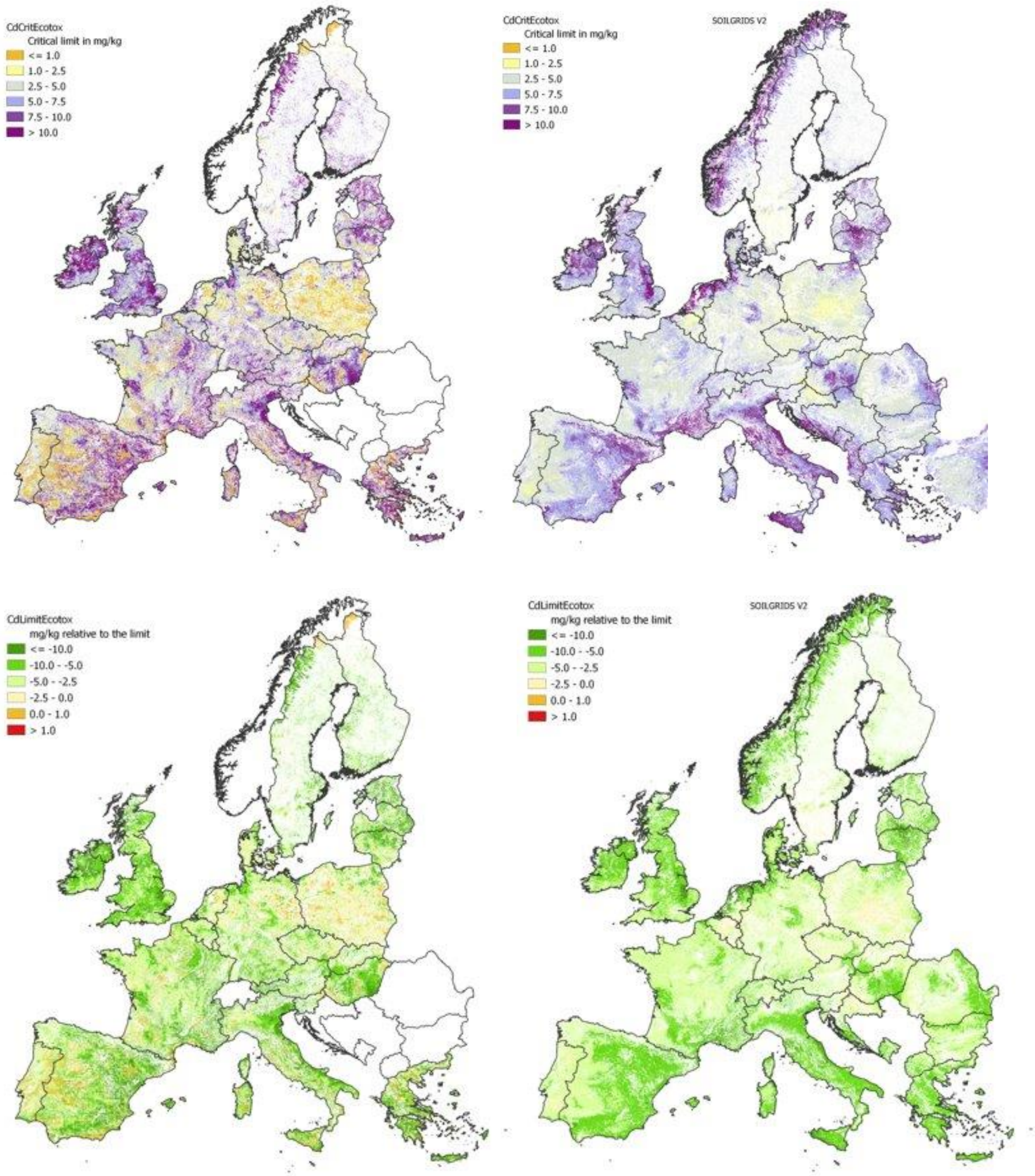

Figure 15 Critical concentrations and exceedance thereof for Cd in view of Ecotoxicological risks. Upper and lower left figures are based on LUCAS data and maps; upper and lower right figures are derived from data from SoilGrids.

Critical concentrations for ecotoxicological risks due to Cd are high (> 1 mg/kg) (Figure 15) and largely in excess of current concentrations in soil. As a result, the exceedance rate is very low. Depending on the base maps used however, exceedance rates increase in areas in Poland which is largely due to the lower soil carbon content in Polish soils as present in the LUCAS database.

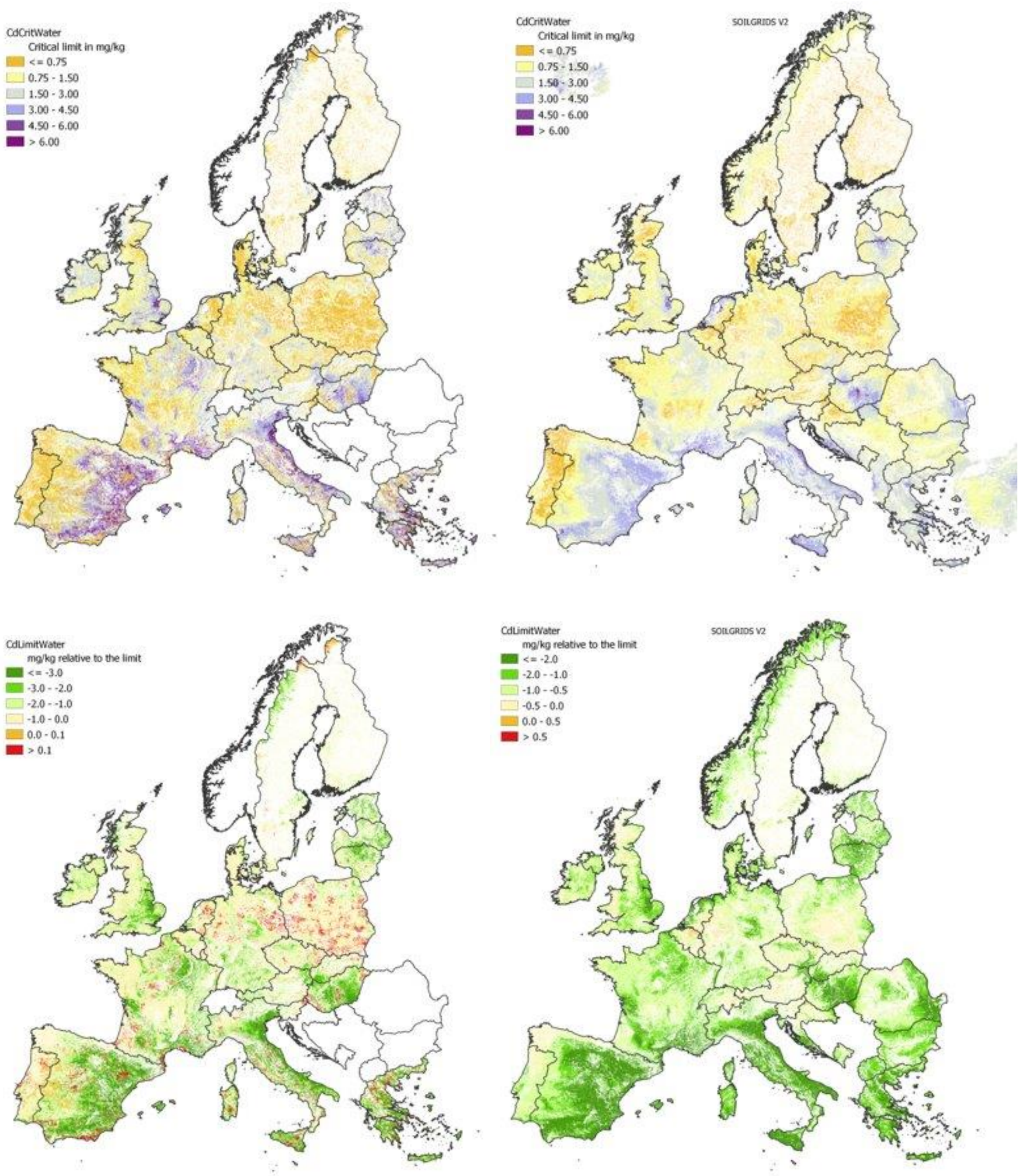

Figure 16 Risk limits and exceedance thereof for Cd in view of Water quality. Upper and lower left figures are based on LUCAS data and maps; upper and lower right figures are derived from data from SoilGrids.

For Cd the WHO guideline for drinking water was used (3 µg/L) to calculate the corresponding critical concentration in soil. The relationship between soil concentrations and corresponding concentrations in water shows that Cd is rather sensitive for pH and, to a lesser extent also SOM more so than most other metals. This results, in areas with either a low SOM (or SOC) content and or a low (pH < 5-6), rather low critical concentrations in soil. This is reflected by the large areas where critical Cd concentrations in soil are below 0.75 mg/kg. This in turn results in the exceedance of the current soil Cd concentration in central Europe Poland, Germany) and areas with a low pH in the north of Portugal. However, differences between the maps based in LUCAS versus those based on SoilGrids are substantial. In contrast to the LUCAS based maps, there is hardly any exceedance for Cd in soil in case of the SoilGrids maps. In addition, these maps need to be used with caution since the approach is based on the apparent relationship between a critical concentration in groundwater applied to the topsoil. In most cases, concentrations of metals in solution tend to decrease during transport from soil to groundwater. An exceedance in the topsoil therefore does not imply that concentrations in drinking water are exceeded as well. As documented in the previous chapter, a more elaborate evaluation that takes into account this delayed transport requires much more information on hydrology and soil properties which at present are not available at the European level. If anything, these maps show areas where elevated Cd concentrations in soil solution can occur due to the combination of low pH and low soil carbon.

4.5 Lead

As for Cd, also the distribution of Pb follows the pattern as observed for pH and clay with low Pb soils dominating in the Scandinavian area and, on average, lower Pb concentrations in Mediterranean soils (Figure 17). More so than for Cd, Cu and Zn, the regional impact of industrialization and aerial deposition, from traffic, have resulted in markedly higher Pb concentrations across the EU near urban areas and industrial areas. This is visible in a.o. the UK, Poland and Germany (Ruhrgebiet).

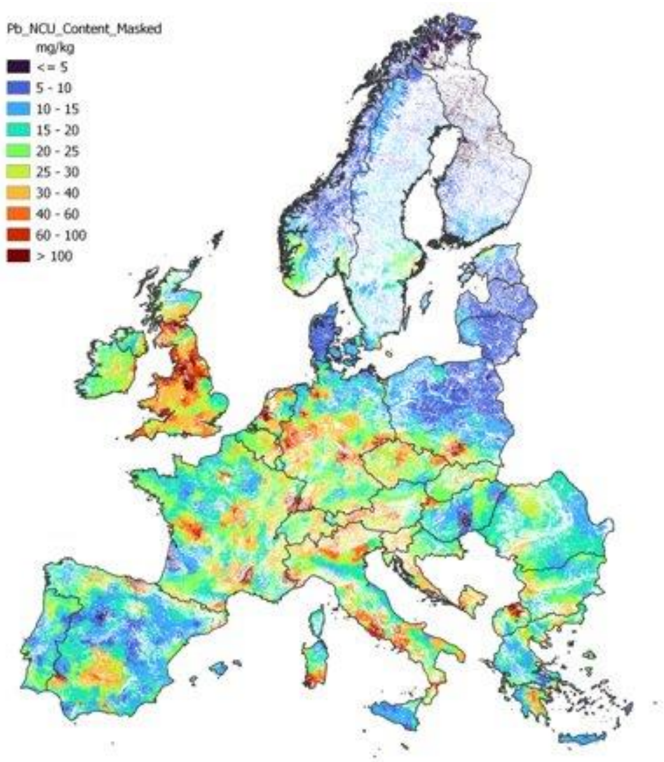

Figure 17 Soil Pb concentration (based on data from GEMAS).

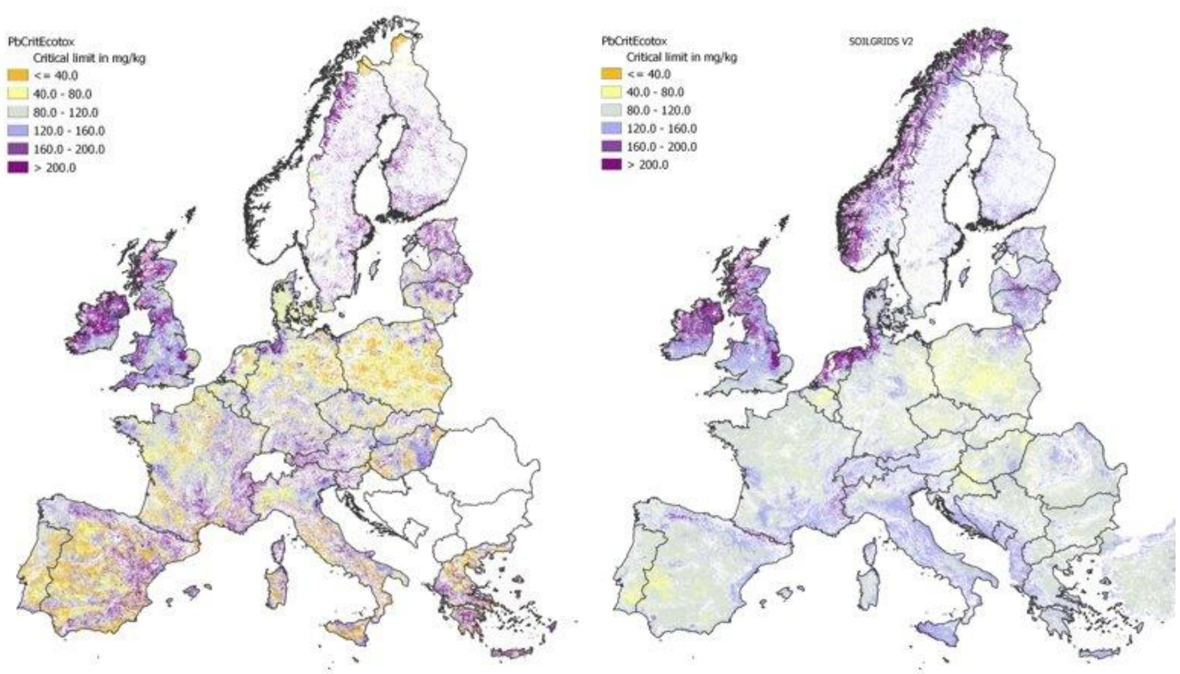

Figure 18 Risk limits and exceedance thereof for Pb in view of Ecotoxicological risks. Upper and lower left figures are based on LUCAS data and maps; upper and lower right figures are derived from data from SoilGrids.

Critical concentrations in soil for Pb in view of ecotoxicological risks range from < 40 to more than 200 mg/kg soil (Figure 18). These values are, in most cases higher than current concentrations of lead in soil that range from 10 to 20 mg/kg (median concentrations) (Figure 17). Obviously higher concentrations occur in and around urban areas as shown by the map of Pb in soil. Since Pb binds quite strongly to organic matter, critical concentrations for Pb are markedly higher in organic carbon rich soils as present in Ireland and other areas in the North-western part of the EU (peat soils). Due to the substantial difference in the organic carbon base maps, maps of the critical concentrations for Pb also reflect this. Regional patterns of the critical concentrations of both maps (LUCAS vs SoilGrids) are different in the area stretching from the NL in the west to Poland in the east. Also in Spain, Portugal and Italy, the regional pattern based on the LUCAS data is more varied and, in general, shows both lower (western parts of Spain and Portugal) and higher (eastern parts of Spain) critical concentrations.

4.6 Copper

Unlike Cd or Pb, sources of Cu are not specifically related to industrialization or urbanization except for the use of Cu in vineyards which has resulted in higher levels of Cu in Mediterranean soils. In most soils, Cu levels are still relatively low even though background levels in the clayey soils in the South (10-20 mg/kg) are higher than those in sandy soils in the North affected by glaciation (5 – 10 mg/kg) (Figure 19). Regionally, as is the case in Italy and parts of the Balkan areas, background levels are however higher that those observed elsewhere. Aside from inputs in areas used for wine growing, such elevated levels are partly of natural origin as well.

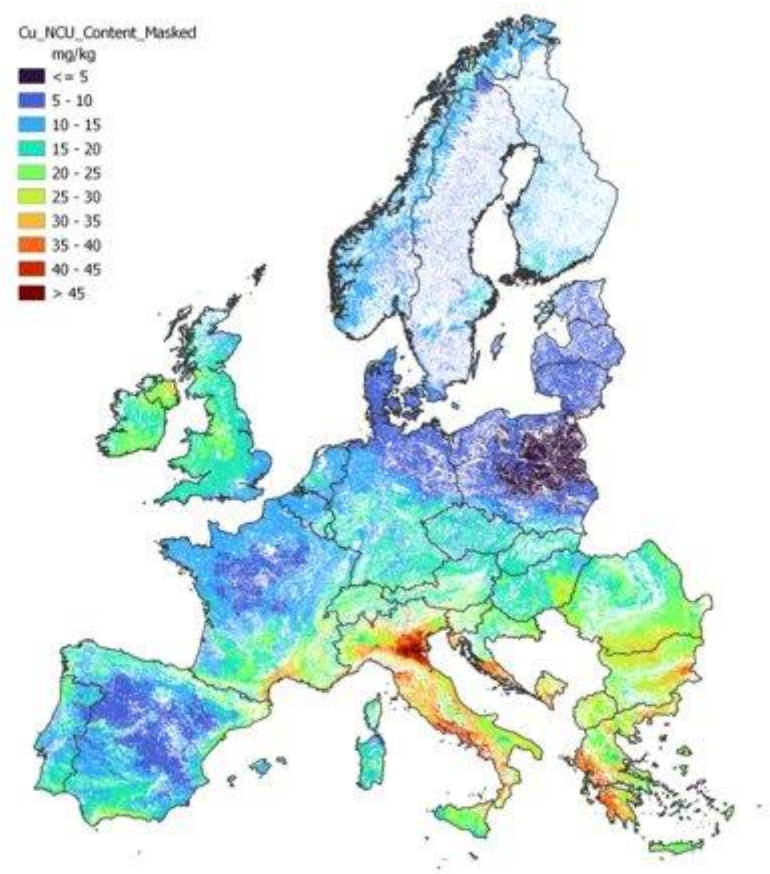

Figure 19 Soil Cu concentration (data: GEMAS).

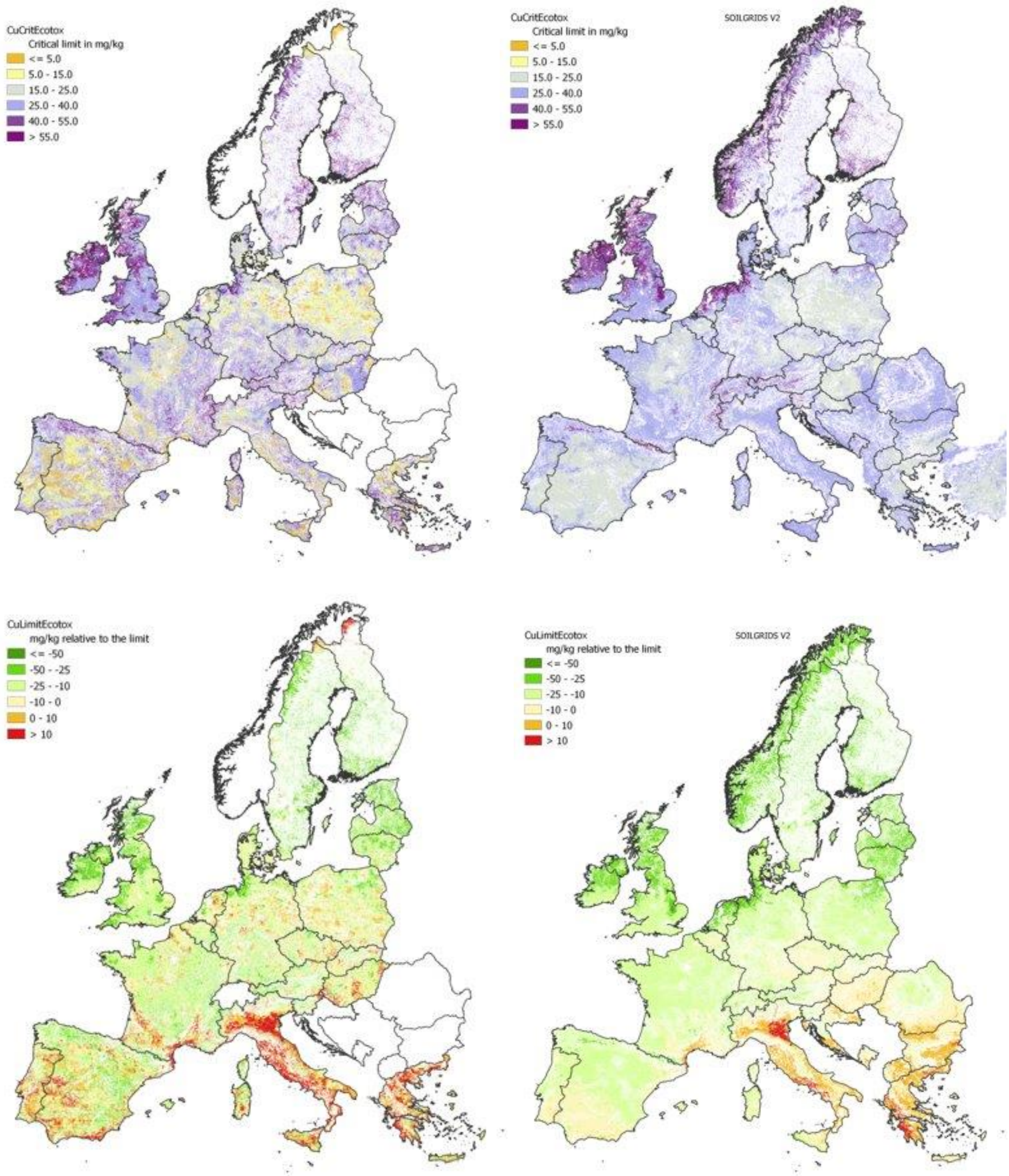

Figure 20 Risk limits and exceedance thereof for Cu in view of Ecotoxicological risks. Upper and lower left figures are based on LUCAS data and maps; upper and lower right figures are derived from data from SoilGrids.

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As for Pb, Cu also preferentially binds to organic carbon and the spatial distribution of the critical concentrations for Cu are rather similar to those for Pb. Higher (> 40 mg/kg) critical concentrations prevail in areas with peat soils whereas soils lower in carbon such as in Poland, show low (< 15 mg/kg) to very low (< 5 mg/kg) critical concentrations (Figure 20). Since total Cu concentrations in soil are also relatively low (median value ranging from 10 mg/kg in soils in the North-west to 20 mg/kg in soils in Mediterranean areas) (Figure 19), critical concentrations are exceeded even though the degree to which this occurs again is related strongly to the base maps used. Especially the LUCAS database shows substantial areas where critical concentrations are close to or below current concentrations. This appears to be the case especially in parts of Italy and selected Balkan countries. Due to the low carbon levels in the LUCAS database in Poland critical concentrations are exceeded more so than in case of the maps based on SoilGrids.

4.7 Zinc

For Zinc data in the LUCAS databased are, at the time of writing not available and the distribution of Zn in soil is solely based on data from the GEMAS database. Patterns of Zn in soil largely coincide with those for Cu with the exception of vineyards areas where Zn levels are not different from those in the surrounding areas. Again, glaciation has resulted in soils low in Zn in the Northern parts of the EU where higher concentrations prevail in central and southern areas (Figure 21). This can partly be explained by the natural presence of Zn in specific clay minerals, especially in river clay deposits. Also higher Zn concentrations can be found in many of the mountain areas across the EU, notably the Alps but also visible in other mountain areas such as the Pennine (UK) and Massif Central (France). For Zn the impact of industrialization is not visible at this scale level.

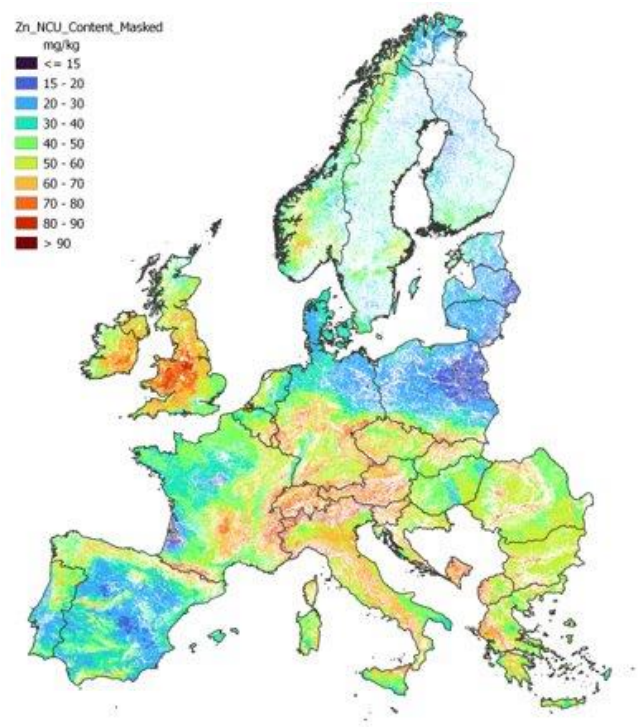

Figure 21 Current soil Zn concentrations (data: GEMAS).

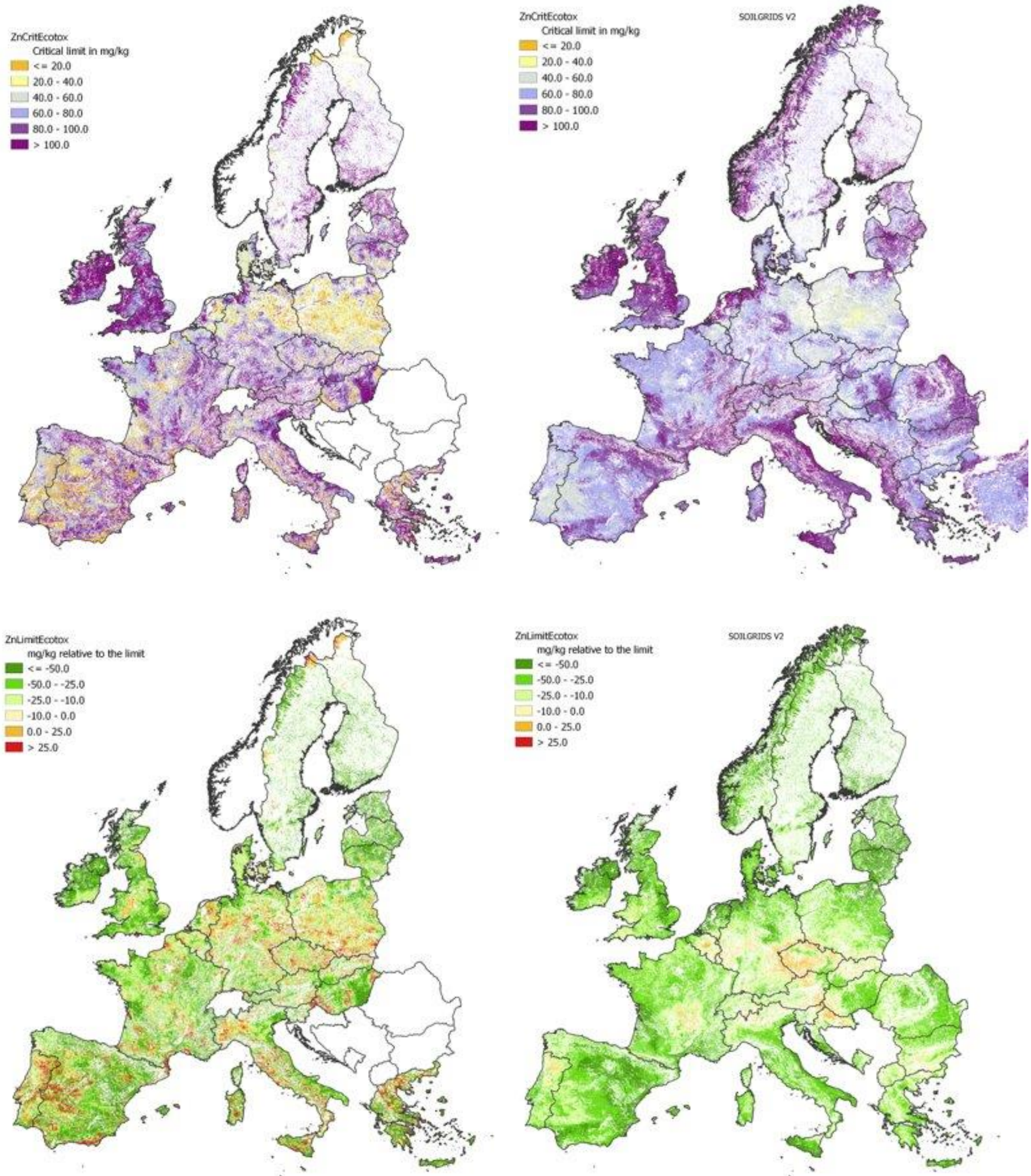

Figure 22 Risk limits and exceedance thereof for Zn in view of Ecotoxicological risks. Upper and lower left figures are based on LUCAS data and maps (soil properties only, metal concentrations to calculate exceedance are based on GEMAS data); upper and lower right figures are derived from data from SoilGrids.

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Critical concentrations for Zn typically range from less than 30 mg/kg to more than 100 mg/kg (Figure 22) and are or can be in the same order of magnitude of current concentrations found in soils (median 30-50 mg/kg). Critical concentrations for Zn in view of ecotoxicology largely depend on organic carbon and pH and the spatial pattern of critical concentrations therefore reflects the combined effect of these two soil properties. Critical concentrations are typically high in organic carbon rich soils and low in soils low in carbon and/or a low pH. This causes a rather patchy pattern for the exceedance maps in case of the LUCAS database, whereas the map based on SoilGrids is more evenly distributed and also shows substantially less areas where current concentrations exceed critical concentrations. In case of the SoilGrids maps exceedance of critical concentrations is limited to areas in B (low carbon, low pH soils) and selected areas in Portugal (low pH soils in the North), Czech Republic (low pH soils in the border areas) and Massif Central in France (Figure 22).

4.8 Conclusions

A major advantage of the risk based approach as outlined in this chapter is that metal concentrations across member states can be compared using the same criteria considering specific risks for humans and the environment. Here risks are expressed in calculated critical concentrations in soil as related to the quality of food, drinking water and ecotoxicology.

Maps of heavy metals are available since the publication of two large databases and corresponding maps, the GEMAS databases (Reimann et al., 2014, point data available) and the LUCAS maps (Toth et al., 2016; Ballabio et al., 2018; point data not available) and are used to construct spatially explicit maps at EU level. The calculation of critical concentrations of metals in soil beyond which the critical concentration in water or food is exceeded requires additional information on soil properties. Key properties include soil organic carbon, pH and clay content.

Here we use the two largest databases currently available (LUCAS and SoilGrids) that do contain all required soil properties. The resulting maps of organic carbon reveal however substantial differences in the spatial pattern and absolute level of soil carbon. As such the spatial pattern of the map based on LUCAS data reveals more detail but also shows that in specific areas lower soil carbon levels are found compared to those on the map based on SoilGrids data.

For pH and clay content the spatial patterns on the map based on either the LUCAS or SoilGrids database are more comparable but also here regional differences are observed.

The differences in organic carbon lead to markedly different critical concentrations for Cd, Cu and Pb. Most noticeable are the lower critical concentrations calculated based on the LUCAS database in Poland, Spain and part of Portugal and Italy. This also leads to differences in the level of exceedance at country level.

In general however, the exceedance risk of Cd critical concentrations appears to be limited as is the exceedance risk in view of ecotoxicology for Pb. For Cu and Zn the exceedance of the ecotoxicological critical concentrations is larger. This is partly related to higher concentrations of Cu in areas in the Mediterranean countries and, for Zn, related to a combination of low pH and low soil carbon concentrations in among others Poland, parts of Spain and Portugal.

However, the difference in the exceedance when comparing results based on LUCAS data versus those based on SoilGrids suggests that these results need to be used with care. Both uncertainty related to differences in basic soil properties as well as model uncertainty (not addressed further in this study) can lead to a substantial range in both the actual concentration of metals and soil carbon and also in the absolute level of the critical concentration.

Despite these shortcomings, the approach outlined here is a promising way to identify areas that are or can be at risk of pollution by the metals addressed in this study. It is however recommended to critically evaluate current soil databases to establish the reliability of maps derived from these databases. In addition, model uncertainty in many of the models used here can be reduced when more data become available. This specifically relates to models used to predict the concentration of metals in food. In contrast to data on soil, data on crop (product) quality ánd soils where these crops are grown are scarce. This is even more of an issue when considering many of the emerging contaminants that are or will become an issue in view of food safety.

5 Identifying sites in the EU affected by point source pollution

5.1 Introduction

The assignment in the GOLD project is to map areas of sites in the EU that are contaminated to some degree, that need cleaning or stabilisation and that may be suitable for bioremediation through the cultivation of biofuel crops. The mapping exercise should result in locations, and surface area of these sites, documented with the prevalent soil properties and contaminations present. The resulting maps will be used in the GOLD project to assess the potential of these terrains for the production of the biofuel crops selected in WP1. The mapped information will also be used in scenario studies to quantify the production of biomass and amounts of soil pollution that could be remediated in next tasks in WP3.

As a first step, datasets and reports on contaminated sites in Europe were sought in data and knowledge platforms of EU institutions (European Commission, EEA, EIONET). Enquiry at ESDAC and consultation of the websites of EEA and Eurostat revealed that at present, there is not database of contaminated sites for Europe that carries spatially referenced information on area and contaminants.

The most recent Europe-wide assessment of contaminated sites is the JRC Technical Report Status of local soil contamination in Europe by Payá Pérez & Eugenio (2018), which was based on questionnaires to experts of national reference centres (NRCs) in the EEA-member countries. Of the 39 countries surveyed, 28 maintain comprehensive inventories for contaminated sites at national or regional level. The study revealed that 65.000 sites that had been remediated or are under aftercare, and 650.000 sites are registered as sites where polluting activities took or takes place (Payá Pérez & Eugenio, 2018).

Among the Land and soil indicators in the EEA indicator management system, the indicator *LSI003: progress in the management of contaminated sites* provides information on contaminated sites. The data were collected through questionnaires from EIONET countries (overlapping with the questionnaire that provided the information for the above-mentioned report by Payá Pérez & Eugenio (2018)). The dataset contains total numbers of contaminated sites and population per country, but no information on degree or type of contamination, nor of the areas affected.

For this reason, we have taken another approach to mapping contaminated sites, i.e. to identify potentially contaminated areas from Open Street Map based on properties of geographical objects, and to cross-check these areas with information on land cover and with recordings of contaminated sites in the literature and the internet. National registers of contaminated sites will be consulted for several countries in 2023.

We will also try to find polluted areas using other methods than OpenStreetMap (OSM), because not all types of pollution are covered using OSM. For example, land currently in use as agricultural land, that was previously used for irrigation with or treatment of wastewater, or for the disposal of sewage sludge.

5.2 Identifying sites with contamination risk in Open Street Map

5.2.1 Open Street Map (OSM)

Open Street Map carries features of geographical objects, that can be queried using 'tags'[6]. These are expressions of the properties of the geographical objects in terms of combinations of 'keys' and 'values', for example: 'landuse = brownfield'. By using tags for the key *land use*[7], areas, sites and land use types can be identified where activities occur or occurred that may have caused or still cause soil pollution.

Not all locations or objects found in this way will be actually polluted, and not all will be suitable for the cultivation of biomass crops, for example when located in built-up area or where the pollution is located below the topsoil layer. Inversely, there are sites that were polluted by activities in the past but cannot be identified from the descriptions of geographical objects in Open Street Map. For example, an ancient land fill that is currently in use as recreational area, and does not match the tag 'key = landfill' in Open Street Map.

For these reasons, a cross-validation between the sites identified in Open Street Map with independent sources of information on contaminated sites is necessary. European registers of contaminated sites and recordings of existing contaminated sites in the literature were used for this purpose. In the approach we adopted, these data sources are presumed to contain the most detailed and reliable information on actually contaminated sites in a country or region. We used indications of these sites in Open Street Map to find potentially contaminated sites in countries where national registers of recordings are not available. The current land cover in these sites was analysed using CORINE Land Cover (CLC2018). For mining sites in OSM we also enriched the analysis with data from the Minerals4EU database.

5.2.2. Identification of possibly contaminated sites in Open Street Map

The *turbo overpass utility* in Open Street Map was used to retrieve point locations of the following geographical objects with presumed potential to have caused soil pollution in the direct neighbourhood:

- (former) quarries and mine tailings
- (former) land fill sites
- (former) military sites
- former industrial sites (brownfields)
- industrial sites
- harbours
- wastewater treatment plants
- fuel stations

Next, the point locations resulting from the tags were used to retrieve the polygons in which the objects are located, with the current land use. The number of properties in Open Street Map that is used to describe geographical objects is unlimited. Therefore there are numerous combinations of keys and values describing

[6] https://wiki.openstreetmap.org/wiki/Map_features
[7] https://wiki.openstreetmap.org/wiki/Map_features#Landuse

potentially polluted areas, and descriptors of contaminated sites that come up after the first query are used in subsequent queries (see Figure 23 and Figure 24).

https://wiki.openstreetmap.org/wiki/Tag:man_made=spoil_heap	Tag:man_made=spoil_heap	https://taginfo.openstreetmap.org/keys/man_made#values
https://wiki.openstreetmap.org/wiki/Tag:man_made%3Dtailings_pond	Tag:man_made=tailings_pond	https://taginfo.openstreetmap.org/keys/man_made#values
https://wiki.openstreetmap.org/wiki/Tag:hazard%3Dcontamination	Tag:hazard=contamination	https://taginfo.openstreetmap.org/keys/contamination#values
https://taginfo.openstreetmap.org/search?q=wasteland#values		https://taginfo.openstreetmap.org/search?q=wasteland#values
https://wiki.openstreetmap.org/wiki/Tag:landuse%3Dbrownfield	Tag:landuse=brownfield	https://taginfo.openstreetmap.org/tags/landuse=brownfield
https://wiki.openstreetmap.org/wiki/Key:military		https://taginfo.openstreetmap.org/tags/landuse=military
https://taginfo.openstreetmap.org/tags/disused=yes#overview		https://taginfo.openstreetmap.org/tags/disused=yes#overview
https://wiki.openstreetmap.org/wiki/Tag:military%3Ddanger_area		

Figure 23 Extract of the tags, keys and values used to identify potentially contaminated sites in Open Street Map.


Tag:man_made=spoil_heap

Tag:man made=spoil heap - Other languages Purge · Help

Deutsch · English · polski · русский ·

Other languages · Translate

Spoil heaps, otherwise known as Spoil tip are piles of waste rock removed during mining.

This tag may also be used for piles of dredge tailings, which is the sediment that is dumped next to a river/lake when the river is dredged (made deeper using excavators and other machinery). Not to be confused with man made=tailings pond

How to map

Map the outline of the spoil heap as a closed polygon and tag it with man_made=spoil_heap.

- If there is a common name add name=*

Example

Monte Kali in Germany

man_made = spoil_heap

Description

Spoil heap, piles of waste rock removed during mining

Group: Man made

Used on these elements

Useful combination

- name=*

Status: in use

taginfo [More...]

	72	0.00 %
	3 815	0.14 %
	46	0.31 %

Figure 24 Example of a tag used to identify potentially contaminated sites in Open Street Map.

An example of the identification of potentially contaminated sites in Open Street Map is illustrated for quarries in Figure 25 and Figure 26. The areas of extracted polygons from Open Street Map with land use types which are likely to contain contaminated sites are listed per country in Table . Only the polygons with an area larger than 1 ha were included, because this area is considered the minimum required size to establish cropping fields for phytoremediation and biomass production.

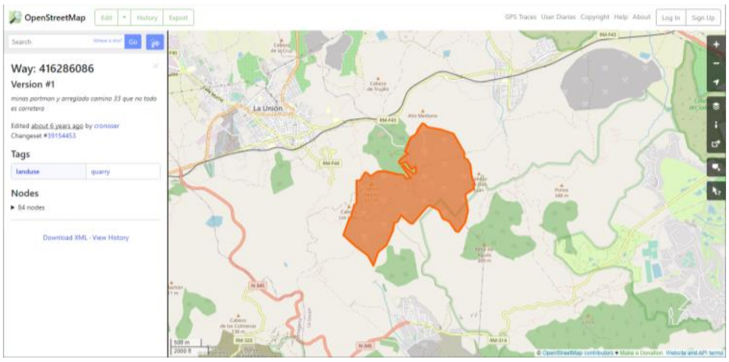


Figure 25 Quarry location in OSM, identified through the tag land use=quarry.

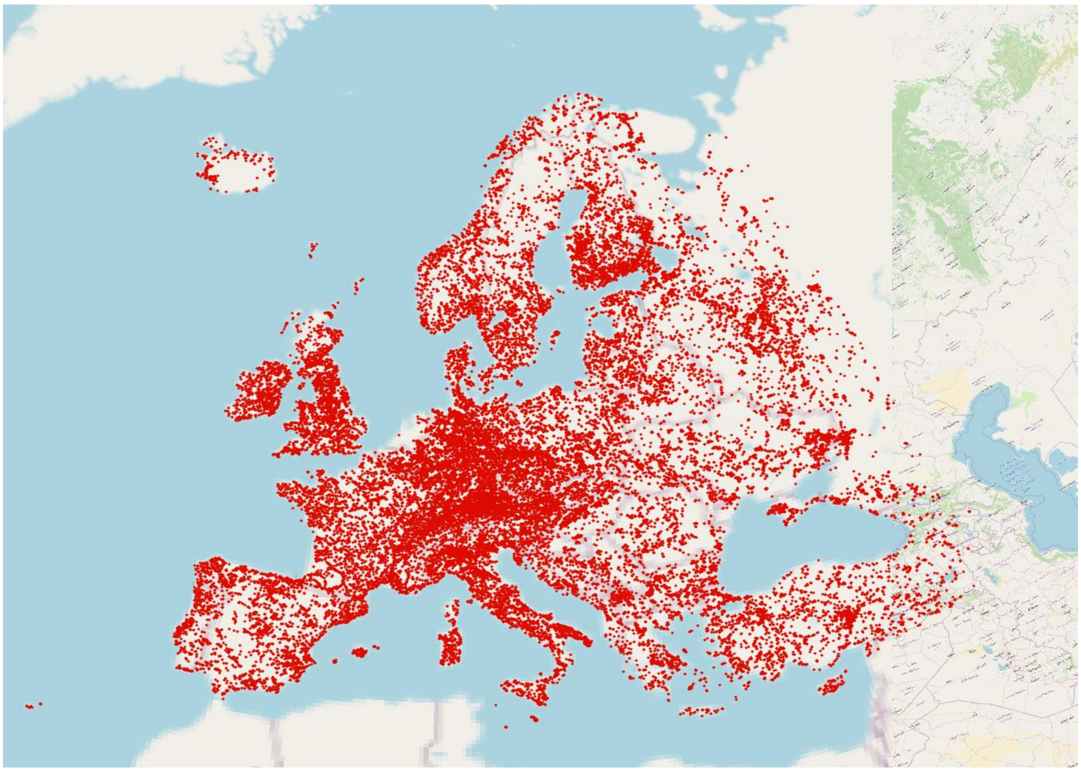


Figure 26 All quarry locations in OSM, identified through the OSM tag=quarry (679.529 sites).

Table 5-1 Surface Area (on land only; in hectares) of extracted polygons from OSM per country larger than 1 ha. (Notice: Polygons of land use categories may overlap!)

Hectares of extracted OSM polygons per country (individually having an area >= 1.0 ha)								
Country	landuse='brownfield'	amenity='fuel'	landuse='industrial'	landuse='landfill'	landuse='military'	landuse='harbour'	landuse='quarry'	man_made='wastewater_plant'
AT	260		22981	1284	31870	34	11037	370
BE	1306		57829	993	24276	7	5950	254
BG	1350		37547	3291	12539	74	31126	
CH	41		18813	422	4824		4556	120
CY	958		2914	137	10985	221	2522	95
CZ	1691		51106	2802	94061		22769	265
DE	9574	9	307889	22764	325056	708	102626	2695
DK	105		36369	255	43842	75	4736	60
EE	322		8599	3445	30059	78	16296	72
ES	2081	34	170180	5124	107788	2212	54299	407
FI	265	1	49612	4746	393405	62	58235	30
FR	4992	16	281080	11696	219228	952	71813	411
GR	313	2	18662	2395	19912	642	26311	9
HR	500	1	8779	799	28521	45	2876	12
HU	2664		54879	2760	18393	1	11078	385
IE	489	8	12846	970	8887	129	12172	28
IT	3562	72	206742	6987	100562	1470	43830	222
LT	270		23217	164	12541		16519	244
LU	125		2074	206	126		408	28
LV	818		10118	197	14667		16650	150
NL	653		94765	858	21115	25	1189	469
NO	71		17020	440	136636	339	10676	
PL	3295	27	118465	12836	152707	73	41925	2179
PT	384	53	30432	1335	15699	123	10973	77
RO	2099	1	115259	6223	21177	1312	24269	154
SE	448		66070	2844	711016	85	20141	85
SI	19		4972	98	2809	3	1228	7
SK	262		28603	1468	61825		4590	24
UK	10578	6	151413	7349	153628	324	48729	1139
Grand Tot	49492	230	2009240	104888	2778153	8996	679529	9990

5.2.3 Further characterization of sites combining OSM selections with CORINE Land Cover (CLC2018) and imperviousness

The surface area (on land) of extracted polygons from OSM per country larger than 1 ha is given in Table . Military sites cover the largest area. The distribution of CORINE land cover classes in OSM features over countries in the EU27 and UK is given in Figure 27 to Figure 36. CLC classes consisting of water bodies were not considered. Figures of land cover are not given for the OSM features harbors, wastewater treatment plants and fuel stations because of their limited areal extent.

Information on the current land cover is drawn from the CORINE Land Cover database (CLC), version 2018, of the Copernicus Land Monitoring Service[8]. The land cover classes that are considered suitable for phytoremediation are listed in Table . Completely built-up areas and forest are excluded. In order to estimate the area available for cropping in the land cover types that include partly built-up area, we use the pan-European High Resolution Layer 'Imperviousness degree' (IMD) (pixel size 100 m, reference year 2018)[9] of the Copernicus Land Monitoring Service. This layer gives the percentage of impervious (sealed) soil surface per area unit.

Table 5-2 Land cover classes considered suitable for phytoremediation in potentially contaminated sites, for the non-built-up part. Source: CORINE Land Cover database (CLC).

CLC class	CLC class label	Description
2	112	Discontinuous urban fabric
3	121	Industrial or commercial units
4	122	Road and rail networks and associated land
5	123	Port areas
6	124	Airports
7	131	Mineral extraction sites
8	132	Dump sites
10	141	Green urban areas
11	142	Sport and leisure facilities
12	211	Non-irrigated arable land
13	212	Permanently irrigated land
14	213	Rice fields
15	221	Vineyards
16	222	Fruit trees and berry plantations
17	223	Olive groves
18	231	Pastures
19	241	Annual crops associated with permanent crops
20	242	Complex cultivation patterns
21	243	Land principally occupied by agriculture with significant areas of natural vegetation
22	244	Agro-forestry areas
26	321	Natural grasslands

[8] https://land.copernicus.eu/pan-european/corine-land-cover
[9] https://land.copernicus.eu/pan-european/high-resolution-layers/imperviousness

CLC class	CLC class label	Description
27	322	Moors and heathland
28	323	Sclerophyllous vegetation
29	324	Transitional woodland-shrub
32	333	Sparsely vegetated areas
33	334	Burnt areas
38	422	Salines

Military sites

Potential pollutions in military sites

Pollution of military sites are usually found in the shooting ranges and areas for maintenance of equipment. Shooting ranges are typically characterized by local hot spots where extremely high concentrations of metals like lead (Pb), copper (Cu) and antimony (Sb) can be found (Mesman et al., 2014; Römkens and Faber, 2020). The highest concentrations typically are found in what is called 'bullet-catchers', usually small man-made ridges where the ammunition fired accumulated. In the Netherlands. Locally pollution with PAHs occurs in areas where machinery is maintained or used.

Typical for military shooting ranges (in contrast to recreational shooting ranges) is the presence of various organic compounds belonging to the chemical groups of nitramines and nitroaromatics. These compounds originate from the use of explosives other than ammunition use by riffles (Broomandi et al., 2020).

In many cases shooting ranges are not used for any form of agriculture and are or have been, by definition, enclosed areas. Abandoned shooting ranges often are covered by forest or other forms of natural vegetation adapted to sometimes extremely high pollution levels (Römkens and Faber, 2020). Lead (Pb) being the most found metal also has limited potential in view of phytoextraction due to the limited transfer from soil to crop. Risks of contamination of ground- and surface water strongly depends on the local soil conditions.

Land cover in military sites

Military sites in Open Street Map are covered for the largest part by forest, woodland and shrubs and natural grassland on the Corine Land Cover map of 2018 (Figure 27). This combination of land cover types with the indication 'military sites' on Open Street Map is expected, because military sites are often positioned in areas with these land cover types for the purpose of military exercises, and to avoid safety issues with areas where people work and live, and at distance from areas with vulnerable objects (e.g. industries, infrastructure, water bodies).

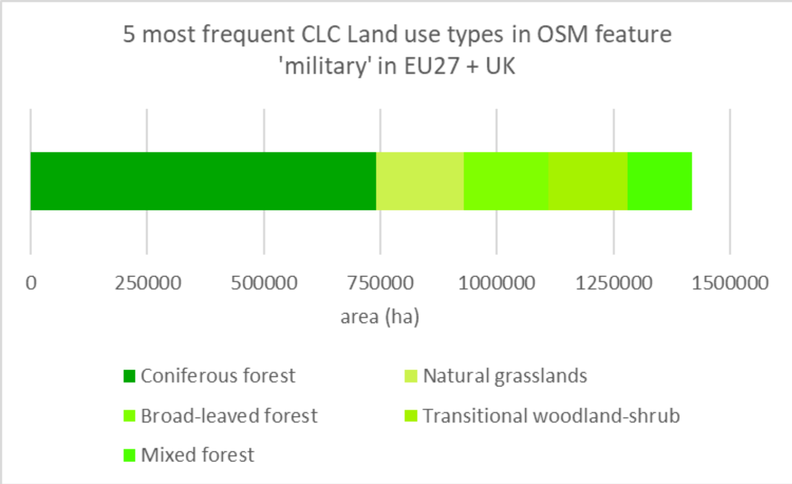


Figure 27 Area in military sites on Open Street Map covered by the 5 land cover classes with the largest areal extent. Source data: Open Street Map and CLC 2018.

The largest areas covered by military sites are found in Germany, Finland and Sweden, with between 250,000 and 350,000 ha (Figure 28). Most of the land cover is in the form of forest, transitional woodland and shrub. In Finland and Sweden part of the cover is by peat bogs. Only a small share of the military sites in Open Street Map is covered by built-up area (152,000 ha; see Figure 28).

Forest and semi-natural vegetation may be able the stabilize polluting substances in place. Considering the large share of these land cover types in military sites, these sites are less relevant for the implementation of phytoremediation with bioenergy crops. However, in case there is polluted land with no or limited vegetation cover and buildings there may be opportunities for biomass cropping with phytoremediation. These situations may occur where military sites overlap with CLC classes indicating presence of open terrain such as transitional woodland shrub, pastures, airports, open land within industrial or commercial units (see also Figure 16). In all other land cover classes establishment of new crops may not be a good choice as it will disturb strongly natural vegetation cover present and strong loss of above and below ground carbon.

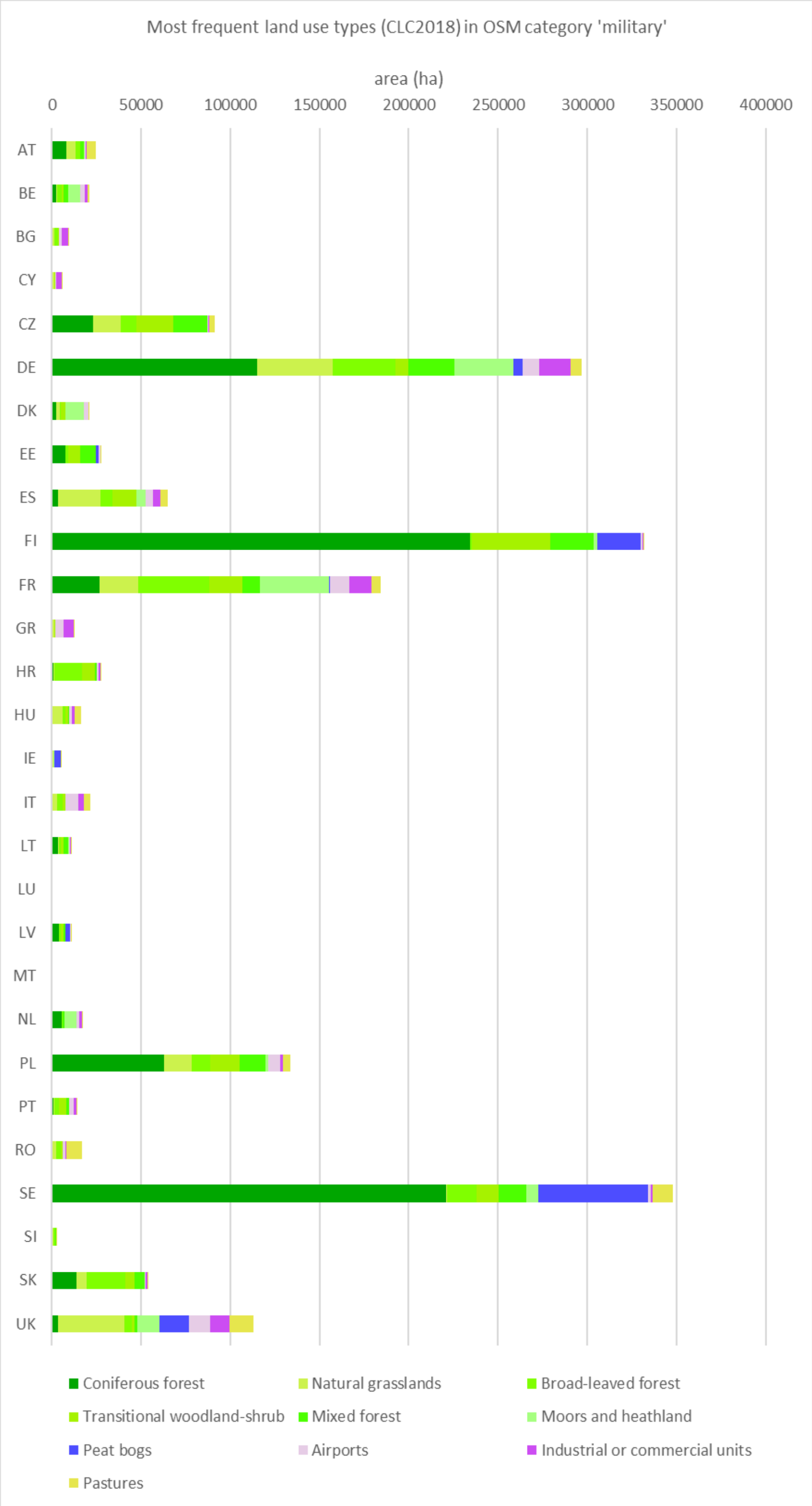


Figure 28 Most frequent (top 10) land cover types in military sites detected in OSM. Source data: Open Street Map and CLC 2018.

Industrial sites

Potential pollutions in industrial sites

Soil pollution in or near industrial areas are characterized by an enormous range in both type of pollutant found, concentration ranges present and the area affected. Depending on the type of industry pollution with both organic and inorganic chemicals can occur. In case of industry that processes specific ores of metals, the range in the type of pollutants is limited. In the Dutch Belgian border area for example, processing of Zn ores imported from various countries has resulted in a wide area affected by 4 metals mostly (Cd, Zn, Pb and As). In other areas petrochemical industries on the other hand have emitted a wide range of organic pollutants including but not limited to PAHs', mineral oil, PCBs etc. This shows that for industrial pollution no specific pollutants can be pointed at; and the type of industry must be considered in order to be able to predict which kind of pollutant can be found and in what concentration ranges. A recent estimate of soil pollution in the EU (FAO and UNEP 2021) reveals that about two-thirds of the number of cases of soil pollution stems from industrial pollution in combination with waste management and disposal thereof.

In contrast to military shooting ranges, industrial soil pollution often can be located near areas used for housing. Especially in case of extreme soil pollution which prevents the area being revegetated, emission of dust via wind erosion can directly affect human health. In those cases, reduction of erosion via phytostabilization can help to reduce human exposure. In addition, this can also reduce leaching and increase the potential for soil life as was demonstrated in extremely polluted soils in the Kempen where soil life increased substantially after revegetation of previously bare soil (Bouwman and Vangrondsveld, 2004.

Land cover in industrial sites

Industrial sites in Open Street Map correspond for the largest part to the land cover class 'industrial or commercial units' in CLC 2018 (1,144,893 ha, Figure 29). The second most widespread land cover in industrial sites is 'discontinuous urban fabric' (247,946 ha). These land cover types are expected in industrial sites.

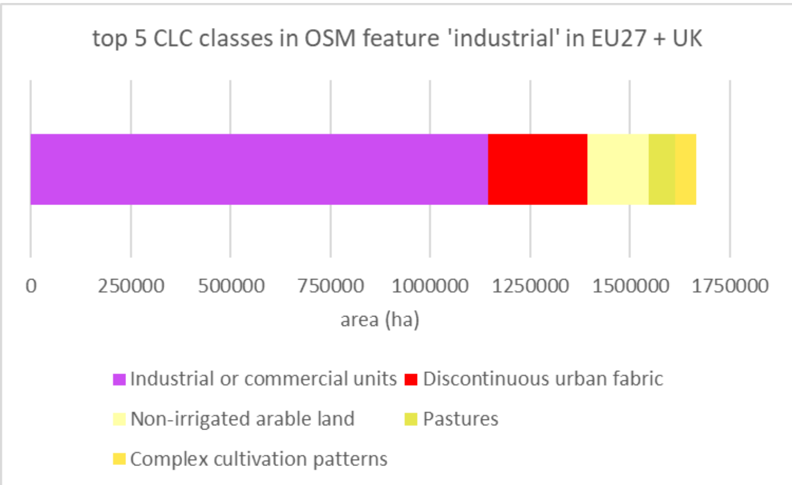


Figure 29 Area in industrial sites on Open Street Map covered by the 5 land cover classes with the largest areal extent. Source data: Open Street Map and CLC 2018.

The largest areas with industrial sites identified in Open Street Map are in Germany, France and Italy (resp. 302,581, 274,506 and 202,531 ha) (Figure 30). These are mostly covered by land cover classes 'industrial or commercial units' and 'discontinuous urban fabric', and therefore are already mostly built-up area, leaving little room for biomass cropping. Still Considering all countries in the EU27 and UK, part of the industrial sites in OSM is covered by agricultural land (classes 'non-irrigated arable land', 'pastures' and 'complex cultivation patterns' in CLC 2018; see Figure 29 and Figure 30). If polluting substances are present in the soils of these areas, they may be relevant for phytoremediation using bioenergy crops. This requires an assessment at the level of these sites.

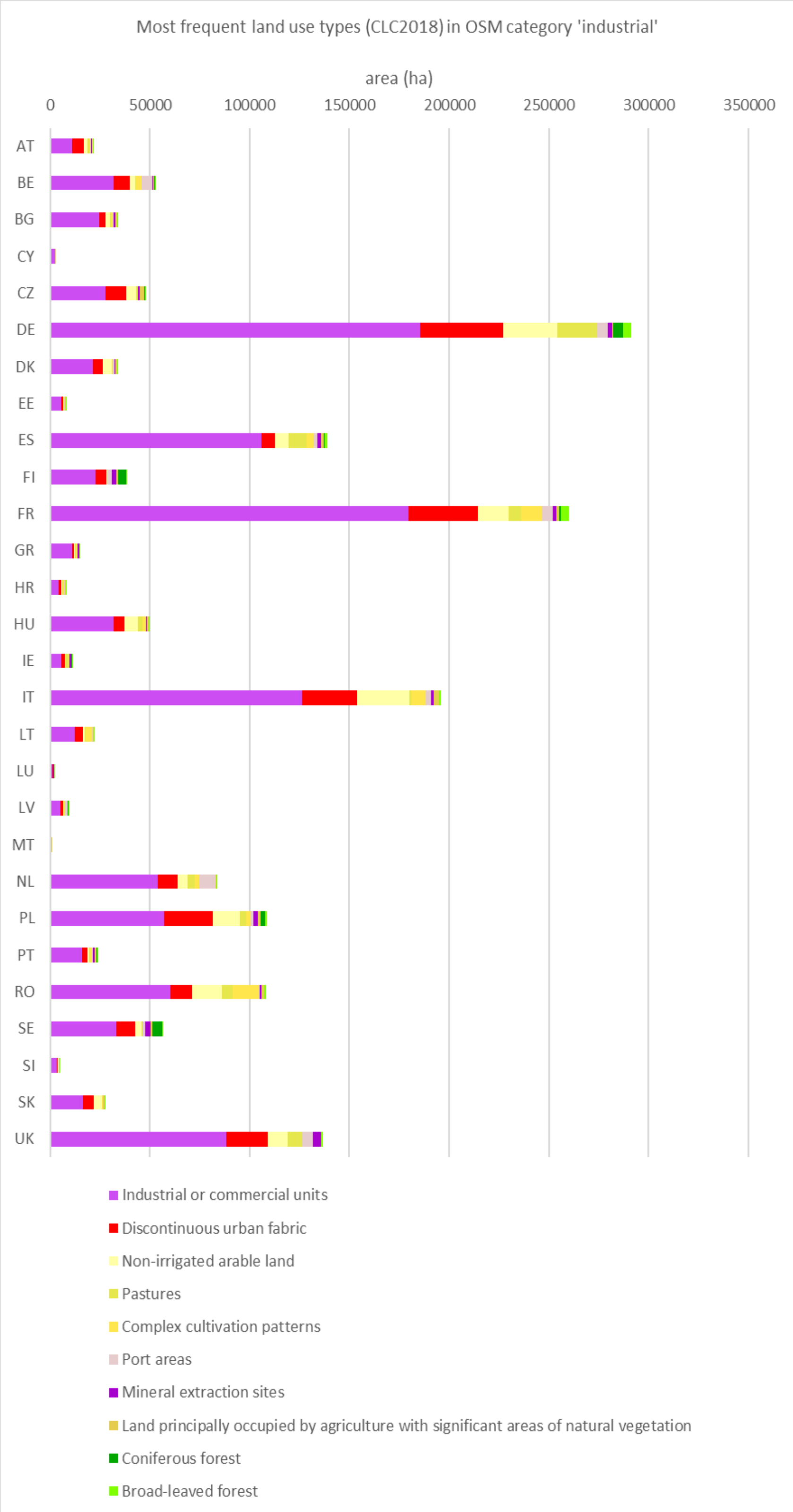


Figure 30 Most frequent land cover types (top 10) in industrial sites detected in OSM. Source data: Open Street Map and CLC 2018.

Quarries

Potential pollutions in quarries

Quarries and mining activities, including open pit mines and below ground mines have caused a large impact on the surrounding areas in view of pollution. Spreading of waste material including processed ores or raw materials like iron sulphides have caused both elevated levels of (mostly) metals in soil or have resulted in extreme acidification because of oxidation of sulphides. Especially the combined effect of acidification in the presence of elevated concentrations of metals in soil has resulted in extreme pollution of both soils and adjacent ground- and surface water bodies. Processing of materials from below ground mining including coal mines locally has resulted in extreme pollution with pollutants like PAHs. Usually, the impact of processing is confined to hot spots at or near the processing site. The impact of waste disposal can affect much larger areas depending on the amount of waste processed and environmental conditions affecting the mobility of contaminants therein

Common examples of below ground and open pit mines in the EU include iron ore mines (typically below ground), brown coal mines (typically open pit mines) and sulphide mines associated with the mining of copper, zinc or lead (typically open pit mines). A recent study requested by the EU parliament shows the considerable social and environmental impact of these types of mines (Mononen et al., 2022).

In most cases pollution levels in mine waste affected soils are such the actual removal of the pollutants is not an option. Reduction of wind and water erosion as can be obtained via phytostabilization seems the most effective way to reduce current risks of affected areas.

Land cover in quarries

In the areas where quarries were identified in Open Street Map, the land cover type on CLC2018 with the largest extent is ‘mineral extraction sites’ (CLC code 131), covering 318,548 ha in the OSM-polygons with quarries in the EU-countries and UK (Figure 31). This land cover type is expected in quarries. Peat bogs were also found in the polygons (76,810 ha, of which 33,536 ha in Finland), which is typical in Scandinavia where peat collection was quite common but is now gradually banned. Considering the need to protect peatland areas, phrased in the new EU Soil Strategy, these areas are less suited to consider for biomass cropping in case they appear to be polluted. Non-irrigated arable land, pastures and transitional woodland take up 108,498 ha in areas designated as quarries. The agricultural land in these categories may be relevant for phytoremediation using bioenergy crops in case soil pollution is detected in the areas of these quarries. This requires an assessment at the level of these sites.

Looking at the distribution of areas with quarries over countries, by far the largest total area is found in Germany (101,227 ha; Figure 32).

The total area of the class ‘mineral extraction sites’ in CLC2018 is 633,848 ha in the EU27 plus UK. Of the areas covered by the class, 50% is not identified as ‘quarry’ in Open Street Map. By country, this share varies from 27% (Czech Republic) to 85% (Netherlands). On the other hand, the total area covered by polygons tagged as ‘quarry’ in Open Street Map, irrespective of the CLC2018 class, is 654,013 ha, and thus larger than the total area of ‘mineral extraction sites’ on CLC2018. The largest absolute differences in areas of both features between OSM and CLC2018 were found in Germany and Finland, where the area in OSM is larger by roughly 18,000 and 43,000 ha, and in Spain and Romania, where the area on CLC2018 is larger by roughly 28,000 and 14,000 ha. This points to the necessity to consult multiple spatial datasets for the purpose of mapping potentially polluted areas by quarries.

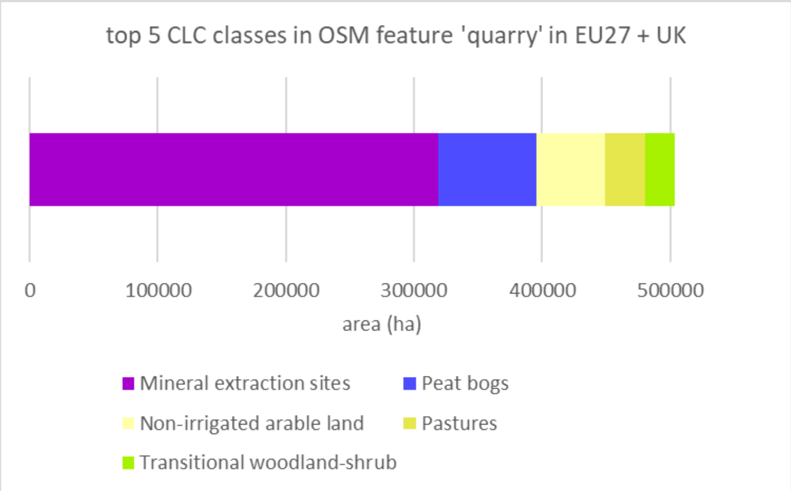


Figure 31 Area in quarries on Open Street Map covered by the 5 land cover classes with the largest areal extent. Source data: Open Street Map and CLC2018.

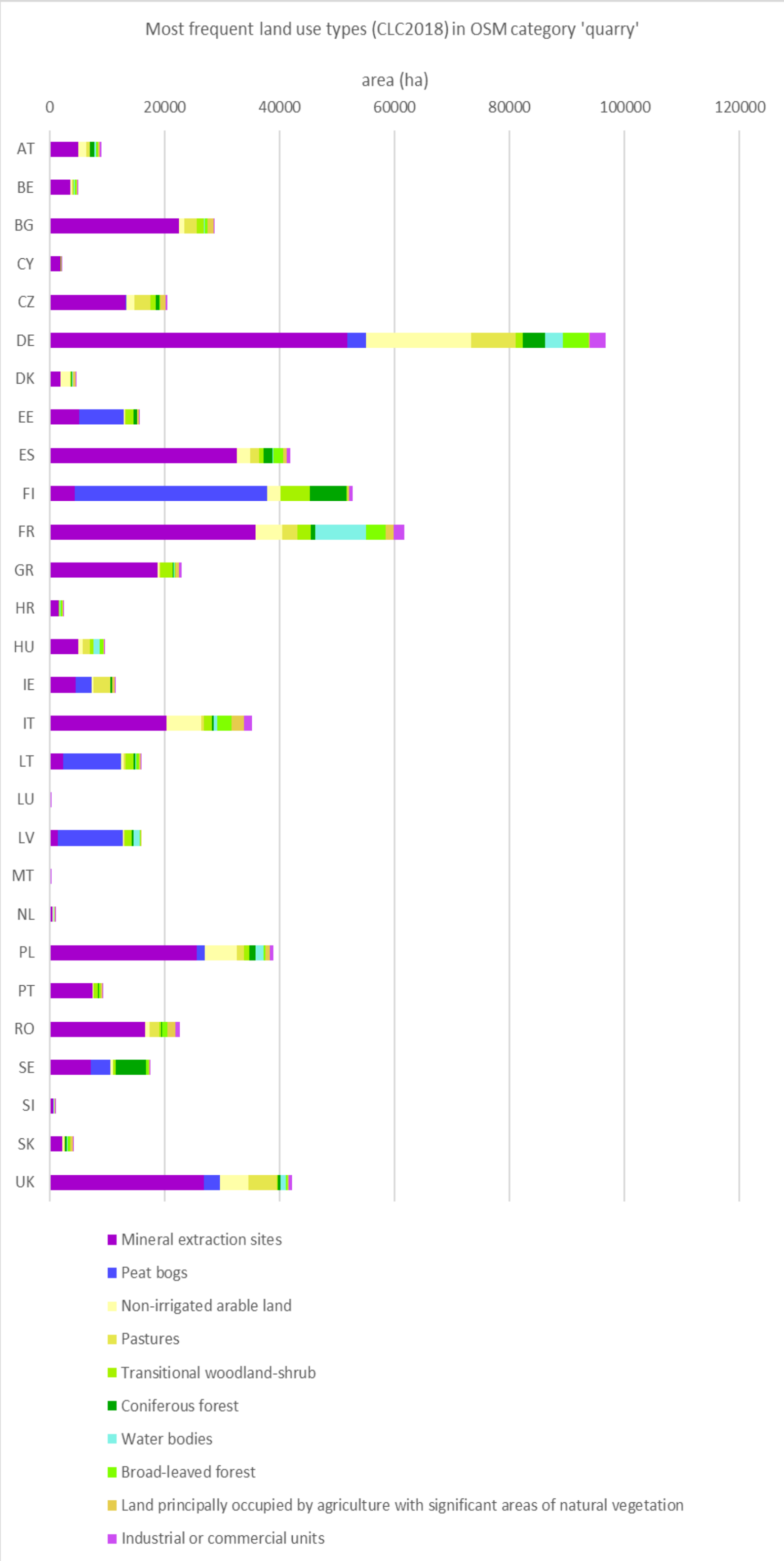


Figure 32 Most frequent land cover types (top 10) in quarries detected in OSM. Source data: Open Street Map and CLC 2018.

Landfill

Potential pollutions in landfills

The impact of landfills present in all member states is restricted to the area of the landfill itself and the immediate surroundings. Both wind erosion (in case of open or active waste disposal heaps) and leaching of contaminants from the bottom of the pile are the most common aspects that need to be addressed to reduce risks arising from such landfills. In addition to such active or recently abandoned landfills, there are numerous historic landfills including small local landfill areas used by local communities. Often the presence of such former sites is not well documented and the presence of the land fill is only recognizable at the location itself in the form of small elevated areas now covered by natural vegetations. The number of such historic sites will increase further in the future due to the reduction of designated landfill sites. In the Netherlands for example the number of formal landfill areas has been reduced from 100 to 20 in the period from 1990 til today (WAR, 2020). This number however is still very small when compared to the number of historic landfill sites in the province of Noord Brabant alone (one of the 12 Provinces in the Netherlands) which is estimated at approx. 600 (website Province Noord Brabant, https://www.brabant.nl/onderwerpen/milieu/bodem-en-stortplaatsen/stortplaatsen).

In most cases dust is the major issue in case of non-covered landfills and the composition thereof strongly depends on the type of waste deposited which can include plastics, metals or organic materials.

Land cover in landfills

Landfills identified in Open Street Map are covered for 37% by dump sites in the CLC2018 map (Figure 33). This is the land cover type that is expected for landfills. The cover with mineral extraction sites (15,485 ha, 15%) might refer to areas where residues from mining operations are piled up next to the mine, and are covered by some form of vegetation. 21% of the areas indicated as landfill in OSM is covered with some form of agricultural land, mainly by non-irrigated arable land and pastures (Figure 33).

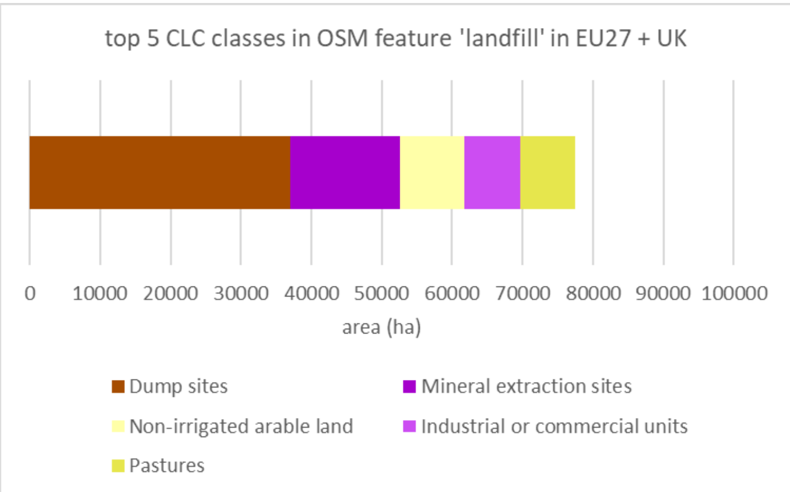


Figure 33 Area in landfills on Open Street Map covered by the 5 land cover classes with the largest areal extent. Source data: Open Street Map and CLC 2018.

Across the EU and UK, the largest areas of landfills are found in Germany (21,932 ha), France (11,379 ha) and Poland (12,412 ha) (Figure 34). Roughly 2/3 of the area labelled as 'landfill' in these countries is covered by dump sites, mineral extraction sites and industrial or commercial units on CLC2018. The area not covered by constructions or disposals is mainly covered by agricultural land use types, forest or transitional forms of natural vegetation (Figure 34). Of these, the agricultural land use types may be relevant for phytoremediation using bioenergy crops, in case soil pollution is present. This requires an assessment at the level of these sites.

The total area of landfills in EU27 and UK on Open Street Map is 99,992 ha, overlapping with 88% of the total area of dump sites on CLC2018, while the total area of dump sites in CLC2018 is 113,095 ha. This might suggest that not all landfills are identified in Open Street Map. However, there are also countries where the total area of polygons tagged as 'landfill' in Open Street Map is larger than the total area covered by dump sites on the CLC2018 map. The difference in area is largest for Germany (4,757 ha on OSM but not classified as 'dump site' on CLC2018) and France (2,707 ha). This points to the necessity to consult multiple spatial datasets for the purpose of mapping potentially polluted areas in or around landfills.

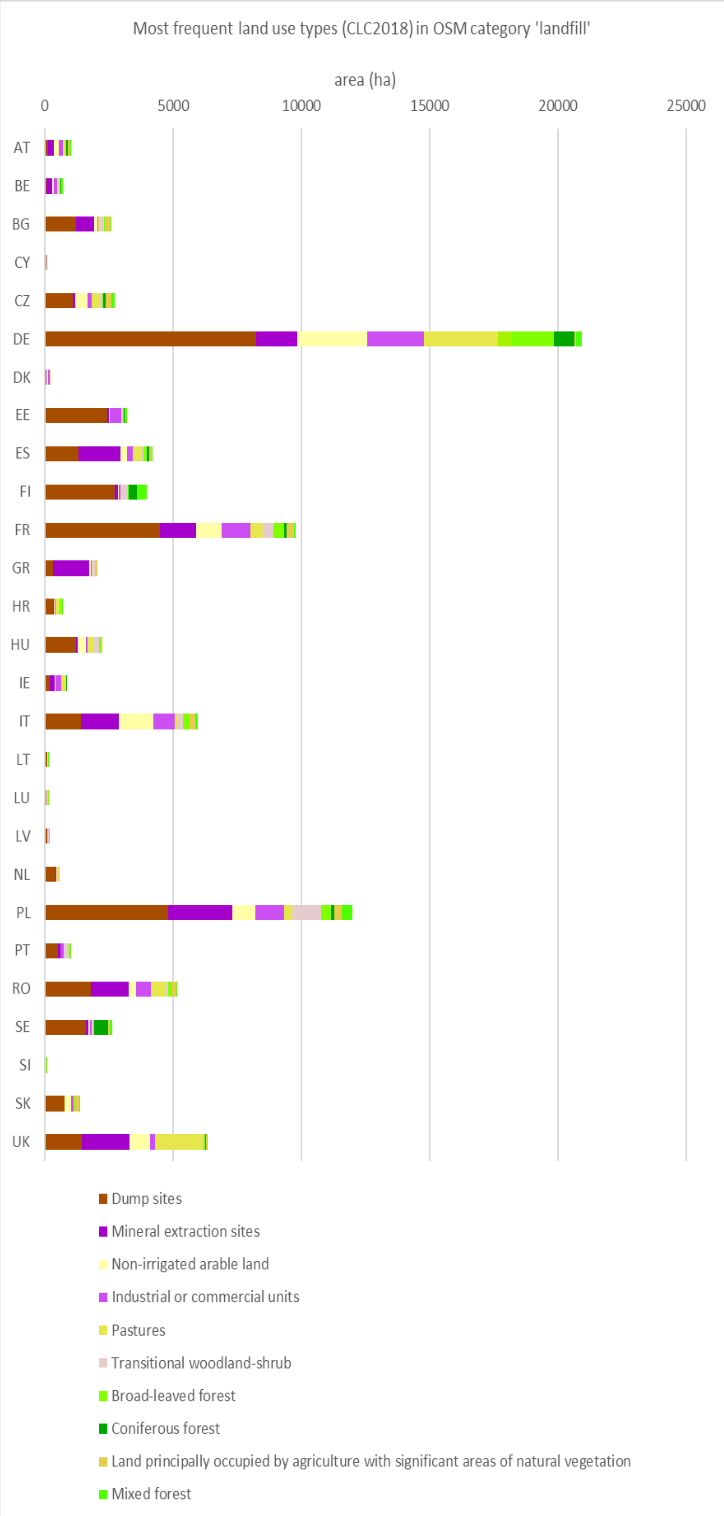


Figure 34 Most frequent (top 10) land cover types in landfills detected in OSM. Source data: Open Street Map and CLC 2018.

Brownfields

The term 'brownfield' is used to designate areas previously in use for industrial activity and where local soil contamination is likely to have occurred. In Open Street Map, areas tagged with land use 'brownfield' in the EU27 and UK cover 70,615 ha. Of these, 23,114 ha is built-up area in use for industrial, commercial or urban purposes (Figure 35). This corresponds to 47% of the total land cover in these areas. These land cover types are expected in areas that were previously in use for industry and where new development is taking place.

The area not covered by buildings, such as brownfields now corresponding to non-irrigated arable land and pastures (almost 7,500 ha), may be relevant for phytoremediation with bioenergy crops if polluting substances are present in the soil.

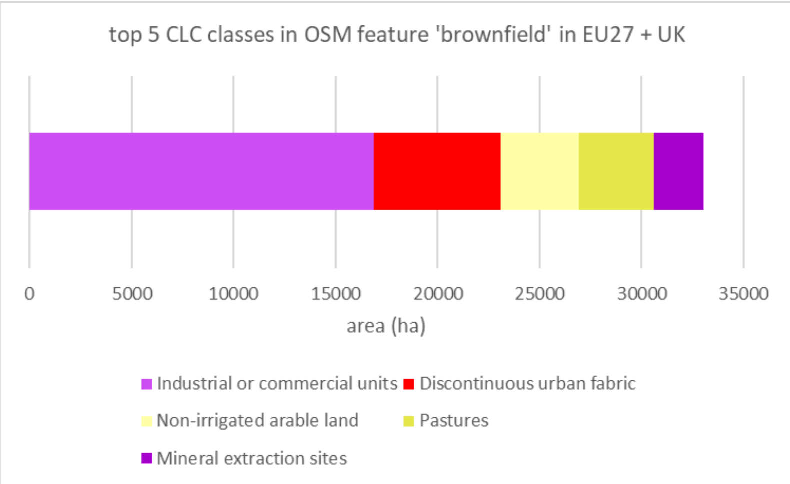


Figure 35 Area in brownfields on Open Street Map covered by the 5 land cover classes with the largest areal extent. Source data: Open Street Map and CLC 2018.

The largest area of brownfields, around 10,000 ha, is found in Germany and the UK (Figure 36). These countries also have the largest areas covered with agricultural land enclosed in brownfields in absolute and relative terms.

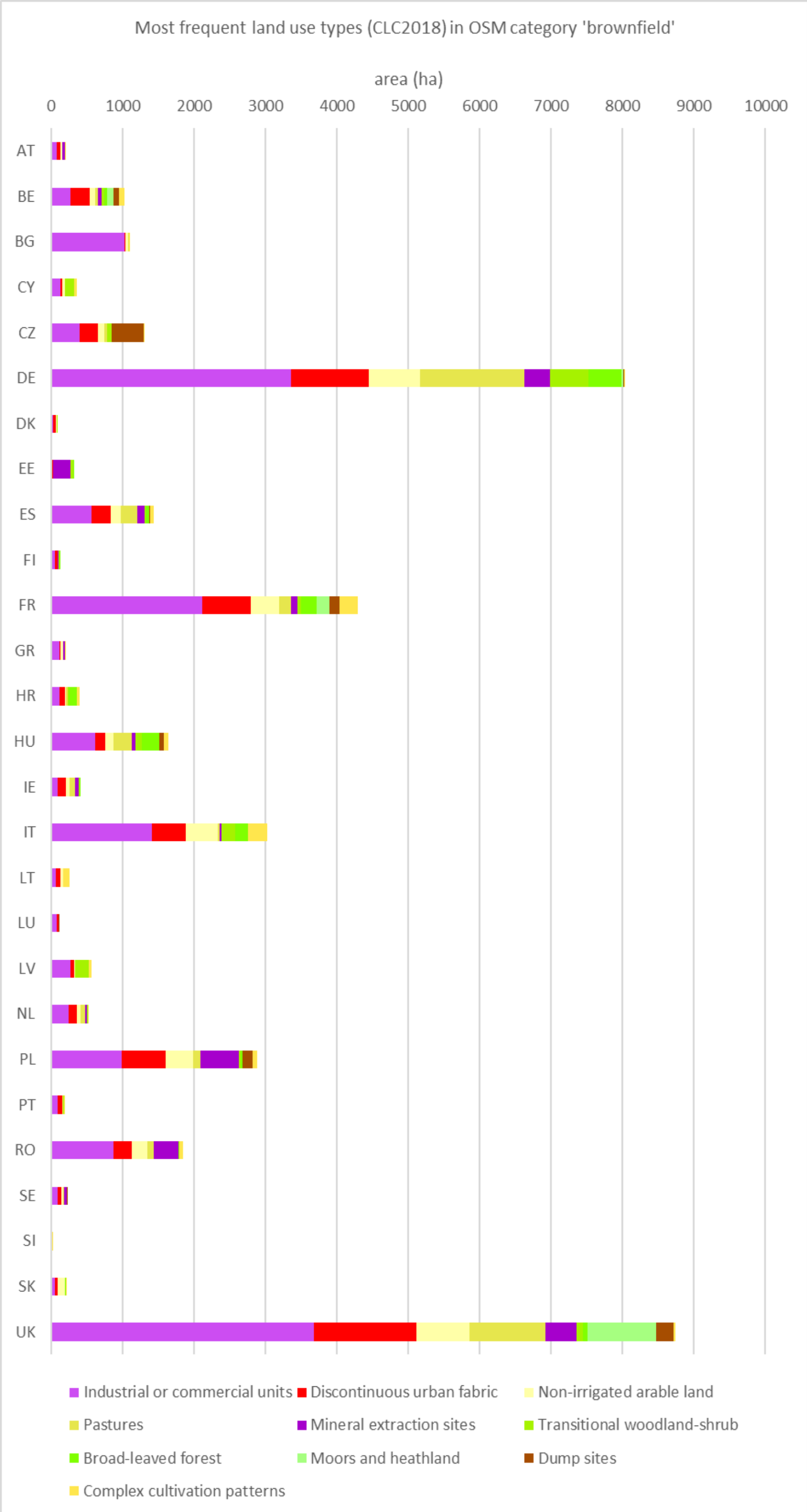


Figure 36 Most frequent land cover types in brownfields detected in OSM. Source: CLC 2018.

Brownfields may be considered a sub-set of industrial areas. In Open Street Map, 66,048 ha was tagged as both types of land use in the EU27 and UK, corresponding to 94% of the total area of brownfields. For the generation of a map of potentially contaminated sites, the polygons tagged as industrial areas and brownfields on Open Street Map were therefore merged. This results in a total of 2,725,502 ha of industrial sites and brownfields, occurring in the EU27 plus the UK.

The areas and distribution across countries of land cover in the industrial sites and brownfields which are relevant for phytoremediation and which concern areas with an imperviousness of <40% are shown in Figure 37. As shown in the figures on industrial sites and brownfields above, the most frequent land cover type are industrial and commercial units, followed by non-irrigated arable land. The largest total areas occur in Romania, France and Germany.

The total area of land in industrial sites and brownfields, that is already in use for some form of agriculture and where imperviousness is <40%, is 167,877 ha. The countries with the largest areas in this category of land cover are Romania (31,640 ha) and Germany (24,981 ha).

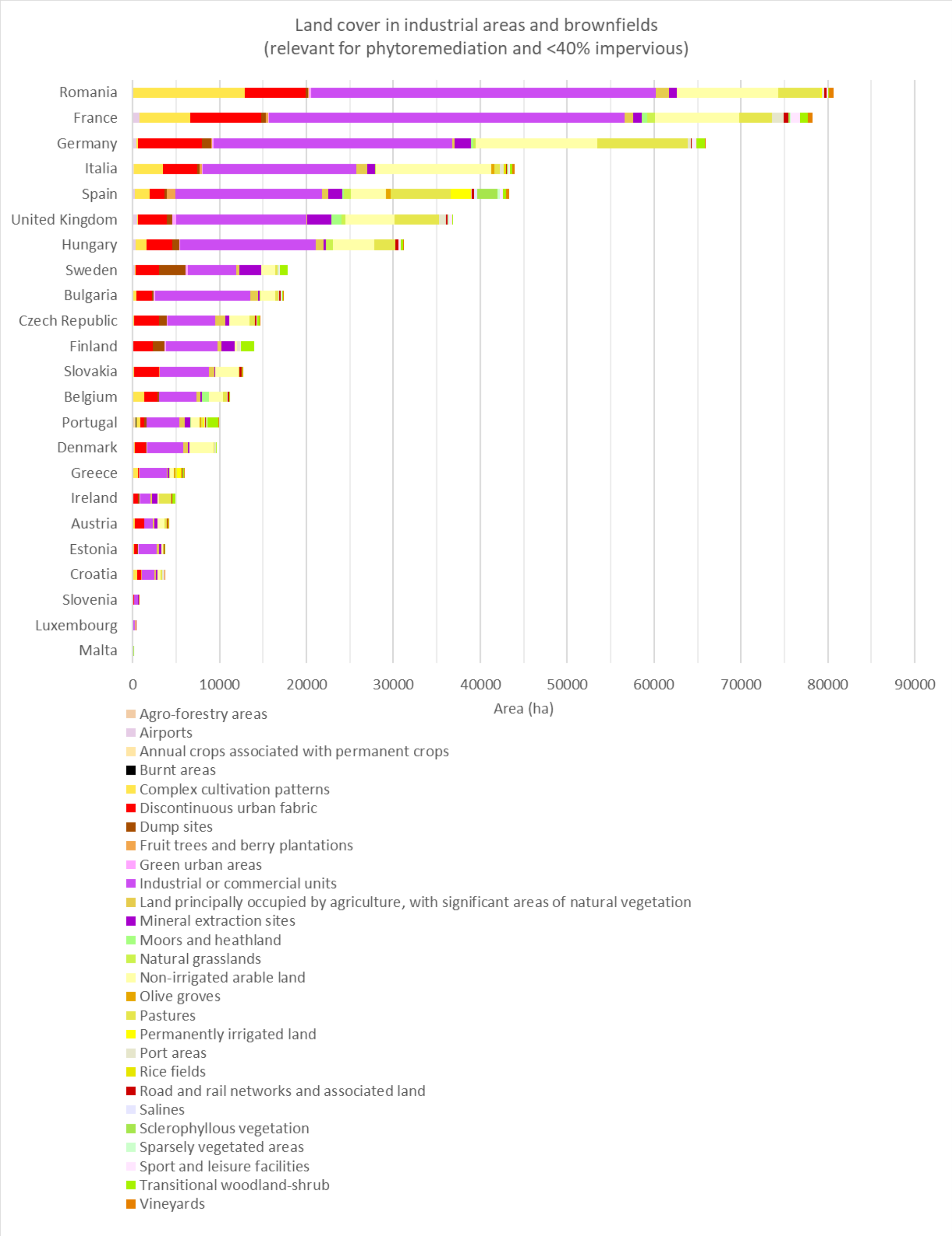


Figure 37 Land cover area in industrial sites and brownfields in the EU27+UK indicated on Open Street Map. Only land cover classes relevant for phytoremediation are shown, and only areas with <40% imperviousness. Source data: Open Street Map, CLC2018, pan-European High Resolution Layer Imperviousness degree (100 m).

5.3 Integration of EU wide OSM & CLC data on potential contaminated sites with other data sources

5.3.1 Mines in EU spatial data

Information on the location and commodities of mines was derived from the electronic Minerals Yearbook in the European Knowledge Base on raw materials. This information was improved in the Minerals4EU database, created in the EU project Mineral Intelligence for Europe (Mintell4EU)[10]. A total of 42.731 mines is included in the Minerals4EU database for 22 EU Member States in 8 commodity groups considered of interest for phytoremediation. The distribution of mines over countries is displayed in Figure 38. The number of mines in Germany is relatively low because only the southern part of the country was covered in the inventory (Figure 39).

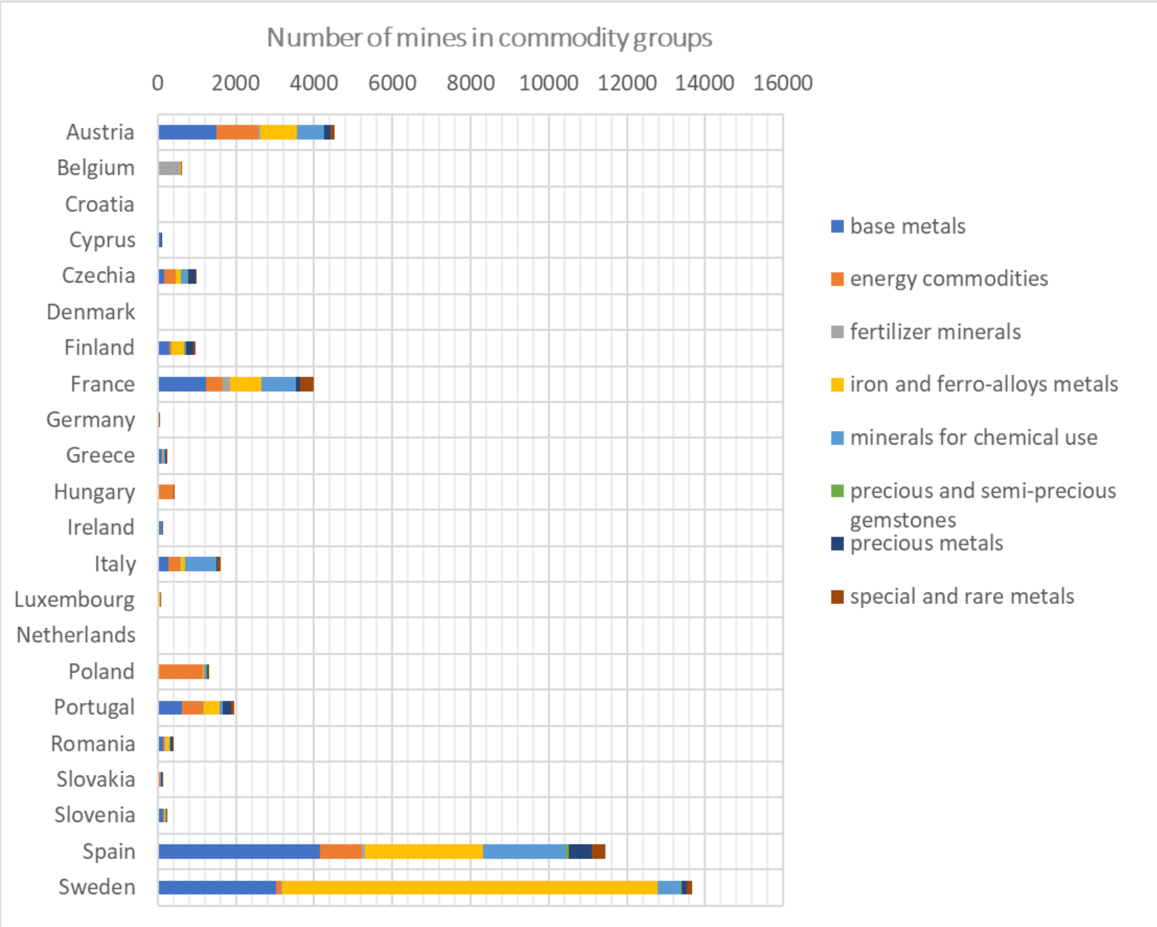


Figure 38 Number of mines in commodity groups in EU countries. Source data: Minerals4EU database.

[10] https://geoera.eu/projects/mintell4eu7/

Figure 39 Visualisation of mine locations in the Minerals4EU database, extract. Source: https://geoera.eu/projects/mintell4eu7/.

The Minerals4EU database does not include information on the area covered by the mines. This information was taken from polygons in Open Street Map (OSM). Polygons were selected from OSM which include one or more point locations of the mines and carry one or more of the 8 land use tags likely to indicate the presence of contaminated sites (selected in section Identification of possibly contaminated sites in Open Street M). A total of 738 mines resulted from this selection.

In order to create an overview of potentially polluted area suitable for phytoremediation using biomass crops, these mines were further explored on their average area per commodity group, their current land cover and the type(s) of commodity produced. The results of these analyses are presented subsequently in sections Combining mines in EU with OSM and CLC data and Risks of mining commodities for human health and suitability for phytoremediation.

5.3.2. Combining mines in EU with OSM and CLC data

Mines in the Minerals4EU database that were identified in Open Street Map are listed according to commodity group in **Error! Reference source not found.**. Most mines were observed in the commodity group base metals. The area covered by the mines according to the polygons identified in OSM varied widely, ranging from 1 to 34.543 ha for all commodity groups. The largest mines (>30.000 ha) were found for the commodity groups iron and ferro-alloys metals and minerals for chemical use, the smallest for precious and semi-precious gemstones (diamond) (<20 ha).

For the estimation of the potentially polluted area around mines, an average area of mines was inferred for each commodity group. This was done by inspecting the frequency distribution of the areas of mines (example in Figure 40). The average area of mines in each commodity group was calculated as the average of the area interval that included >90% of the mines in the group. The area intervals and average mine areas for the commodity groups are listed in the last two columns of Table .

Mines in the commodity groups base metals, energy commodities, fertilizer minerals, iron and ferro-alloys metals and minerals for chemical use cover around 100 ha according to the indications on Open Street Map. The area of mines where precious metals are extracted (gold, silver, platina) is an order of magnitude larger on average (1045 ha). Average areas of mines for precious and semi-precious gemstones and special and rare metals are small compared to the other mines (<50 ha).

Table 5-3 Area of mines(in ha) as indicated in OSM for various commodity groups. Source data: OSM and Minerals4EU database.

Commodity group	Total nr of mines in OSM polygons	Average area of mine in OSM-polygon (all included) (ha)	Minimum area of mine (ha)	Maximum area of mine (ha)	Standard deviation of area (ha)	Area interval with >90% of mines (ha)	Average area of mines in <90% interval (ha)
base metals	221	754	1	23.551	2334	1-1400	106
energy commodities	154	609	1	12.344	1866	1-1200	117
fertilizer minerals	47	423	1	5389	1315	1-1300	85
iron and ferro-alloys metals	151	1375	1	34.543	4058	1-2700	117
minerals for chemical use	102	477	1	33.644	3365	1-2500	99
precious and semi-precious gemstones	12	15	6	18	3	6-18	15
precious metals	35	1965	2	23.551	4558	2-4900	1045
special and rare metals	16	364	4	5099	1264	3-1800	48
Total	**738**						

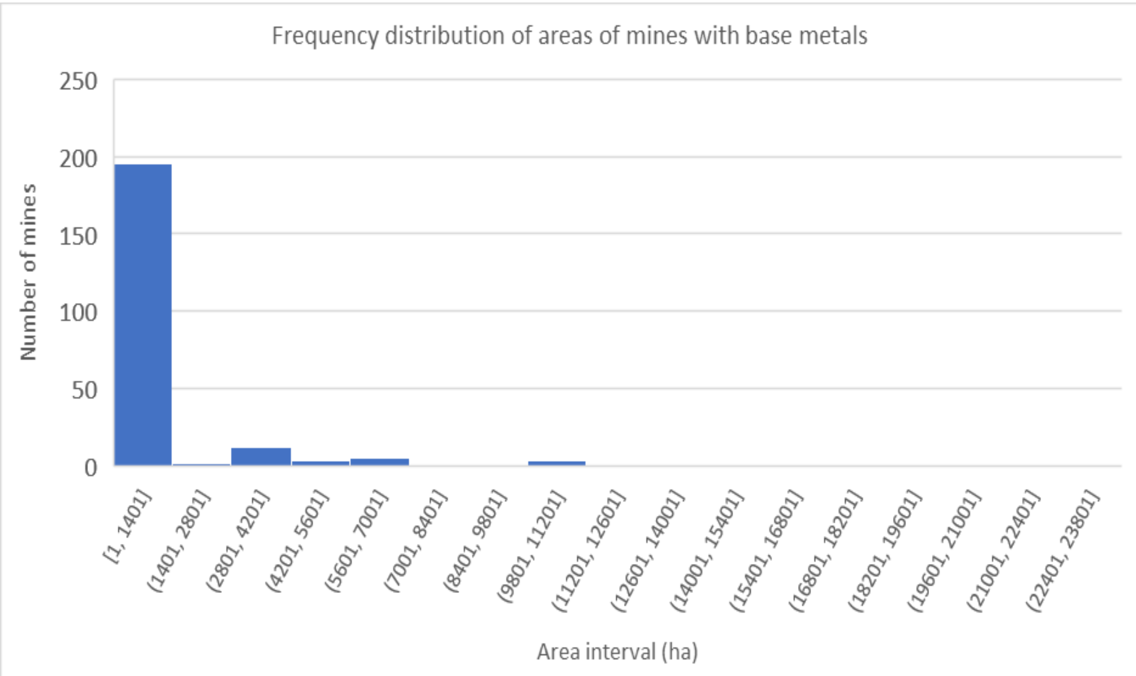


Figure 40 Frequency distribution of mine area (in ha) in commodity group 'base metals'. Source data areas: Open Street Map. Source data on number of mines: Minerals4EU database.

The current land cover in mine areas is relevant for the investigation of areas potentially suitable for phytoremediation. Built-up areas and areas covered with vegetation or ecosystems that should not be converted to biomass crops are not considered suitable and are excluded from the inventory.

The areas of mines in commodity groups and land cover classes considered suitable for phytoremediation are listed in **Error! Reference source not found.**. The table includes counts of mines from the Minerals4EU database in commodity groups relevant for phytoremediation (Table), that were identified in one of the land use tags for potentially contaminated sites in Open Street Map (OSM). The table also includes counts of mines that were not identified in OSM.

Of the mines identified on OSM, the largest total area was observed in polygons tagged as 'industrial' (196) or 'quarry' (314 ha). This gives some confidence in the identification of mines in OSM. However, many mines in the Minerals4EU database (20,137) is not identified in OSM, and of this number, only 204 are indicated as mines in the land cover class 'mineral extraction sites' in the Corine Land Cover database (class nr 7). This shows us that for a more accurate estimation of potentially contaminated sites, dedicated databases are required with spatial information on geographical objects associated with local contamination, and that we cannot rely solely on topographical information from Open Street Map and land cover information that cover the European domain.

The largest numbers of mines in commodity groups relevant for phytoremediation (>2,000) were found in the Minerals4EU database in the land cover classes 'non-irrigated arable land', 'pastures' and 'transitional woodland-shrub' on the CLC2018 map (Table). Of the total of 20,708 mines observed in land cover classes considered relevant for phytoremediation, almost half (10,206) are located in areas with agricultural land use (CLC classes 12-22). This offers potential for options to use existing agricultural land for biomass crop production.

Table 5-4 Area of mines (ha) potentially suitable for phytoremediation in combinations of CORINE land cover classes (CLC2018) and land use tags identified in Open Street Map (OSM). Areas of mines in selected CLC classes that were not identified in OSM are given in column 'No polygon in OSM'. Source data: land cover: CLC2018 (Copernicus Land Monitoring Service); occurrence of mine locations: Minerals4EU database; areas of mines: Open Street Map.

			OSM land use tag					
CLC class	Description	No polygon in OSM	Brownfields	Industrial	Landfill	Military	Quarry	Total
2	Discontinuous urban fabric	1198		26		1	5	1230
3	Industrial or commercial units	151	1	68	1	2	4	227
4	Road and rail networks and associated land	41						41
5	Port areas	1						1
6	Airports	9				1		10
7	Mineral extraction sites	204	1	36	13		174	428
8	Dump sites	39		15	7		3	64
10	Green urban areas	26	3	1				30

			OSM land use tag					
CLC class	Description	No polygon in OSM	Brownfields	Industrial	Landfill	Military	Quarry	Total
11	Sport and leisure facilities	97		2	1		2	102
12	Non-irrigated arable land	2967		11	2		30	3010
13	Permanently irrigated land	92						92
14	Rice fields	1						1
15	Vineyards	167						167
16	Fruit trees and berry plantations	127			1			128
17	Olive groves	725		1			1	727
18	Pastures	2169		4	1		13	2187
19	Annual crops associated with permanent crops	94				1		95
20	Complex cultivation patterns	1220		6	1		8	1235
21	Land principally occupied by agriculture with significant areas of natural vegetation	1378		7		1	6	1392
22	Agro-forestry areas	1170					2	1172
26	Natural grasslands	1889				6	15	1910
27	Moors and heathland	1455	1	3		3	9	1471
28	Sclerophyllous vegetation	1929		13		3	11	1956
29	Transitional woodland-shrub	2297	1	2		2	26	2328
32	Sparsely vegetated areas	617				6	5	628
33	Burnt areas	49				1		50
38	Salines	25		1				26
Total		**20137**	**7**	**196**	**27**	**27**	**314**	**20708**

5.3.3. Risks of mining commodities for human health and suitability for phytoremediation

From the Minerals4EU database a sub-selection of 65 commodities was made that is produced in the mines. These commodities were ranked according to two aspects:

- The risk of the commodity for human health (and the need to reduce the risk) and the possibility to manage the mining site with biomass crops, such that risks of the commodity for human health are reduced. If the risk for human health of the commodity is high and can be remediated to some extent by growing plants, a score of 1 is attributed. If the risk for human health is low, for example in case of a sand pit, and therefore the need to remediate the commodity is low too, a score of 3 is attributed. A score of 2 indicates positions in between.

- The suitability of phytoremediation as a means to manage the commodity. The ranking in this aspect is as follows:
  1. How likely is it that plants can remove the chemical listed from the soil through extraction?
  2. How likely is it that by using plants (in combination with other chemicals), the compound of interest can be immobilized such that risks are reduced?
  3. How likely is it that plants are able to assist in in situ degradation of the compound of interest?

The scoring for these ranking is: 1: likely or proven, 2: unknown or questionable, 3: most likely not effective.

The scores for all commodities on both aspects are presented in Annex 2. We estimated areas of mines with the scores on levels of risk, need and suitability for phytoremediation by combining the numbers of mines in each combination of aspects with the typical area of the commodity group to which the mine belongs from Table . The total area of mines so obtained from the Minerals4EU database is 6.208.776 ha. This number is one order of magnitude higher than the total area of quarries indicated on Open Street Map (654.013 ha) and the total area of mineral extraction sites in CORINE Land Cover 2018 (633.848 ha). This indicates that the area of mines cannot be inferred from the typical size area (Table) per commodity group. For this reason, we present the assessment of risk for human health and suitability of mines for phytoremediation in numbers of mines in classes, instead of in area covered.

Table shows that the largest number of mines (24.074, 57% of the total number) is in the category of mines with high risk and need to remediate contamination by the commodities produced in the mines, and where phytoremediation might be possible to reduce the risk (score 1). Another 40% of the mines (17.869) is in the category where commodities do not pose a high risk for human health, and where consequently the need to apply remediation is low (score 3). Our interest is in the area contained in the first mentioned category of mines.

If we look at the possibility to apply different modes of phytoremediation for this category, we find that the largest numbers of mines are estimated likely to treat with phytoremediation through extraction or stabilization of the commodities (Table). There is less potential for or degradation/ volatilization of the compound by planting vegetation. The numbers with possibility for extraction or stabilization correspond to resp. 28% and 37% of the total number of mines in the Minerals4EU database.

Table 5-5 Numbers of mines with rankings of level of risk, need and possibility to phytoremediate for all commodities.

Level of risk, need and possibility to phytoremediate	Number of mines
High (1)	24.074
Medium (2)	702
Low (3)	17.869
Total	42.645

Table 5-6 Numbers of mines with scores on level of risk, need and possibility to phytoremediate commodities in mines for phytoremediation by extraction, stabilization or degradation/ volatilization.

Number of mines	Likeliness of phytoremediation by extraction		
Level of risk, need and possibility to phytoremediate	Likely (1)	Unknown/ questionable (2)	Not effective (3)
High (1)	12087	5899	6088
Medium (2)	600		102
Low (3)	724	12708	4437

Total	13411	18607	10627

Number of mines	Likeliness of phytoremediation by stabilization		
Level of risk, need and possibility to phytoremediate	Likely (1)	Unknown/ questionable (2)	Not effective (3)
High (1)	15792	6334	1948
Medium (2)		600	102
Low (3)	334	186	17349
Total	16126	7120	19399

Number of mines	Likeliness of phytoremediation by degradation or volatilization		
Level of risk, need and possibility to phytoremediate	Likely (1)	Unknown/ questionable (2)	Not effective (3)
High (1)	199	266	23609
Medium (2)	102		600
Low (3)	1417		16452
Total	1718	266	40661

5.3.4. Combining industrial sites and dump sites with the locations of the known blast furnace steel industries

A special category within industrial sites that may deliver pollution risks are steel factories with blast furnaces (Blast Furnace-Basic Oxygen Furnace, BF-BOF). In these sites, iron is produced from iron ore. Carbon is used to separate iron from oxygen. Approximately 60% of steel in the EU is produced via this production route (EUROFER, 2020).

In the production process, CO2 and fine particles are emitted, but pollution of soils due to deposition of fine particles has not been demonstrated. The risk for human health in this production process that is proven is the inhalation of fine particles which are emitted by the furnaces. It is conceivable that vegetation might be used to prevent further distribution of particulate matter in the vicinity of the steel production sites with blast furnaces. For this reason we consider land use in the area of 5 km distance from the production sites.

Locations of the production sites were obtained from the European Steel Association (EUROFER) (Figure 41). Land cover within 5 km from the sites was derived from CLC 2018. Total areas per class for the 27 production sites are displayed in Figure 42. Only areas were considered with land cover relevant to phytoremediation (Table) and with an imperviousness <40%. The imperviousness of the soil surface was derived from the pan-

European High Resolution Layer 'Imperviousness degree' (IMD) (pixel size 100 m, reference year 2018)[11] of the Copernicus Land Monitoring Service. 60% of the area considered (and 31% of the total area around the production sites) currently has land cover reflecting agricultural use. This might offer potential to deploy the area for stabilization of fine particulate matter by biomass crops.

Location	Hot Metal Capacity ('000 tonnes/ year)	Finished Steel Capacity ('000 tonnes/ year)	No. of furnaces
AUSTRIA			
DONAWITZ (Leoben)	1370	1570	2
LINZ	4340	6000	3
BELGIUM			
GHENT	4430	5000	2
CZECH REPUBLIC			
OSTRAVA	3200		3 – BF only
TRINEC	2100	2400	2
FINLAND			
RAAHE	2400	2600	2
FRANCE			
DUNKERQUE	6800	6750	3
FOS-SUR-MER	5160	5100	2
GERMANY			
BREMEN	3960	3800	2
DILLINGEN	4790	2760	2
DUISBURG	11600	11560	4
EISENHÜTTENSTADT	2340	2400	2
SALZGITTER	4800	5200	3
VÖLKLINGEN		3240	BOF only
HUNGARY			
DUNAUJVAROS	1310	1650	2
ITALY			
TARANTO	9590	11500	4
NETHERLANDS			
IJMUIDEN (Velsen-Noord)	6310	7500	2
POLAND			
DABROWA GORNICZA	4500	5000	2
KRAKOW	1310	2600	1
ROMANIA			
GALATI	3250	3200	2
SLOVAKIA			
KOSICE	2850	4500	2
SPAIN			
AVILES		4200	BOF only
GIJON	4480	1200	2
SWEDEN			
LULEA	2200	2200	1
ÖXELÖSUND	1800	1700	2
UNITED KINGDOM			
PORT TALBOT	4770	4900	2
SCUNTHORPE	3590	3200	3

Figure 41 Locations of steel production sites with blast furnaces. Source: EUROFER (www.eurofer.eu).

[11] https://land.copernicus.eu/pan-european/high-resolution-layers/imperviousness

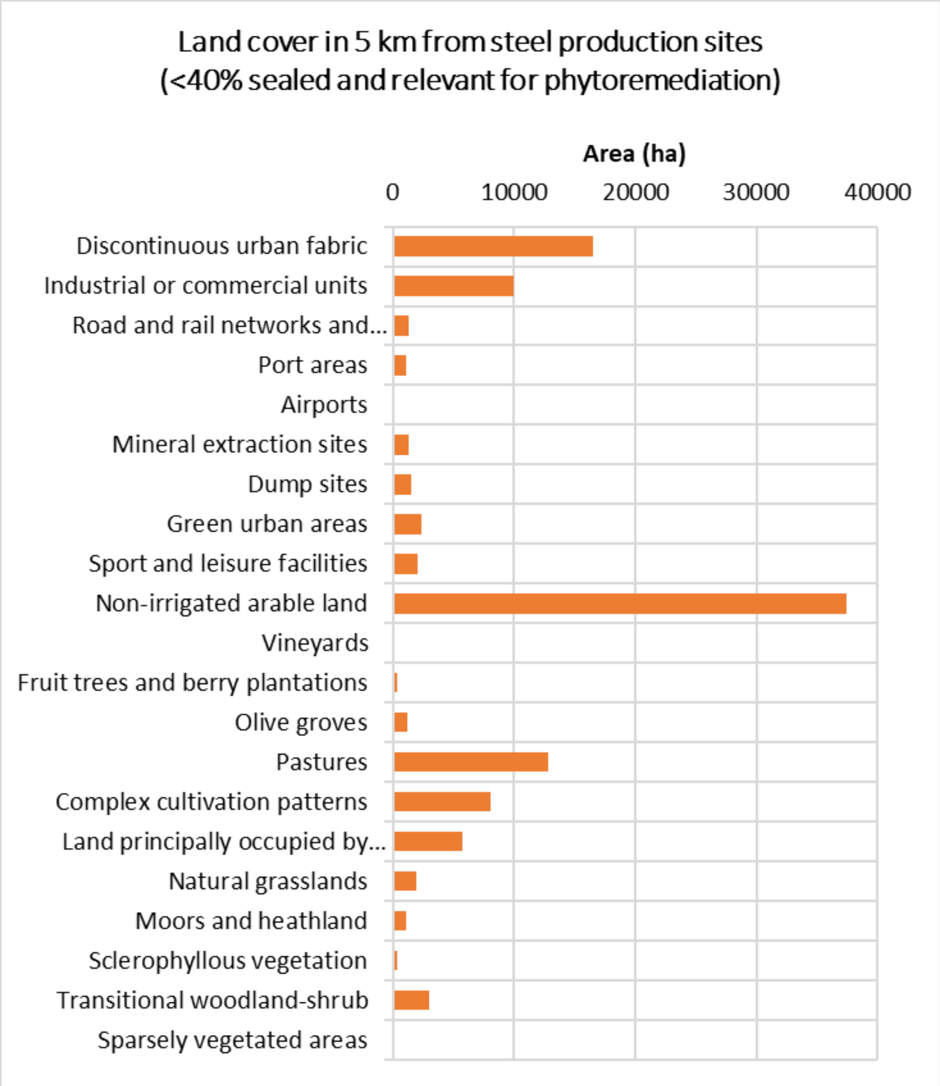


Figure 42 Land cover in an area within 5 km distance from 27 steel production sites with blast furnaces in Europe, with <40% soil sealing. Source data: CLC2018 and pan-European High Resolution Layer 'Imperviousness degree' (IMD) (pixel size 100 m, reference year 2018)[12] of the Copernicus Land Monitoring Service.

[12] https://land.copernicus.eu/pan-european/high-resolution-layers/imperviousness

5.4 Integrated spatial database on contaminated sites

A summary of the mapping of potentially contaminated sites in the EU27 and UK using the data sources presented above is given in Table . The total area estimated in potentially contaminated sites due to military training activities, industrial activities, mining and landfills, of which less than 40% is sealed, amounts to 2,013,722 ha. This corresponds to 0.5% of the total area of the countries considered. In individual countries, the area of potentially contaminated sites identified on Open Street Map is at most 1% of the total surface area of the country. France, Germany and Spain have the largest total areas of all types of potentially contaminated sites, amounting to more than 150,000 ha in each of the countries.

France and Germany are in the top 3 of countries with the largest total area in all types of potentially contaminated sites, with total areas between 10,000 and 290,000 ha. In the category of military sites, United Kingdom is also in the top 3 countries with the largest total potentially contaminated area, amounting to almost 100,000 ha. For industrial sites and brownfields, the total area is largest in Romania, with more than 80,000 ha. For quarries, Spain follows behind France and Germany with almost 50,000 ha. Poland, France and Germany have the largest total area of landfills (between 10,000 and 19,000 ha).

Table 5-7 Total area (in ha) of potentially contaminated sites identified in Open Street Map, with land cover types relevant for phytoremediation and less than 40% imperviousness. Source data: OSM, CLC2018, HRL IMD2018.

Area (ha)	Type of potentially contaminated site					
Country	**Military**	**Industrial & brownfields**	**Quarries**	**Landfills**	**Total area**	**Total area country**
Austria	17.551	4.157	8.423	927	31.058	8.387.900
Belgium	13.777	11.208	4.948	739	30.672	3.052.800
Bulgaria	9.973	17.400	30.418	2.462	60.253	11.037.000
Croatia	10.217	3.679	2.352	708	16.956	5.659.400
Czech Republic	42.335	14.581	21.100	2.504	80.520	7.886.800
Denmark	21.218	9.711	4.306	181	35.416	4.292.400
Estonia	8.538	3.733	7.523	3.163	22.957	4.522.700
Finland	48.932	13.971	14.586	3.844	81.333	33.844.000
France	126.163	78.260	55.948	10.164	270.535	63.318.660
Germany	118.826	65.857	84.051	18.652	287.386	35.737.600
Greece	15.373	5.904	25.319	2.325	48.921	13.204.900
Hungary	14.785	31.178	9.151	2.311	57.425	9.301.100
Ireland	2.253	4.924	8.605	821	16.603	6.979.700
Italia	36.462	43.963	37.393	5.967	123.785	30.207.300
Latvia	5.720	4.973	3.287	180	14.160	6.457.300
Lithuania	5.207	12.962	5.281	146	23.596	6.528.600
Luxembourg	21	355	197	201	774	258.600
Malta	10	55	280		345	31.540
Netherlands	11.026	19.076	810	783	31.695	4.154.000
Poland	56.571	44.248	36.234	11.003	148.056	31.267.900

Portugal	9.299	9.873	9.976	1.094	30.242	9.222.600
Romania	16.755	80.607	22.453	5.401	125.216	23.839.070
Slovakia	14.641	12.640	3.543	1.302	32.126	4.903.500
Slovenia	1.177	737	863	61	2.838	2.027.300
Spain	90.160	43.326	49.973	4.588	188.047	50.594.400
Sweden	39.404	17.848	10.261	2.051	69.564	43.857.400
United Kingdom	97.519	36.874	42.057	6.793	183.243	24.361.000
Total area	833.913	592.100	499.338	88.371	2.013.722	

6. Conclusions and further steps

6.1 Introduction

This report presents the results of the first part of task 3.1 in GOLD that aims at mapping in detail the contaminated sites in the EU and their characteristics. A distinction is made between sites/or rather areas affected by diffuse and by point source pollution:

1) Diffuse pollution (def. EEA: ***Pollution*** *from widespread activities with no one discrete source, e.g. acid rain, pesticides, urban run-off, etc*.)

2) Point source pollution (def. EPA: *Pollution from any single identifiable source (e.g. landfill, mine, industrial site)*

Areas affected by diffuse pollution cover a larger area than sites affected by point source pollution. For diffuse pollution the source of pollution is not clear either and can be caused by sources like traffic, industry (both leading to deposition via air) as well as agriculture. Point source pollutants often have a direct link to a specific industry or activity, like mining, and usually leads to a limited number of pollutants present but at high levels. The pollutions from diffuse sources can range from metals (in fertilizers and manure) to nutrients (N and P), biocides, persistent organic pollutants present in sludge applied to land as well as soil acidifying substances like ammonia emitted from nearby intensive animal husbandry farms.

For contaminated sites, the aim here is to map areas of sites in the EU that are contaminated to some degree, that need cleaning or stabilisation and that may be suitable for bioremediation through the cultivation of biofuel crops. In the following the main conclusions in relation to the spatial identification of areas affected by diffuse and points source pollution is presented.

6.2 Conclusion for bioremediation of diffuse pollution sites

A method was developed and applied to estimate risks from diffuse pollution from various substances for human health and ecosystems, and to map these risks for Europe in terms of the deviation of the current content of the substance to a critical limit for each substance. Maps were generated for cadmium, lead, copper and zinc. Input data for the analysis include soil organic carbon and clay contents and pH and actual contents of the metals considered. These data were derived from ESDAC and SoilGrids .

Now this approach is limited to a few selected metals for which we have the requested information. For food this approach is, for now, limited to Cd. For Pb the relation between soil and crops is poor so we cannot predict at what levels in soil food quality criteria are exceeded. For other metals specific health-based quality criteria only exist for mercury (Hg) and arsenic (As) but this refers almost exclusively to products of marine or freshwater organisms (fish, mollusks etc.). Also, there is growing concern about potential food safety issues related to organic emerging pollutants including pesticides, antibiotics and flame retardants (a.o. PFAS). For most of these substances (with the exception of plant protection chemicals or metabolites thereof) food quality criteria are not available. Also, there is a lack of reliable soil to crop transfer models. Finally, also spatial data to prepare maps at the desired scale level are lacking since, as of now, there is no systematic monitoring of most of these substances.

The resulting maps of deviations of the four metal contents from the critical limits vary between the two data sources. Uncertainties in both sources should be taken into account in the interpretation of the result maps. Basically, the results included in this report are to be used as an example of the approach. It is impossible to generate exact data on the extent of the areas where current metal concentrations exceed critical limits in view of food safety, ecology or water quality. This requires a more in-depth analysis of the quality of input data and models used as well. So, work will continue to improve the models and to obtain a better understanding of the influence of the input data on the final results.

A qualitative assessment of the maps provided nevertheless shows that there are specific areas where a combination of soil conditions and metal concentrations in soil is such that it is likely that critical limits are exceeded.

In case of food safety related to cadmium (Cd) uptake by wheat, soils in Central Europe including areas in the Netherlands and Belgium can be considered vulnerable. This is largely due to a combination of soil type (sandy soils, loamy soils) with initial low pH levels. Liming has increased soil pH to some extent but a majority of these soils can be prone to acidification once liming is stopped. In addition, in areas in Belgium, the Netherlands and Poland, regional industrial pollution has contributed to the increase of the initially low Cd concentrations in soils. This in combination with low soil carbon and moderate soil pH results in the regional exceedance of critical concentrations of Cd in soil. These low to moderate pH soils are also vulnerable in view of water quality although the risk of Cd leaching from soil to groundwater is reduced substantially during the transport from upper soil to groundwater. However, Cd leaching to groundwater has been reported especially in areas affected by proximity pollution, i.e. diffuse pollution caused by industry. This is the case both in the Dutch Belgian border area (Kempen) as well as in the heavily industrialized areas in southern Poland.

Risks in view of ecotoxicology are noticeable in Italy and Greece and areas in France and Spain with viniculture. In most cases a combination of higher natural concentrations of copper in soil and the extensive use of plant protection chemicals that contain copper are the reason for this exceedance.

For cadmium (Cd) and lead (Pb) risks in view of ecotoxicology seem minor, for both metals the calculated critical concentrations appear to be substantially higher than current concentrations in soil. For zink (Zn) exceedance of the calculated critical limit is confined to areas with low pH soils similar to those with a reported exceedance of the critical limit for cadmium (Cd) in food.

6.3 Conclusions for potentially contaminated sites

Enquiry at JRC-ESDAC and consultation of the websites of EEA and Eurostat revealed that at present, there is not database of contaminated sites for Europe that carries spatially referenced information on area and contaminants. The most recent Europe-wide assessment of contaminated sites is the JRC Technical Report Status of local soil contamination in Europe by Payá Pérez & Eugenio (2018), which was based on questionnaires to experts of national reference centres (NRCs) in the EEA-member countries. I revealed that of the 39 countries surveyed, 28 maintain comprehensive inventories for contaminated sites at national or regional level. It became clear that in total 65,000 sites had been remediated or are under aftercare, and 650,000 sites are registered as sites where polluting activities took or takes place (Payá Pérez & Eugenio, 2018).

Because of lack of EU wide spatially explicit sources on contaminated sites an other approach to mapping these contaminated sites was developed. For this reason, potentially contaminated sites were identified first in Open Street Map (OSM). These were reviewed by comparing the sites against polluted site locations that were known to WR-experts, to current land cover as represented on CLC2018 and - for mines only - to locations in the Minerals4EU database.

From OSM the following sites were selected with presumed potential to have soil pollution present:

- (former) quarries and mine tailings
- (former) land fill sites
- (former) military sites
- former industrial sites (brownfields)
- industrial sites
- harbours
- wastewater treatment plants
- fuel stations

Conclusions from the review of OSM in combination with other spatial data sources are:

- The largest areas of potentially contaminated sites are in areas tagged on OSM as military sites (41%), industrial sites and brownfields (29%), quarries (25%) and landfills (4%). The total area on land covered by these sites is 4,718,773 ha. Harbours, wastewater treatment plants and fuel stations cover minor areas compared to these categories. This total area has been further reviewed for suitability for phytoremediation by overlaying it with additional spatial data sources.
- The land cover from CLC2018 in the considered OSM sites corresponded to the expected land cover for the larger part, i.e. forest and other semi-natural vegetation for military sites, industrial or commercial units for industrial sites and brownfields, mineral extraction sites for quarries and dump sites for landfills. This supports the correct selection of the sites in OSM.
- In sites where pollutants may occur, land cover consisting of densely built-up area, forest or other natural vegetation is considered unsuitable for phytoremediation as these types of land cover areas are either already vegetated by trees & shrubs or sealed by buildings and roads. This also applies to other land cover types unsuitable for cropping, such as beaches and dunes, bare rocks and water bodies. Land cover types in potentially contaminated sites with discontinuous urban fabric (e.g. mineral extraction sites) and with some form of agricultural land use are considered suitable for phytoremediation, provided that less than 40% of the area is artificially sealed (impervious).
- The total area of potentially contaminated sites with land cover types suitable for phytoremediation, and with less than 40% of the area sealed (impervious), amounts to 2,013,722 ha in the EU27 and UK. This area corresponds to 0.5% of the total surface area of these countries.

- France, Germany, Spain and UK have the largest total areas of all types of potentially contaminated sites, amounting to more than 150,000 ha in each of the countries.
- Land currently in use for agriculture covers between 7% (in military sites) and 20% (in landfills) of the area in potentially contaminated sites identified in OSM. These areas offer opportunities for phytoremediation through biomass cropping, because less effort is required for conversion of the land use than if the area would be covered by constructions or natural areas.
- In the areas where quarries were identified in Open Street Map, the land cover type on CLC2018 with the largest extent is 'mineral extraction sites' (CLC code 131), covering 318,548 ha in the OSM-polygons with quarries in the EU and UK, with Germany having the largest coverage (101,227 ha). The total area of the class 'mineral extraction sites' in CLC2018 is 633,848 ha in the EU27 plus UK. This implies that 50% of the CLC2018 class 'mineral extraction sites' is not identified as 'quarry' in Open Street Map. This points to the necessity to consult multiple spatial datasets for the purpose of mapping potentially polluted areas by quarries.
- The Minerals4EU database features 42,731 mines in 22 EU Member States in 8 commodity groups considered of interest for phytoremediation. Of these, only 738 were found in proximity of potentially contaminated sites identified in Open Street Map. A large number of mines in the Minerals4EU database (20,137) was not identified in OSM, and of this number, only 204 are indicated as mines in the land cover class 'mineral extraction sites' in the Corine Land Cover database (class nr 7). These findings show that the databases with European coverage OSM and CLC2018 represent only a small part of the potentially contaminated sites, and that dedicated databases with spatial information on geographical objects associated with local contamination are required to map contaminated sites.
- Commodities produced in mines, as specified per mine in the Minerals4EU database, were ranked according to the risk for human health and the possibility to reduce the risk in the site with biomass crops, and the likeliness of three modes of phytoremediation to manage the commodity. In 57% of the mines, commodities pose a high risk to human health and there is a need to remediate the contamination. For the commodities in this group phytoremediation might be possible to reduce the risk. In 40% of the mines, commodities do not pose a high risk for human health and the need to apply remediation is low.
- Most mines with commodities relevant for phytoremediation have land cover 'non-irrigated arable land', 'pastures' and 'transitional woodland-shrub' (>2,000 in the EU for each land cover class). Of the total of 20,708 mines observed in land cover classes considered relevant for phytoremediation, almost half (10,206) are located in areas with agricultural land use. These findings suggest a potential for options to use existing agricultural land in (former) mine areas for biomass crop production.
- In the group of mines with high risk for human health with commodities suitable for phytoremediation, the largest numbers of mines are estimated likely to treat with phytoremediation through extraction (28% if the total number of mines) or stabilization of the commodities (37% of the total number). There is less potential for or degradation/ volatilization of the compound by planting vegetation.
- Landfills identified in Open Street Map are covered for 37% by dump sites on the CLC2018 map and for 15% by mineral extraction sites (15,485 ha). The latter might refer to areas where residues from

mining operations are piled up next to the mine, and are covered by some form of vegetation. 21% of the areas indicated as landfill in OSM is covered with some form of agricultural land, mainly by non-irrigated arable land and pastures, which may be relevant for phytoremediation using bioenergy crops, in case soil pollution is present. This requires an assessment at the level of these sites.

- The total area of landfills in EU27 and UK on Open Street Map is 99,992 ha, overlapping with 88% of the total area of dump sites on CLC2018 (113,763 ha). This might suggest that not all landfills are identified in Open Street Map. However, there are also countries where the total area of polygons tagged as 'landfill' in Open Street Map is larger than the total area covered by dump sites on the CLC2018 map. Again it confirms to the need to consult multiple spatial datasets for the purpose of mapping potentially polluted areas in or around landfills.
- Brownfields may be considered a sub-set of industrial areas. In Open Street Map, 66,048 ha was tagged as both types of land use in the EU27 and UK, corresponding to 94% of the total area of brownfields. For the generation of a map of potentially contaminated sites, the polygons tagged as industrial areas and brownfields on Open Street Map were therefore merged. This results in a total of 2,725,502 ha of industrial sites and brownfields, occurring in the EU27 plus the UK. Of this area 167,877 ha is in use by some form of agriculture (according to the overlay with CLC2018), which may be relevant for phytoremediation using bioenergy crops, in case soil pollution is present.
- In the category of industrial sites, steel production sites with blast furnaces may deliver pollution risks through the emission of fine particles, but pollution of soils has not been demonstrated. It is however conceivable that vegetation might be used to stabilize particulate matter in the vicinity of the steel production sites and to prevent transport to other areas.
- 27 steel production sites with blast furnaces were mapped in the EU, with land cover in an area of 5 km around these sites. Considering only land cover types suitable for phytoremediation with <40% imperviousness, 60% of the area currently has land cover reflecting agricultural use. This might offer potential to deploy the area for stabilization of fine particulate matter by biomass crops.

6.4 Further review of information

Further review of information will be done by the project team in 2023 on the following aspects:

- The collection and analysis of national and regional data on potentially contaminated sites will be considered for several EU countries in 2023. The aim is to refine the mapping of contaminated sites based on OSM and CLC2018.
- Continuation of the modelling of diffuse pollution with updated (better) input data where possible on soils and contamination data in soils and pedo-transfer models.
- Based on the mapped information presented in this report the sites will be selected for the spatially explicit modelling of the selected value chains in task 3.2.2.

References

Amlinger F, Pollak M, Favoino E. Heavy metals and organic compounds from wastes used as organic fertilisers. ENV.A.2./ETU/2001/0024. Final Report to DG Environment, Brussels; 2004.

Antoniadis, V., Shaheen, S. M., Levizou, E., Shahid, M., Niazi, N. K., Vithanage, M., Ok, Y. S., Bolan, N. & Rinklebe, J. 2019. A critical prospective analysis of the potential toxicity of trace element regulation limits in soils worldwide: Are they protective concerning health risk assessment? A review. Environment International, 127, 819-847.

Ballabio, C., Jones, A., Montanarella, L. and Toth, G., Cadmium in the soils of the EU, EUR 28525 EN, Publications Office of the European Union, Luxembourg, 2023, ISBN 978-92-79-66930-96

Ballabio, C., Panagos, P., Lugato, E., Huang, J., Orgiazzi, A., Jones, A., Fernandez Ugalde, O., Borrelli, P. and Montanarella, L., Copper distribution in European topsoils: An assessment based on LUCAS soil survey, SCIENCE OF THE TOTAL ENVIRONMENT, 2018, p. 282-298.

Baritz et al 2021 Soil monitoring in Europe. Indicators and thresholds for soil quality assessments. EEA ETC/ULS Report | 2021

Bouwman, L.A., and J. Vangronsveld. 2004**.** Rehabilitation of the Nematode Fauna in a Phytostabilized, Heavily Zinc-contaminated, Sandy Soil. J Soils & Sediments 4 (1) 17 – 23 (2004)

Broomandi, P., M. Guney, J.R. Kim and F. Karaca. 2020. Soil Contamination in Areas Impacted by Military Activities: A Critical Review. Sustainability 2020, 12, 9002; doi:10.3390/su12219002

Carlon, C. 2007. Derivation Methods of Soil Screening Values in Europe: A Review of National Procedures Towards Harmonisation: a Report of the ENSURE Action, EUR-OP.

COUNCIL DIRECTIVE 98/83/EC of 3 November 1998 on the quality of water intended for human consumption.

EC. 2000. Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy

EEA. 2022. European groundwater – a key resource under pressure. Briefing no. 03/2022. ISBN: 978-92-9480-460-0 - ISSN: 2467-3196 - Doi: 10.2800/629513 COUNCIL DIRECTIVE 98/83/EC. COUNCIL DIRECTIVE 98/83/EC of 3 November 1998 on the quality of water intended for human consumption.

EU 2021/1317. Commission Regulation (EU) 2021/1317 of 9 August 2021 amending Regulation (EC) No 1881/2006 as regards maximum levels of lead in certain foodstuffs (Text with EEA relevance)

EU 2021/1323. Commission Regulation (EU) 2021/1323 of 10 August 2021 amending Regulation (EC) No 1881/2006 as regards maximum levels of cadmium in certain foodstuffs (Text with EEA relevance)

EUROFER. 2020. European Steel Association AISBL (EUROFER).

FAO and UNEP. 2021. Global assessment of soil pollution: Report. Rome. https://doi.org/10.4060/cb4894en

Hirschmugl, M. and Sobe, C. 2020. Bioplat EU. Promoting sustainable use of underutilised lands for bioenergy production through a web-based platform for Europe.

Koopmans, G.F., P.F.A.M. Römkens, M.J. Fokkema, J. Song, Y.M. Luo, J. Japenga, and F.J. Zhao. 2008. Feasibility of phytoextraction to remediate cadmium and zinc contaminated soils. Environmental Pollution 156 (2008) 905–914.

Laumanns, U., Panaretou, V., Pisimisi, M., Panagiotakis, I. (2021). Improved hazardous waste & contaminated soil management in Greece. Deutsche Gesellschaft für Internationale Zusammenarbeit GmbH (GIZ) provides under contract 81262364 in EC Structural Reform Support Programme (SRSP): *"Technical support for the implementation of the National Waste Management Plan (NWMP) of Greece"*

Liang, H.M., Lin, T.H., Chiou, J.M., and Yeh, K.C. 2009. Model evaluation of the phytoextraction potential of heavy metal hyperaccumulators and non-hyperaccumulators. Environmental Pollution 157 (2009) pp 1945–1952.

Liederkerke, M van; Masson, J.; Perez, A.P.; Jones, A.; Geissen, V.; Baritz, R.; Havlicek, E; Silva, V.; Yang, X.. Regional status of soil pollution: Europe. Global Symposium on soil pollution Rome, Italy 2-4 May 2018 https://www.researchgate.net/publication/327981472_Regional_status_of_soil_pollution_Europe

Lofts, S., D.J. Spurgeon, C. Svendsen, E. Tipping. 2004. Deriving Soil Critical Limits for Cu, Zn, Cd, and Pb: A Method Based on Free Ion Concentrations. Environ. Sci. Technol. 2004, 38, 3623-3631

M. Mesman, M. Rutgers, A.J. Schouten, J.J. Bogte, E.M. Dirven-van Breemen. 2014. Evaluatie van Triade-onderzoeken op schietterreinen van Defensie. RIVM rapport 2014-0077.

Mononen, T., S. Kivinen, J.M. Kotilainen and J. Leino. 2022. Social and environmental impacts of mining activities in the EU. STUDY Policy Department for Citizens' Rights and Constitutional Affairs. Directorate-General for Internal Policies PE 729.156 - May 2022

Ok et al., 2020

Orgiazzi, A., Ballabio, C., Panagos, P., Jones, A., Fernández-Ugalde, O. 2018. LUCAS Soil, the largest expandable soil dataset for Europe: A review. European Journal of Soil Science, 69(1): 140-153. https://doi.org/10.1111/ejss.12499

Ossai et al., 2020

Panagos, P., Van Liedekerke, M., Borrelli, P., Köninger, J., Ballabio, C., Orgiazzi, A., Lugato, E., Liakos, L., Hervas, J., Jones, A., & Montanarella, L. (2022). European Soil Data Centre 2.0: Soil data and knowledge in support of the EU policies. *European Journal of Soil Science*, 73(6), e13315. https://doi.org/10.1111/ejss.13315

Payá Pérez, A. and Rodríguez Eugenio N. (2018), Status of local soil contamination in Europe: Revision of the indicator "Progress in the management Contaminated Sites in Europe, EUR 29124 EN, Publications Office of the European Union, Luxembourg, 2018, ISBN 978-92-79-80072-6, doi:10.2760/093804, JRC107508

Peco, J.D.; Higueras, P.; Campos, J.A.; Esbrí, J.M.; Moreno, M.M.; Battaglia-Brunet, F.; Sandalio, L.M. Abandoned Mine Lands Reclamation by Plant Remediation Technologies. Sustainability 2021, 13, 6555. https://doi.org/10.3390/ su13126555

Pérez, A.P. and Eugenio, N.R, 2018. Status of local soil contamination in Europe: Revision of the indicator "Progress in the management Contaminated Sites in Europe, EUR 29124 EN, Publications Office of the European Union, Luxembourg, 2018, ISBN 978-92-79-80072-6, doi:10.2760/093804, JRC107508

Pinedo, J., Ibáñez, R., Lijzen, J. & Irabien, A. 2013. Assessment of soil pollution based on total petroleum hydrocarbons and individual oil substances. Journal of environmental management, 130, 72-79.

Reimann, C. and R.G. Garrett. 2005. Geochemical Background-Concept and Reality. Science of the Total Environment, 350, 12-27.

Reimann, C., Fabian, K., Birke, M., Filzmoser, P., Demetriades, A., Négrel, P., Oorts, K., Matschullat, J., de Caritat, P. & Albanese, S. 2018. GEMAS: Establishing geochemical background and threshold for 53 chemical elements in European agricultural soil. Applied geochemistry, 88, 302-318.

Rodríguez Lado, L.; Hengl, T.; Reuter, H. (2008). Heavy metals in European soils: A geostatistical analysis of the FOREGS Geochemical database. Geoderma 148 (2008) 189–199

Rodrigues, S. M., Pereira, M. E., da Silva, E. F., Hursthouse, A. & Duarte, A. 2009. A review of regulatory decisions for environmental protection: Part I—Challenges in the implementation of national soil policies. Environment International, 35, 202-213.

Römkens, P.F.A.M., en J.H. Faber, 2020. Bodemverontreiniging in het militair oefenterrein 'De Dellen' Beschikbaarheid van lood, koper en antimoon in de bodem in relatie tot ecologische en verspreidingsrisico's. Wageningen, Wageningen Environmental Research, Rapport 3105

Swartjes, F., Cornelis, C., Wcislo, E., Muller, D., Hazebrouck, B., Jones, C. & Nathanail, C. 2009. Towards consistency in Risk assessment tools for contaminated sites management in the EU. RIVM letter report 711701091.

Tóth, G., T. Hermann, M.R. Da Silva, and L. Montanarella. 2016. Heavy metals in agricultural soils of the European Union with implications for food safety, Environment International, Volume 88, 2016, Pages 299-309.

Trombetti, M, H. Kros, P. Römkens, and W. de Vries. 2022. Soil health and soil degradation maps. Report on soil quality mapping. EEA report. in preparation.

Van-Camp. L., Bujarrabal, B., Gentile, A-R., Jones, R.J.A., Montanarella, L., Olazabal, C. and Selvaradjou, S-K. (2004). Reports of the Technical Working Groups Established under the Thematic Strategy for Soil Protection. EUR 21319 EN/4, 872 pp. Office for Official Publications of the European Communities, Luxembourg. Volume V – Monitoring.

Verweij, P., Janssen, S., Braat, L., Eupen, van M., Perez Soba, M., Winograd, M. Winter, de W., Cormont, A. (2016). QUICKScan as a quick and participatory methodology for problem identification and scoping in policy processes. Environmental Science & Policy Volume 66, December 2016, Pages 47-61 10.1016/j.envsci.2016.07.010

Vries, de W., P.F. Römkens, and G. Schütze. 2007. Critical soil concentrations of cadmium, lead, and mercury in view of health effects on humans and animals. Reviews of Environmental Contamination and Toxicology, 191, pp. 91 – 130.

WAR (2020). Afvalverwerking in Nederland, gegevens 2018. Werkgroep Afvalregistratie, Utrecht.

Wille, E. and Isenborghs, C. (2022). Data collection on soil contamination in the region of Flanders (Belgium) Data management of landfill sites Applicable laws.

Annex 1 Overview of data on contaminated sites at national level

Netherlands

Georeferenced data on (potentially) contaminated sites in The Netherlands are collected and documented at regional level by the larger municipalities (a.o. Rotterdam, Amsterdam) and the regional environmental agencies, on behalf of municipalities. At the moment of writing, there is no central registration at national level that is publicly accessible, although there is an online office for information on soil quality that refers to the websites and geoportals of municipalities and regions (www.bodemloket.nl). Data on substances in soil en subsoil that are relevant for the quality of life (no only pollutants) will be included in the Dutch National Key Registry of the Subsurface[13] as off 2024.

A national register of all contaminated sites in The Netherlands was built in 2021 by a consultancy firm for the construction sector[14], but this is not publicly accessible. However, the identification of contaminated sites for phytoremediation focusses on areas that are not considered for construction and are intended to remain vegetated. Therefore, we will not consider this register for cross-checking the identification of contaminated sites in Open Street Map.

The municipalities and regional environmental agencies manage information on contaminated and remediated sites in soil information systems. We will illustrate the way in which the information is stored and hosted in an example for a region in the province of Noord-Brabant by the environmental agency *Omgevingsdienst Zuidoost-Brabant*[15]. This region was selected to cross-check the identification of contaminated sites in Open Street Map, because the occurrence of areas polluted with heavy metals is known in the area, for example by ashes of zinc coming from a factory in the municipality of Budel (Figure 43, Figure 44).

[13] https://basisregistratieondergrond.nl/english
[14] https://anteagroup.nl/diensten/milieu-en-omgevingsdata/bodem-digitaal-op-de-kaart
[15] https://odzob.nl/

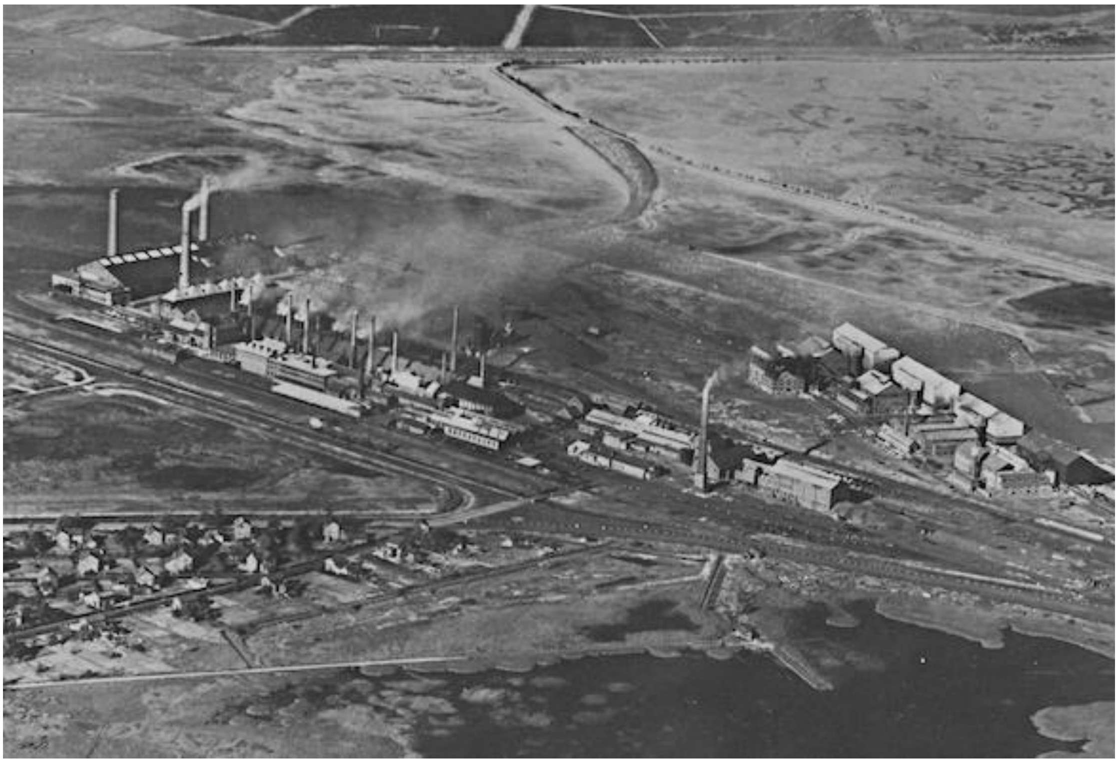


Figure 43 Zinc factory of the N.V. Kempensche Zinkmaatschappij in the municipality of Budel, The Netherlands in 1973. Source: http://www.historiekzm.nl/.

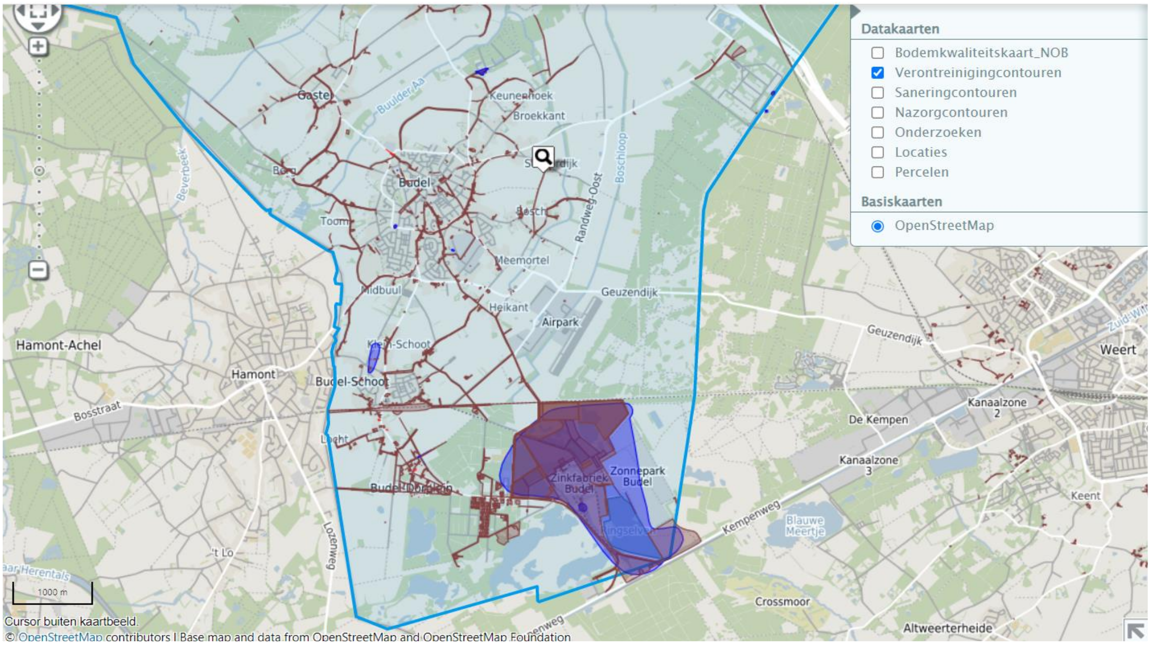


Figure 44 Fragment of a map with contours of contaminated areas in the soil information system of the environmental agency in the southeast of the province of Noord-Brabant in The Netherlands. Source: https://noord-brabant.omgevingsrapportage.nl/#.

We requested geoinformation on the contours of contaminated and remediated sites in the region from the soil information system of the environmental agency, including information on the cause of the pollution and the type and amounts of contaminants found in each site. We received detailed information from the soil information system in xml-format, including information on:

- Polygons delineating areas where soil surveys were carried out based on suspicion or mention of contamination, including locations of soil borings;
- Polygons delineating areas that were identified as being contaminated and areas where contamination was remediated.

An impression of how the information is organized when imported in GIS is given in Figure 45.
The information from the soil information system is coded according to the SIKB 0101 protocol[16] for the exchange of soil information between authorities and parties with an obligation to provide soil information. The attribute information on the polygons contains information on exceedance of thresholds for contamination, but information on the type(s) of contaminating substances is not clearly indicated and was found only for some of the polygons in the attribute field with remarks.

The information in the soil surveys to each polygon in the geodataset can be requested at the environmental agency. For each surveyed lot, this consists of a series of reports in pdf format, made by engineering offices with detailed information on the current and historic use of the lot, the soil properties and contaminated substances observed.

In summary, detailed information on contaminated sites is stored in the regional soil information system at the level of plots of land in the land register, grouped per municipality. For the Netherlands as a whole, a geoinformation system with contours of contaminated sites and types and contents of contaminating substances, is not available.

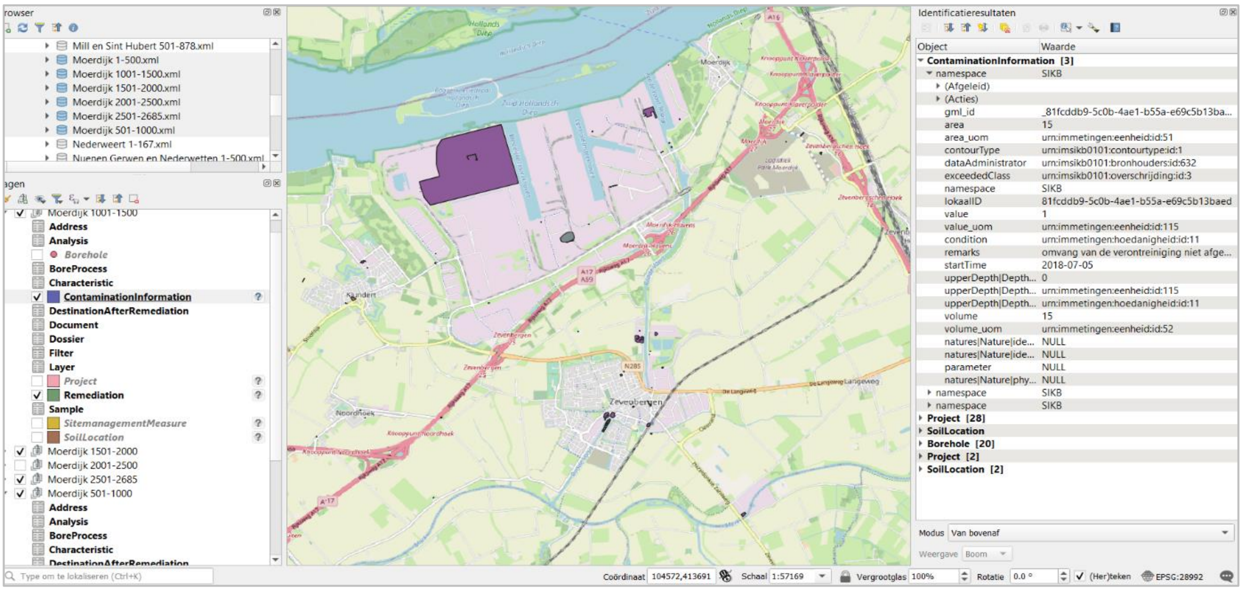


Figure 45 Example of contents of the soil information system of the environmental agency Omgevingsdienst Zuidoost-Brabant for the municipality of Moerdijk in The Netherlands. An industrial site of the Shell company is shown in light purple. Polygons in dark purple indicate areas with information on contamination. Source: Omgevingsdienst Zuidoost-Brabant, information received on 31-3-2022.

16 https://www.sikb.nl/datastandaarden/sikb0101-bodembeheer

Flanders (Belgium)

In Belgium, the Public Waste Agency of Flanders (OVAM) is responsible for the management of waste, materials and soil remediation in the region of Flanders. A public geoportal (Geopunt) provides georeferenced information on investigated sites that were reported at OVAM (Figure 46). The sites include locations where soil pollution was reported and where claims of damages were made. Locations of soil surveys and soil decontamination are also included in the geoportal. Each site is characterized by an identification number referring to the underlying reports and uploaded data. The color of the polygons indicates the type of report (preliminary investigation, detailed investigation, remediation project, remedial actions, monitoring) (Wille and Isenborgs, 2022).

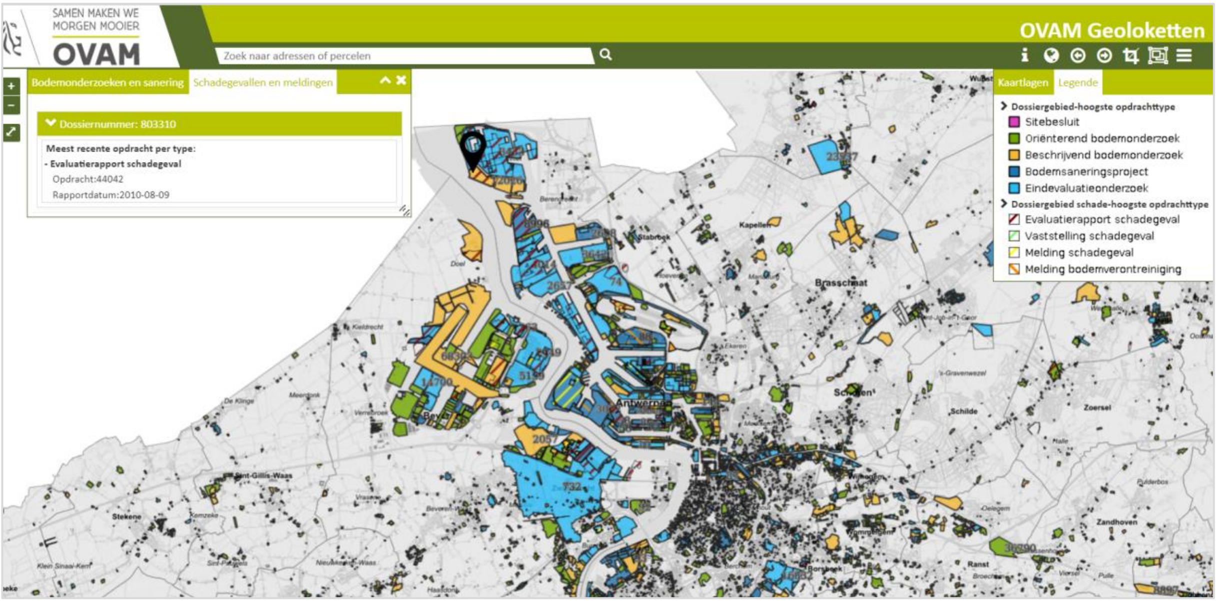


Figure 46 Presentation of (potentially) contaminated sites in Flanders in the geoportal Geopunt. Coloured polygons show areas for which files are available. Reports can be requested from the Flemish government through the file numbers. Source: https://www.vlaanderen.be/geopunt/kaarttoepassingen/ovam-geoloket-bodemdossierinformatie.

For landfills, OVAM maintains a map with the contours of administrative parcels where activities of waste dumping are known to occur. The map of landfills indicate if the waste dumping is still active, the presence of hazardous waste and the type of waste. The map of landfills can be consulted through the geoportal Geopunt.

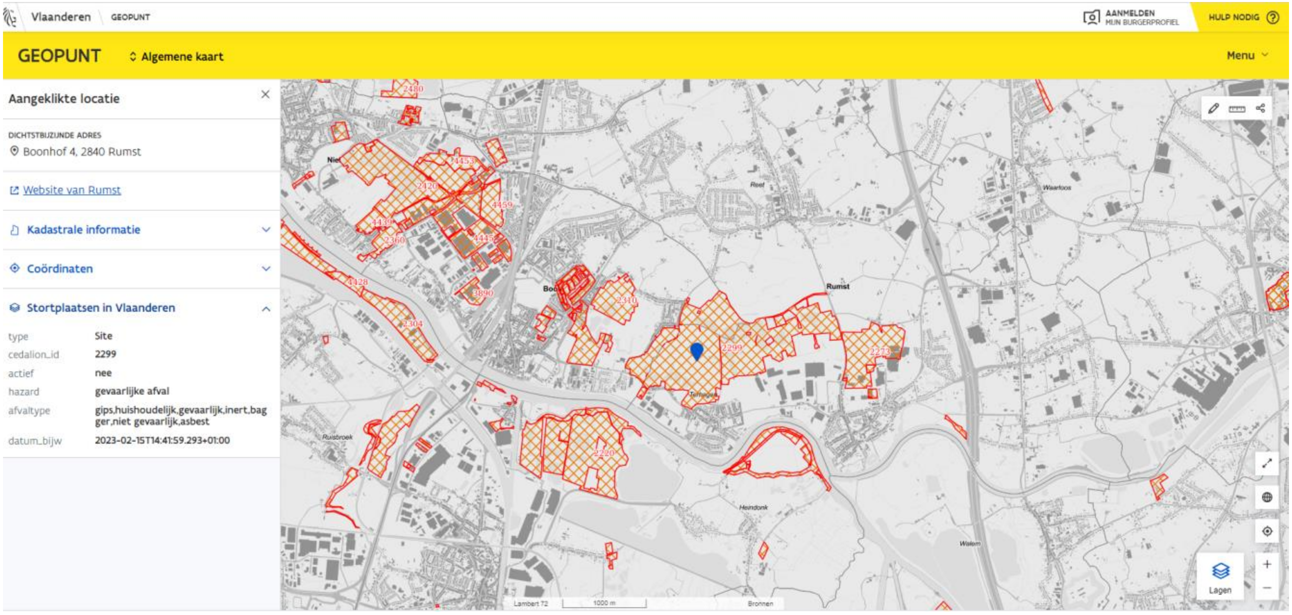


Figure 47 Map of landfills for the region of Flanders in the geoportal Geopunt. Source: OVAM, version 23-01-2023. https://www.vlaanderen.be/datavindplaats/catalogus/stortplaatsen-in-vlaanderen

France

The national database of contaminated or potentially contaminated sites for France is *Base des sols pollués (BASOL)*. There is also the national Inventory of Abandoned Industrial Sites (*Inventaire des Anciens Sites Industriels et Activités de Service (BASIAS)*). Spatial data on the sites in BASIAS are included in the map of abandoned industrial sites held by the national geological service BRGM, entitled CASIAS (*Carte des Anciens Sites Industriels et Activités de Services*).

There is also a public website of brownfields in France, where citizens can provide and consult local information on brownfields, entitled *Cartofriches*. The site contains the data from the databases BASIAS and BASOL.

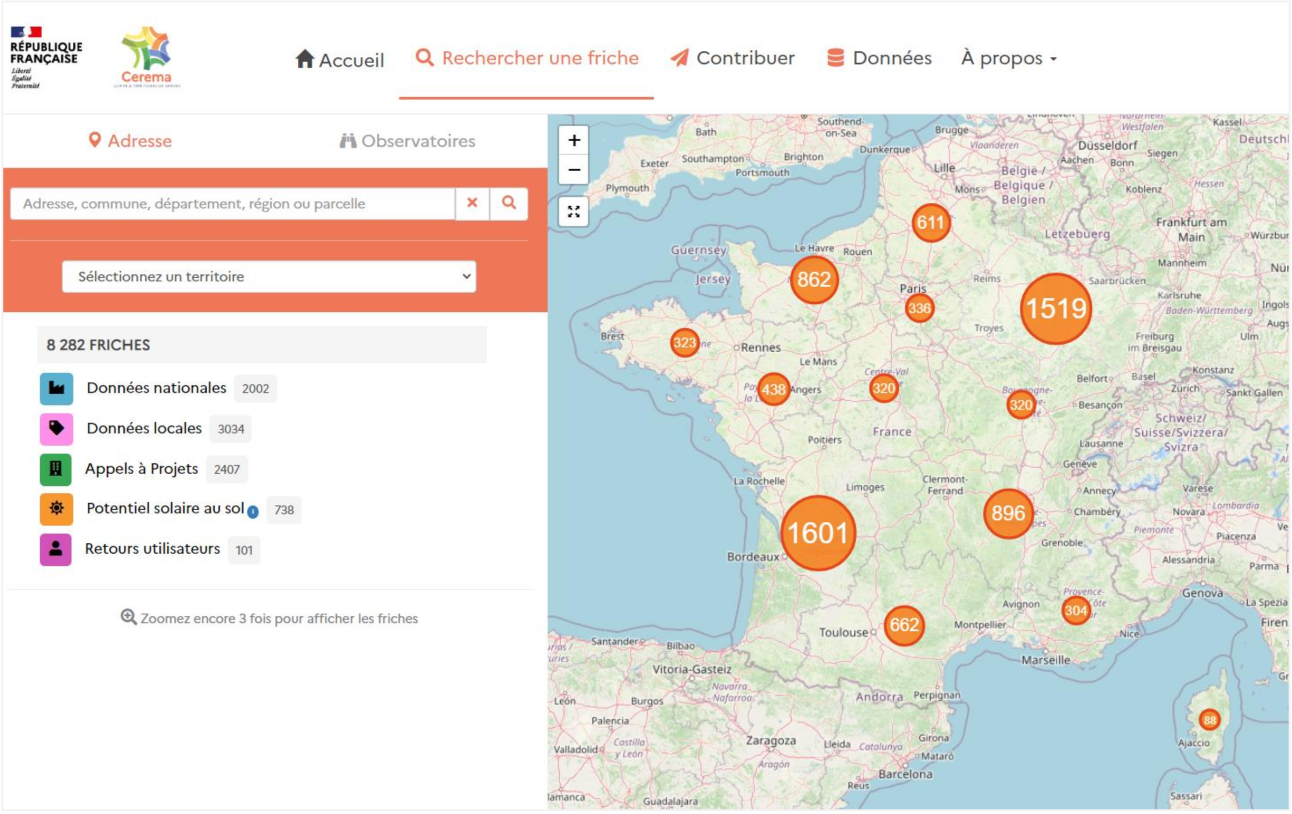


Figure 48 Presentation of Cartofriches, the public webportal of brownfields in France. Source: https://cartofriches.cerema.fr/.

Greece

The existing contaminated soil management in Greece is fragmentary and complicated and the authorities involved are understaffed without the appropriate expertise, in most of times. The Greek inventory for contaminated sites started a couple of years ago but was never implemented (Laumanns et al., 2021). In 2009 a study was completed for the investigation, evaluation and remediation of uncontrolled (illegal) contaminated sites with industrial and hazardous wastes. In 2013 another study was initiated for recording and evaluation of the contaminated sites by industrial hazardous wastes in the region of Attica and the prefecture of Thessaloniki, Viotia, Evia, Kozani, Achaia, Heraklion, Magnisia, Kavala and Chalkidiki (the areas that account for most of the country's industrial activity) (Tsompanidis et al 2016). The goal of this study was the detection, recording and the initial characterisation of potentially contaminated sites focusing on areas with heavy industrial activity, storage areas of industrial and hazardous waste, waste-management areas, mining activities, shipyards etc. All sites were classified into two categories: as controlled (legal) or uncontrolled (illegal) sites. 2.029 potentially contaminated sites were identified and prioritized. The 300 most important contaminated sites were selected for further investigation through questionnaires and on-site assessment. This project was the first approach and indicates that more research is needed, including ecotoxicological studies, a setting out of 36 polluting parameters and thresholds, clarification of reference sampling and robust site sampling and monitoring (Tsompanidis et al 2016).

Laumanns et al. (2021) concludes that with regard to sites contaminated by illegal landfills, Greece has an analytical database. According to official data reported to the European Commission in the context of the relevant decision of the European Court of Justice imposing fines on Greece for the case of illegal landfills, there were 293 illegal landfills in December 2014. By December 2017 the number had dropped to 44. The rest (149) have been rehabilitated. It should be noted that the number of illegal landfills exceeded 3.000 landfills in 2010 but, in the meantime, most of them have been rehabilitated (Pérez and Eugenio 2018). Therefore, based on the above it is obvious that the contaminated site in Greece is still pending.

In the meantime many more scientific approaches have been implemented to mapping contaminated sites in Greece. In the following an overview is given of these as summarized by Eleni Papazoglou in 2023.

Literature on soil contamination

1. Level of Contamination Assessment of Potentially Toxic Elements in the Urban Soils of Volos City (Central Greece) (Golia et al., 2021).

Research area: Volos City (Central Greece)

Surface: 3.65 km^2

Source of pollution: industrial area that also includes a steel plant and large cement industry, producing seven types of cement, clinker, solid fuels, and aggregates, is located a distance of 4 km to the east part of the city

Type of pollutants present: Co, Mn, Cd, Zn, Cu, Pb, Cr, Ni

Level of contamination: low to moderate

Results: The mean values of metal concentrations were found to be lower than maximum permitted values, except Co and Mn mean concentrations ha were higher than maximum permitted values. The largest number of soil samples had CF values belonging to class II (1–3) and were characterized as “moderate contamination”. In class II, the order of CFs was as follows: Cd = Zn > Ni > Co > Cu = Pb = Cr = Mn. In class III (CF: 3–6), characterized as “considerable contamination”, the order changed as follows: Ni > Co > Zn = Cd. There were no soil samples with CF > 6, i.e., belonging to class IV with “very high contamination”.

2. A Study of Chromium, Cooper, and Lead Distribution from Lignite Fuels Using Cultivated and Non-cultivated Plants as Biological Monitors (Sawidis et al., 2011).

Research area: Ptolemais, Agios Dimitrios region

Source of pollution: four coal power plants

Type of pollutants present: Cr, Cu, Pb

Level of contamination: low

Results: The mean heavy metal content in the soil is described in the descending order of Cr>Pb>Cu. Stations in the vicinity of the CPP showed a distinctly high load of chromium in the soil, whereas for the other metals, no such correlation has been noted. In the case of lead, higher concentrations were found in the most remote stations. Results showed that there is no serious heavy metal pollution in the area of the coal power plant.

3. Spatial diversity of Cr distribution in soil and groundwater sites in relation with land use management in a Mediterranean region: The case of C. Evia and Assopos-Thiva Basins, Greece (Megremi et al., 2019).

Research area: Assopos-Thiva Basins and C. Evia

Source of pollution: widespread occurrence of ophiolites and Fe-Ni-laterite deposits

Type of pollutants present: Cr, Ni, Co

Level of contamination: high

Results: Average metal values in soils range from 50 to 190 mg/kg Cr in the Oropos area and from 130 to 520 mg/kg at the Avlona area in the Assopos-Thiva Basin, 230 to 310 mg/kg Cr (occasionally 800 mg/kg) in the north part of the basin (Thiva area), with an increasing trend from south to north.

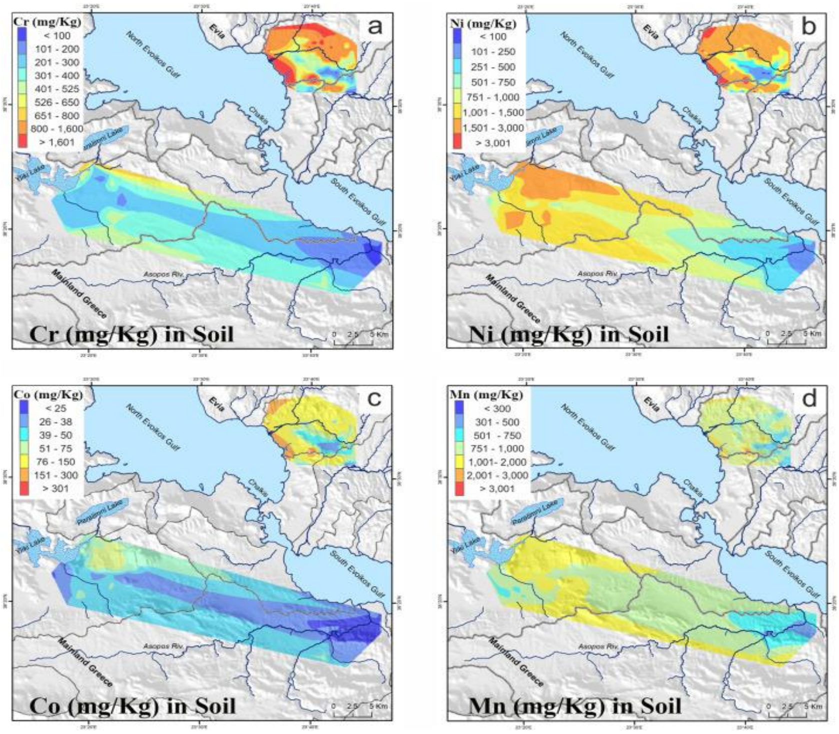


Fig. 1. Maps of Cr, Ni, Co and Mn (a–d) spatial distribution in soils of the Evia and Assopos-Thiva basins.

4. Species adaptation in serpentine soils in Lesbos Island (Greece): metal hyperaccumulation and tolerance (Kazakou et al., 2010).

Research area: Lesbos Island. Four serpentine sites were selected in the following localities: Loutra, Ampeliko, Olympos and Vatera

Type of pollutants present: Ni, Co, Cr and Zn

Results: Data showed that there is a gradient of increasing heavy metal concentration from the Vatera to Ampeliko localities. Ampeliko has the highest heavy metal concentrations (Ni, Co, Cr and Zn), whereas Vatera has the lowest Ni concentration and the lowest Mg/Ca quotient. Olympos has the highest Mg values. Nickel concentrations were always >1,000 mg kg^{-1} across all locations.

5. Investigation of heavy metal pollution of the soils of the areas of Inofyta – Oinoi – Schimatari (Tsoumani, 2021).

Research areas: Avlona, Schimatari, Oinoi and Inofyta

Source of pollution: industrial activity

Type of pollutants present: Cu, Ni, Co, Cr, Zn, Pb

Level of contamination: high

Results: Total heavy metal concentrations showed that soils are polluted in terms of lead, nickel, chromium and less in terms of manganese, while they do not show much pollution from copper and zink. The highest concentrations of bioavailable nickel and manganese were determined in Oinofyta and Avlona, while bioavailability of lead in Oinofyta and Schimatari. The highest values of the available forms of copper and zink are found in the wider area of Inofyta. According to the soil pollution indices calculated, in general soils are moderately polluted by heavy metals, but the indicators indicate stronger soil pollution by the elements lead and nickel.

6. Soil and maize contamination by trace elements and associated health risk assessment in the industrial area of Volos, Greece (Antoniadis et al., 2019).

Research area: Volos

Source of pollution: steel factory

Type of pollution: metals and metalloids

Type of pollutants present: Ag, As, Co, Cr, Mn, Mo, Ni, Sb, Se, Sn, Tl, Ni, Cr

Land use: highly intensive agricultural activities

Results: Ag was 0.49 with background of 0.13, As was 100.33 vs. background of 6.83, Co was 34.85 vs. 11.3, Cr 438.29 vs. 59.5, Mn 762.02 vs. 480, Mo 10.68 vs. 1.1, Ni 327.46 vs. 29, Sb 18.47 vs. 0.67, Se 14.40 vs. 0.44, Sn 10.49 vs. 2.5, and Tl 17.51 vs. 0.5. It should be noted that some of the studied elements showed low levels: Cd, Pb, and V were well below their background levels, while average Cu (39.78) and Zn (69.23 mg kg^{-1}) were similar to their respective background levels. Also, from the elements covered by the EU Directive (CEC, 1986), Cd (3), Cu (140), Pb (300), and Zn (300) were found well below that threshold (legal limit thresholds in parentheses), while average Ni was 4.4-fold higher than its limit of 75 mg kg^{-1}. soils were extremely enriched with Tl, a highly toxic metal. Along with it, other less expected elements were also found severely enriched, i.e., Se, Sb, and Mo; soils also contained high concentrations of As, Ni and Cr.

7. Magnetic signature, geochemistry, and oral bioaccessibility of “technogenic” metals in contaminated industrial soils from Sindos Industrial Area, Northern Greece (Bourliva et al., 2016).

Research area: Sindos Industrial Area

Source of pollution: major part of the industrial activity of the Thessaloniki plain

Type of pollution: metals and metalloids

Type of pollutants present: As, Cd, Co, Cr, Cu, Ni, Pb, Mo, Zn

Results: The 28% of the studied soil samples were characterized as ''moderately to heavily polluted'' with reference to Cd and Zn, while especially for Zn 14% of the samples are classified as ''heavily to extremely heavily polluted''. One specific soil sample collected in the area north-west of the industrial unit was characterized as ''extremely polluted'' for Cd and Pb and ''heavily to extremely polluted'' for Zn.

8. Heavy Metals in Agricultural Soils of the Mouriki-Thiva Area (Central Greece) and Environmental Impact Implications (Antibachi et al., 2012).

Research area: Mouriki-Thiva Area, situated in the prefecture of Viotia

Surface: 150 km^2

Source of pollution: major part of the industrial activity of the Thessaloniki plain

Type of pollution: metals and metalloids

Type of pollutants present: Co, Cr, Cu, Ni, Pb, Zn

Land use: Potatoes, carrots, cotton, grain, and beans are mainly cultivated in the plain area

Results: The soils of the Mouriki-Thiva area showed elevated concentrations of Ni, Cr, Co, Fe and Mn. The studied soils are significantly contaminated by Ni, presenting concentrations that are extremely higher than the Dutch proposed guideline value.

9. Public health risk assessment associated with heavy metal and arsenic exposure near an abandoned mine (Kirki, Greece) (Nikolaidis et al., 2013).

Research area: 'Agios Philippos' in the Kirki region (NE Greece)

Surface: 20 km^2

Source of pollution: abandoned lead–zinc mine

Type of pollution: metals and metalloids

Type of pollutants present: As, Cd, Pb, Zn

10. Soil contamination by toxic metals in the cultivated region of Agia, Thessaly, Greece. Identification of sources of contamination (Skordas et al., 2005).

Research area: Agia area in the eastern part of the Larissa town, central Greece

Coordinates: latitudes 39°41'00''–39°45'00'' and longitudes 22°41'00''–22°47'00''

Type of pollution: metals and metalloids

Type of pollutants present: Cu, Pb, Zn, Ni, Co, Mn, As, V, Cr, Fe and Mg

Level of pollution: slightly contaminated to contaminated

Land use: apple trees

Results: According to the concentrations of Pb and Zn the soils of the studied area are characterized as uncontaminated (category I). According to the concentration values of As and Cu the area of Agia is characterized as uncontaminated (category I) 91 and 88%, respectively, or slightly contaminated (category II) 9 and12%, respectively. Almost the whole area is characterized as slightly to contaminated (categories I–IV) by Ni. Only two samples have values >1000 ppm (category V). According to the concentration values of Cr the soils of the Agia area are characterized as slightly contaminated or contaminated. The largest part of the area studied is slightly contaminated (category II, 88%) or contaminated by Mn (category III, 88%), 4% of the soil samples are heavy contaminated. The largest part of the Agia area is characterized as slightly contaminated (category II) by V, 3% as contaminated and 9% as uncontaminated (category I). Agia soils are mainly contaminated by Ni, Cr, Fe, V and Mn. The results of soil analyses from the region of Agia, Central Greece showed elevated concentrations of Ni, Cr, Co, Fe, Mn, Zn, Cu, V and As. All these metals studied are present in soil with concentrations above the mean values of global soils. Some elements like Ni, Cr, Mn and V have concentrations that according to G.L.C.

11. Environmental geochemical research for the levels and the sources of toxic metals in the agricultural soils of dimitraeleftherion and platycampos region, Thessaly, Greece (Skordas et al., 2017).

Research area: Platycampos region, Thessaly, Greece

Type of pollution: metals and metalloids

Type of pollutants present: Cu, Pb, Zn, Ni, Co, Mn, Fe, As, V, Cr and Mg

Level of pollution: slightly contaminated to contaminated

12. Arsenic accumulation in irrigated agricultural soils in Northern Greece (Casentini et al., 2011).

Research area: Prefecture of Chalkidiki close to the municipality of Nea Triglia

Type of pollution: metalloids

Type of pollutants present: As

Land use: agricultural fields

Results: Arsenic content in sampled soils ranged from 20 to 513 mg/kg inside to 5–66 mg/kg outside the geothermal area.

13. Investigating the sources and potential health risks of environmental contaminants in the soils and drinking waters from the rural clusters in Thiva area (Greece) (Kelepertzis, 2014).

Research area: Thiva

Type of pollution: metals and metalloids

Type of pollutants present: Ni, Cr, Co, Mn, Cu, Pb, Zn, Cd

Results: Copper is the only metal with more than half of the collected samples showing pollution indexes above 1 with a maximum of 1.67. Lead and Zn for a small number of soils also demonstrate a relative enrichment with respect to the maximum background value.

14. Concentration of heavy metals and trace elements in soils, waters and vegetables and assessment of health risk in the vicinity of a lignite-fired power plant (Noli and Tsamos, 2016).

Research area: area is part of the Kozani Ptolemaida-Amyntaion basin, northwestern Greece

Surface: 400 km^2
Source of pollution: coal mining

Type of pollution: metals and metalloids

Type of pollutants present: As, Ba, Co, Cr, Sr, Sc, Th, U, Zn

Level of pollution: slightly contaminated

Results: The obtained data in most of the cases did not exceed the normal levels and indicated that the investigated area was only slightly contaminated.

15. TOPSOIL POLLUTION AS ECOLOGICAL FOOTPRINT OF HISTORICAL MINING ACTIVITIES IN GREECE (Kalyvas et al., 2018).

Research area: Lavrion area

Surface: 150 km^2

Type of pollution: metals and metalloids

Source of pollution: mining and metallurgical activities

Type of pollutants present: Zn, Cu, Ni, Cr, Cd, Pb, As

Level of contamination: severe

Results: Zinc, Pb, Cd, and As median total concentration values were 4, 9, 1.4, and 17 times higher than the respective intervention thresholds, indicating severe soil pollution

Table 1. Concentrations of heavy metals in different areas in Lavrio

Location	Source	Pb (mg/kg)	Zn (mg/kg)	Cd (mg/kg)	Cu (mg/kg)	Ni (mg/kg)	As (mg/kg)

Neraki	Theodoratos et al., 2000)	11.560	9380	63,52	220	330	940
Thorikos Beach	Panagopoulos et al., 2009)	2.615,25	2.295,1	9,925	106,69	<2	10.719
Thorikos Beach	(Panagopoulos et al., 2009)	4.286,10	9.843	29,903	295,29	47,346	11.974
Cultural Technological Park	Moutsatsou et al., 2006	64.195	55.900		4.100		7.540
Kabodokano	Environmental Laboratory 2015	9.993	31.600	14	812		1.225

Table 2. Contamination cases and reported trace elements in Greece as published since 2007. Elements in italics are those elevated at concentrations higher than legislation limits.

Area/(Possible contamination sources)	Site description	Elements	References
Urban Athens (City activities)	Old cemetery	As, Cr, Cu, Pb, *Ni*, *Zn*	Massa et al. (2018a)
	Historic shooting range	*Pb*, *Ni*, Zn	Urrutia-Goyes et al. (2018)
	Children's playground	Co, Cr, Cu, *Ni*, Pb, *Zn*	Massas et al. (2010)
Lavrio (Pb/Zn historic mines from c. 5,000 BC to c. 1900 AD)		*As*, *Cd*, *Cr*, Cu, *Ni*, *Pb*, *Zn*	Kalyvas et al. (2018)
		As, *Cd*, Cr, Cu, Mo, *Ni*, *Pb*, Se, *Zn*	Panagopoulos et al. (2009)
Thriasio (Industrial area near Athens)		*As*, *Cd*, Co, *Cr*, *Cu*, *Mo*, *Ni*, *Pb*, Se, V, *Zn*	Antoniadis et al. (2017b)
		Co, Cr, *Cu*, *Ni*, *Pb*, *Zn*	Massas et al. (2013)
Attica (The Athens Prefecture)	Int'l Airport	Cr, Cu, *Ni*, Pb, *Zn*	Massas et al. (2018b)
	NE Attica, agricultural-forest area	*As*, Cd, *Cr*, Cu, *Ni*, Pb, V, *Zn*	Kampouroglou and Economou-Eliopoulos (2017)
	Around the Prefecture	*As*, Co, *Cr*, Cu, *Ni*, Pb, V, Zn	Kampouroglou and Economou-Eliopoulos (2016)
	NE Attica, agricultural-forest area	*As*, Cu, Mo, Pb, Zn	Kampouroglou and Economou-Eliopoulos (2013)
	Rural area near the Int'l Airport	Co, Cr, Cu, Ni, *Pb*, *Zn*	Kaitsantzian et al. (2013)
Thessaloniki (City activities)	Sindos industrial area	*As*, *Cd*, Co, Cr, *Cu*, Ni, *Pb*, Mo, *Zn*	Bourliva et al. (2017a)
	Historic centre road dust	*Cd*, Cr, *Cu*, *Ni*, *Pb*, *Zn*	Bourliva et al. (2018)
	Historic centre road dust	*Cd*, Cr, *Cu*, *Ni*, *Pb*, *Zn*	Bourliva et al. (2017b)
	City soils	As, Cr, Cu, Ni, Pb, Zn	Topalidis et al. (2017)
	City soils	As, Cd, Co, Cr, *Cu*, Mo, Ni, *Pb*, V, *Zn*	Bourliva et al. (2016)
Assopos basin (High natural Cr and Ni plus		*Cr*, Co, *Ni*	Megremi et al. (2019)
		Cr, Ni	Lilli et al. (2015)
		Co, Cr, Cu, *Ni*, Pb, Zn	Antibachi et al. (2012)

Area/(Possible contamination sources)	Site description	Elements	References
under-regulated factories)		Co, *Cr*, Cu, *Ni*, Pb, Zn	Economou-Eliopoulos et al. (2012)
Kozani (Highland lignite-bearing plain with many lignite-fired power plants)		As, Cu, *Ni*, Pb, Zn	Nanos et al. (2015)
		As, *Cd*, Co, *Cr*, Pb, *Zn*	Noli and Tsamos (2016)
		Cd, Cu, Ni, Zn	Papadopoulos et al. (2017)
Drama (Abandoned Mn mines)		*As*, *Cd*, Cu, *Pb*, *Zn*	Sofianska and Michailidis (2016)
		As, *Cd*, Cu, *Pb*, *Zn*	Sofianska and Michailidis (2015)
Kirki (Abandoned Pb/Zn mines)		*As*, *Cd*, *Pb*, *Zn*	Nikolaidis et al. (2013)
Kavala (Around a major motorway)		*As*, *Cd*, *Cr*, *Cu*, *Ni*, *Pb*, *Zn*	Christoforidis et al. (2009)
Thiva (Town activities)		*Co*, *Cr*, *Ni*	Kelepertzis and Stathopoulou (2013)
Argolida (Citrus growing area)		*As*, *Cd*, *Co*, *Cu*, *Ni*, Pb, Zn	Kelepertzis et al. (2014)
Ermoupoli (Aegean island town—touristic flow)		Cr, *Cu*, *Ni*, *Pb*, *Zn*	Massas et al. (2009)
Volos (industrial area)		*Co*, *Mn*, Cd, Zn, Cu, Pb, Cr, *Ni*	Golia et al. (2021)
		Ag, *As*, *Co*, *Cr*, *Mn*, *Mo*, *Ni*, *Sb*, *Se*, *Sn*, *Tl*, *Ni*, *Cr*	Antoniadis et al. (2019)
Ptolemais, Agios Dimitrios region (four coal power plants)		*Cr*, Cu, Pb	Sawidis et al. (2011)
Agia, Thessaly, Greece *(apple trees)*		*Cu*, Pb, Zn, *Ni*, Co, Mn, As, *V*, Cr, *Fe*, Mg	Skordas and Kelepertsis (2005)
Prefecture of Chalkidiki close to the		*As*	Casentini et al. (2011)

Area/(Possible contamination sources)	Site description	Elements	References
municipality of Nea Triglia (agricultural fields)			
Lesbos Island. Four serpentine sites: Loutra, Ampeliko, Olympos and Vatera		*Ni*, Co, Cr, Zn	Kazakou et al. (2010)

References

Anastopoulos, I., Margiotoudis, I., & Massas, I. (2018). The use of olive tree pruning waste compost to sequestrate methylene blue dye from aqueous solution. International journal of phytoremediation, 20(8), 831-838.

Antibachi, D., Kelepertzis, E., & Kelepertsis, A. (2012). Heavy metals in agricultural soils of the Mouriki-Thiva area (central Greece) and environmental impact implications. Soil and Sediment Contamination: An International Journal, 21(4), 434-450.

Antibachi, D., Kelepertzis, E., & Kelepertsis, A. (2012). Heavy metals in agricultural soils of the Mouriki-Thiva area (central Greece) and environmental impact implications. Soil and Sediment Contamination: An International Journal, 21(4), 434-450.

Antoniadis V, Golia EE, Shaheen SM, Rinklebe J (2017b) Bioavailability and health risk assessment of potentially toxic elements in Thriasio Plain, near Athens Greece. J Environ Geochem Health 39:319–330

Antoniadis, V., Golia, E. E., Liu, Y. T., Wang, S. L., Shaheen, S. M., & Rinklebe, J. (2019). Soil and maize contamination by trace elements and associated health risk assessment in the industrial area of Volos, Greece. Environment international, 124, 79-88.

Bourliva, A., Christophoridis, C., Papadopoulou, L., Giouri, K., Papadopoulos, A., Mitsika, E., & Fytianos, K. (2017). Characterization, heavy metal content and health risk assessment of urban road dusts from the historic center of the city of Thessaloniki, Greece. Environmental Geochemistry and Health, 39(3), 611-634.

Bourliva, A., Christophoridis, C., Papadopoulou, L., Giouri, K., Papadopoulos, A., Mitsika, E., & Fytianos, K. (2017). Characterization, heavy metal content and health risk assessment of urban road dusts from the historic center of the city of Thessaloniki, Greece. Environmental Geochemistry and Health, 39(3), 611-634.

Bourliva, A., Kantiranis, N., Papadopoulou, L., Aidona, E., Christophoridis, C., Kollias, P., ... & Fytianos, K. (2018). Seasonal and spatial variations of magnetic susceptibility and potentially toxic elements (PTEs) in road dusts of Thessaloniki city, Greece: A one-year monitoring period. Science of the Total Environment, 639, 417-427.

Bourliva, A., Papadopoulou, L., & Aidona, E. (2016). Study of road dust magnetic phases as the main carrier of potentially harmful trace elements. Science of the Total Environment, 553, 380-391.

Bourliva, A., Papadopoulou, L., Aidona, E., & Giouri, K. (2017). Magnetic signature, geochemistry, and oral bioaccessibility of "technogenic" metals in contaminated industrial soils from Sindos Industrial Area, Northern Greece. Environmental Science and Pollution Research, 24(20), 17041-17055.

Casentini, B., Hug, S. J., & Nikolaidis, N. P. (2011). Arsenic accumulation in irrigated agricultural soils in Northern Greece. Science of the total environment, 409(22), 4802-4810.

Christoforidis, A., & Stamatis, N. (2009). Heavy metal contamination in street dust and roadside soil along the major national road in Kavala's region, Greece. Geoderma, 151(3-4), 257-263.

Economou-Eliopoulos, M., Antivachi, D., Vasilatos, C., & Megremi, I. (2012). Evaluation of the Cr (VI) and other toxic element contamination and their potential sources: The case of the Thiva basin (Greece). Geoscience Frontiers, 3(4), 523-539.

Golia, E.E.; Papadimou, S.G.; Cavalaris, C.; Tsiropoulos, N.G. Level of Contamination Assessment of Potentially Toxic Elements in the Urban Soils of Volos City (Central Greece). Sustainability 2021, 13, 2029. https://doi.org/10.3390/su13042029

Kalyvas, G., Gasparatos, D., & Massas, I. (2018). A critical assessment on arsenic partitioning in mine-affected soils by using two sequential extraction protocols. Archives of Agronomy and Soil Science, 64(11), 1549-1563.

Kalyvas, G., Gasparatos, D., Papassiopi, N., & Massas, I. (2018). Topsoil pollution as ecological footprint of historical mining activities in Greece. Land Degradation & Development, 29(6), 2025-2035.

Kampouroglou, E. E., & Economou-Eliopoulos, M. (2017). Assessment of arsenic and associated metals in the soil-plant-water system in Neogene basins of Attica, Greece. Catena, 150, 206-222.

Kampouroglou, E. E., & Economou-Eliopoulos, M. (2017). Assessment of arsenic and associated metals in the soil-plant-water system in Neogene basins of Attica, Greece. Catena, 150, 206-222.

Kampouroglou, E., & Economou-Eliopoulos, M. (2013). Natural contamination by As and heavy metals in soil, their bio-accumulation and potential sources: the case of a travertine limestone quarry, Greece. Open Geosciences, 5(1), 174-188.

Kazakou, E., Adamidis, G. C., Baker, A. J., Reeves, R. D., Godino, M., & Dimitrakopoulos, P. G. (2010). Species adaptation in serpentine soils in Lesbos Island (Greece): metal hyperaccumulation and tolerance. Plant and Soil, 332(1), 369-385.

Kazakou, E., Adamidis, G. C., Baker, A. J., Reeves, R. D., Godino, M., & Dimitrakopoulos, P. G. (2010). Species adaptation in serpentine soils in Lesbos Island (Greece): metal hyperaccumulation and tolerance. Plant and Soil, 332(1), 369-385.

Kelepertzis, E. (2014). Accumulation of heavy metals in agricultural soils of Mediterranean: Insights from Argolida basin, Peloponnese, Greece. Geoderma, 221, 82-90.

Kelepertzis, E. (2014). Investigating the sources and potential health risks of environmental contaminants in the soils and drinking waters from the rural clusters in Thiva area (Greece). Ecotoxicology and environmental safety, 100, 258-265.

Kelepertzis, E. (2014). Investigating the sources and potential health risks of environmental contaminants in the soils and drinking waters from the rural clusters in Thiva area (Greece). Ecotoxicology and environmental safety, 100, 258-265.

Kelepertzis, E., & Stathopoulou, E. (2013). Availability of geogenic heavy metals in soils of Thiva town (central Greece). Environmental monitoring and assessment, 185(11), 9603-9618.

Lilli, M. A., Moraetis, D., Nikolaidis, N. P., Karatzas, G. P., & Kalogerakis, N. (2015). Characterization and mobility of geogenic chromium in soils and river bed sediments of Asopos basin. Journal of hazardous materials, 281, 12-19.

Massas, I., Ehaliotis, C., Gerontidis, S., & Sarris, E. (2009). Elevated heavy metal concentrations in top soils of an Aegean island town (Greece): total and available forms, origin and distribution. Environmental Monitoring and Assessment, 151(1), 105-116.

Massas, I., Ehaliotis, C., Kalivas, D., & Panagopoulou, G. (2010). Concentrations and availability indicators of soil heavy metals; the case of children's playgrounds in the city of Athens (Greece). Water, Air, & Soil Pollution, 212(1), 51-63.

Massas, I., Kalivas, D., Ehaliotis, C., & Gasparatos, D. (2013). Total and available heavy metal concentrations in soils of the Thriasio plain (Greece) and assessment of soil pollution indexes. Environmental Monitoring and Assessment, 185, 6751–6766.

Megremi, I., Vasilatos, C., Vassilakis, E., & Economou-Eliopoulos, M. (2019). Spatial diversity of Cr distribution in soil and groundwater sites in relation with land use management in a Mediterranean region: The case of C. Evia and Assopos-Thiva Basins, Greece. Science of The Total Environment, 651, 656-667.

Megremi, I., Vasilatos, C., Vassilakis, E., & Economou-Eliopoulos, M. (2019). Spatial diversity of Cr distribution in soil and groundwater sites in relation with land use management in a Mediterranean region: The case of C. Evia and Assopos-Thiva Basins, Greece. Science of The Total Environment, 651, 656-667.

Nanos, N., Grigoratos, T., Rodríguez Martín, J. A., & Samara, C. (2015). Scale-dependent correlations between soil heavy metals and As around four coal-fired power plants of northern Greece. Stochastic Environmental Research and Risk Assessment, 29(6), 1531-1543.

Nikolaidis, C., Orfanidis, M., Hauri, D., Mylonas, S., & Constantinidis, T. (2013). Public health risk assessment associated with heavy metal and arsenic exposure near an abandoned mine (Kirki, Greece). International journal of environmental health research, 23(6), 507-519.

Nikolaidis, C., Orfanidis, M., Hauri, D., Mylonas, S., & Constantinidis, T. (2013). Public health risk assessment associated with heavy metal and arsenic exposure near an abandoned mine (Kirki, Greece). International journal of environmental health research, 23(6), 507-519.

Noli, F., & Tsamos, P. (2016). Concentration of heavy metals and trace elements in soils, waters and vegetables and assessment of health risk in the vicinity of a lignite-fired power plant. Science of the Total Environment, 563, 377-385.

Noli, F., & Tsamos, P. (2016). Concentration of heavy metals and trace elements in soils, waters and vegetables and assessment of health risk in the vicinity of a lignite-fired power plant. Science of the Total Environment, 563, 377-385.

Panagopoulos, I., Karayannis, A., Adam, K., & Aravossis, K. (2009). Application of risk management techniques for the remediation of an old mining site in Greece. Waste Management, 29(5), 1739-1746.

Sawidis, T., Metentzoglou, E., Mitrakas, M., & Vasara, E. (2011). A study of chromium, cooper, and lead distribution from lignite fuels using cultivated and non-cultivated plants as biological monitors. Water, Air, & Soil Pollution, 220(1), 339-352.

Skordas, K., & Kelepertsis, A. (2005). Soil contamination by toxic metals in the cultivated region of Agia, Thessaly, Greece. Identification of sources of contamination. Environmental Geology, 48(4), 615-624.

Sofianska, E. (2015). Assessment of heavy metals contamination in the western Drama plain soils (Macedonia, N. Greece), using combined geochemistry and GIS mapping techniques. Πανελλήνια και Διεθνή Γεωγραφικά Συνέδρια, Συλλογή Πρακτικών, 248-263.

Sofianska, E., & Michailidis, K. (2015). Chemical assessment and fractionation of some heavy metals and arsenic in agricultural soils of the mining affected Drama plain, Macedonia, northern Greece. Environmental Monitoring and Assessment, 187(3), 1-16.

Sofianska, E., & Michailidis, K. (2016). Assessment of heavy metals contamination and potential ecological risk in soils affected by a former Mn mining activity, drama district, northern Greece. Soil and Sediment Contamination: An International Journal, 25(3), 296-312.

Topalidis, V., Harris, A., Hardaway, C. J., Benipal, G., & Douvris, C. (2017). Investigation of selected metals in soil samples exposed to agricultural and automobile activities in Macedonia, Greece using inductively coupled plasma-optical emission spectrometry. Microchemical Journal, 130, 213-220.

Tsompanidis (2016). Evaluation of the contaminated sites by industrial hazardous wastes in the region of Attica and the prefecture of Thessaloniki, Viotia, Evia, Kozani, Achaia, Heraklion, Magnisia, Kavala and Chalkidiki. ***Not public***

Tsoumani, Z. (2021). Investigation of heavy metal pollution of the soils of the areas of Inofyta – Oinoi – Schimatari (Unpublished master's thesis). Agricultural University of Athens, Athens.

Italy

In Italy the Environmental Protection and Research (ISPRA) and the Superior Health Institute (ISS) focusses on monitoring contaminated sites. A digital database of contaminated sites and the associated maps has been published in 2021: ISPRA (2021). Lo stato delle bonifiche dei siti contaminate in Italia: I dati regionali. ISPRA, Rapporti 337/21 ISBN 978-88-448-1043-6

The GIS data used in the report have not been made public sofar.

An overview of reviewed sites and the status of remediation is presented in this report for all regions. See underneath:

Tabella 3.1 *Siti oggetto di procedimento di bonifica registrati nelle anagrafi/banche dati delle Regioni/PA*

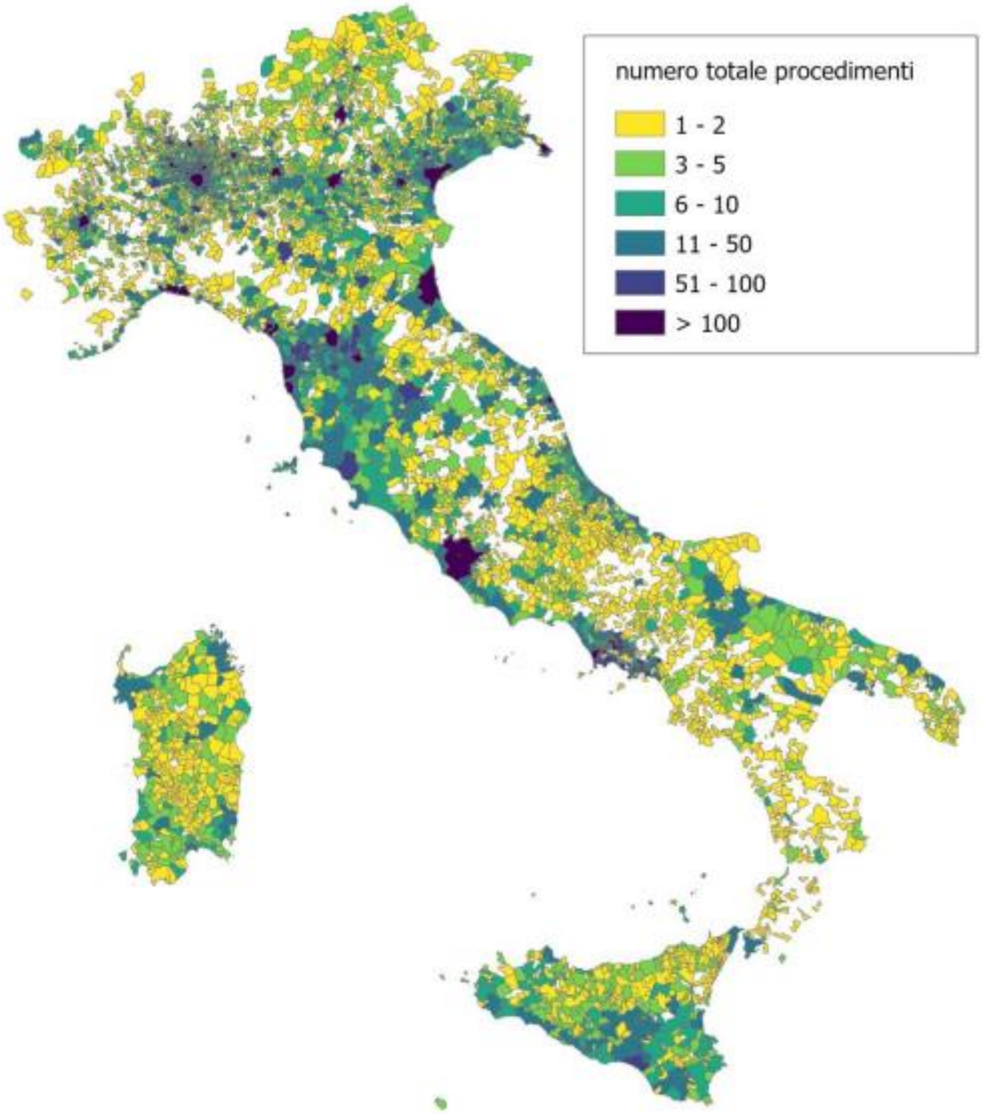
Regione/Provincia Autonoma	Siti con procedimento in corso	Siti con procedimento concluso	Totale siti oggetto di procedimento di bonifica
Abruzzo	862	346	1.208
Basilicata	237	65	302
Calabria	114	107	221
Campania	3.252	449	3.701
Emilia-Romagna	548	533	1.081
Friuli-Venezia Giulia	200	1.034	1.234
Lazio	1.015	197	1.212
Liguria	333	447	780
Lombardia	2.827	7.489	10.316
Marche	570	508	1.078
Molise	30	59	89
Piemonte	829	990	1.819
Puglia	357	184	541
Sardegna	592	463	1.055
Sicilia	1.060	325	1.385
Toscana	1.901	2.459	4.360
Trento	143	340	483
Bolzano	27	332	359
Umbria	91	76	167
Valle d'Aosta	31	168	199
Veneto	1.245	1.291	2.888
TOTALE ITALIA	16.264	17.862	34.478

In underneath table also area coverage is reported per region.

Tabella 6.7 *Superfici (e % siti che concorrono al calcolo) dei procedimenti conclusi senza e con intervento*

Regione / Provincia Autonoma	Procedimenti conclusi			
	Senza necessità intervento		Con intervento	
	Superficie (ha)	% siti con indicazione superficie	Superficie (ha)	% siti con indicazione superficie
Abruzzo	15	50%	4	77%
Basilicata	21	88%	9	78%
Calabria	109	35%	4	61%
Campania	494	92%	43	100%
Emilia-Romagna	840	100%	810	100%
Friuli-Venezia Giulia	8.870	47%	196	98%
Lazio	0	0%	0	0%
Liguria	11	6%	93	72%
Lombardia	5.346	82%	2.632	90%
Marche	6	0%	0	0%
Molise	15	98%	7	100%
Piemonte	477	38%	419	46%
Puglia	145	96%	149	100%
Sardegna	1.114	48%	392	29%
Sicilia	2	1%	13	52%
Toscana	4.921	100%	743	100%
Trento	114	96%	77	95%
Bolzano	0	0%	173	85%
Umbria	0	0%	0	0%
Valle d'Aosta	45	95%	7	95%
Veneto	144	14%	285	24%
ITALIA	**22.689**	**63%**	**6.056**	**77%**

The spatial data available, but not made publicly available are presented in underneath map:


numero totale procedimenti
1 - 2
3 - 5
6 - 10
11 - 50
51 - 100
> 100

Annex 2 Level of risk and need to remediate commodities from mines and suitability for phytoremediation

Commodities were ranked on two aspects:

- The risk of the commodity for human health and the need to reduce the risk, and the possibility to manage the mining site with biomass crops, such that risks of the commodity for human health are reduced.
  If the risk for human health of the commodity is high and can be remediated by phytoremediation, a score of 1 is attributed. If the risk for human health is low, for example in case of a sand pit, and therefore the need to remediate the commodity is low too, a score of 3 is attributed. A score of 2 indicates positions in between.

- The suitability of phytoremediation as a means to manage the commodity. The ranking in this aspect is as follows:
  4. How likely is it that plants can remove the chemical listed from the soil through extraction?
  5. How likely is it that by using plants (in combination with other chemicals), the compound of interest can be immobilized such that risks are reduced?
  6. How likely is it that plants are able to assist in in situ degradation of the compound of interest?

  The suitability scores for the three modes of phytoremediation are respectively listed in the columns *Extraction, Stabilization* and *Degradation/ volatilization*. The scoring for these ranking is: 1: likely or proven, 2: unknown or questionable, 3: most likely not effective.

Source data: Locations and commodities of mines: Minerals4EU database (https://geoera.eu/projects/mintell4eu7/); ranking: own elaboration; estimated area of mines: Minerals4EU database and Open Street Map.

		Potential for phytoremediation by:			
Commodity	Level of risk and need 1) High 2) Medium 3) low	Extraction	Stabilization	Degradation/ volatilization	Number of point locations of mines
Aluminium	3)	3	1	3	188
Antimony	1)	2	1	3	531
Arsenic	1)	1	1	3	763
Asphalt	1)	3	3	3	92
Baryte	1)	2	1	3	1410
Base Metal	1)	1	1	3	52
Bauxite	3)	3	1	3	128
Beryl	1)	2	2	3	28
Beryllium	1)	2	2	3	113
Bismuth	1)	2	2	3	136
Boron	1)	1	2	3	1
Cadmium	1)	1	1	3	16

		Potential for phytoremediation by:			
Commodity	Level of risk and need 1) High 2) Medium 3) low	Extraction	Stabilization	Degradation/ volatilization	Number of point locations of mines
Carbon	3)	3	3	1	71
Cesium	1)	2	1	3	8
Chemical Compound	1)	2	1	1-3	1
Chromium	1)	2	1	2	132
Coal	3)	3	3	3	2523
Cobalt	1)	2	1	3	525
Copper	1)	1	1	3	6220
Ferrous Metal	1)	1	1	3	583
Fluospar	1)	2	2	3	835
Gallium	1)	3	2	3	6
Gemstones	3)	3	3	3	161
Germanium	1)	2	2	3	15
Gold	1)	3	2	3	1580
Indium	1)	2	2	3	8
Industrial rocks and minerals	1)	3	3	3	14
Iron	3)	2	3	3	12708
Lead	1)	3	1	3	3840
Lithium	1)	3	2	3	112
Magnesia	3)	1	2	3	183
Magnesium	3)	1	2	3	3
Manganese	2)	1	2	3	600
Mercury	1)	2	1	2	133
Metal	1)	2	2	3	2
Molybdenum	1)	1	2	3	173
Natural Gas	3)	3	3	1	854
Nickel	1)	1	1	3	110
Niobium	1)	2	2	3	45
Oil	1)	3	3	1	199
Oil Shale	2)	3	3	1	102
Organic Material	3)	3	3	1	492
Palladium	1)	2	2	3	8
Phosphate	1)	1	2	3	920
Platinum	1)	3	2	3	14
Potash	3)	1	1	3	18
Potassium	3)	1	3	3	7
Precious Gemstones	3)	3	3	3	20
Rare Earth Element	1)	2	2	3	64
Rare Earth Oxide	1)	2	2	3	5

		Potential for phytoremediation by:			
Commodity	Level of risk and need 1) High 2) Medium 3) low	Extraction	Stabilization	Degradation/ volatilization	Number of point locations of mines
Rhodium	1)	2	2	3	1
Rubidium	1)	2	2	3	2
Salt	3)	1	3	3	513
Selenium	1)	1	2	3	2
Silver	1)	3	2	3	107
Strontium	1)	2	2	3	21
Sulphur	1)	1	3	3	1519
Tantalum	1)	2	2	3	16
Thorium	1)	2	2	3	17
Tin	1)	2	1	3	982
Titanium	1)	3	3	3	124
Tungsten	1)	2	2	3	846
Uranium	1)	1	2	3	1242
Vanadium	1)	2	2	3	15
Zinc	1)	1	1	3	486