Providing support in relation to the implementation of the EU Soil Thematic Strategy

The impact of soil degradation on human health

Revision: final

13 December 2019

Service contract No 07.0201/2016/742739/SER/ENV.D.l

PREPARED FOR:

EUROPEAN COMMISSION
DG ENVIRONMENT

PREPARED BY:

INSTITUTE OF SOIL SCIENCE AND PLANT CULTIVATION –
STATE RESEARCH INSTITUTE

Client European Commission, DG Environment
Title The impact of soil degradation on human health

Providing support in relation to the implementation of the EU Soil Thematic Strategy is a three-year contract commissioned by the Directorate-General (DG) for Environment (ENV) of the European Commission (Service contract No 07.0201/2016/742739/SER/ENV.D.I, duration 6 Dec 2016 to 5 Dec 2019).

The overall objective is to support the DG ENV with technical, scientific and socioeconomic aspects of soil protection and sustainable land use, in the context of the implementation of the non-legislative pillars (awareness raising, research and integration) of the Soil Thematic Strategy and the implementation of the European Soil Partnership.

The support includes the production of six in-depth reports that provide scientific background on a range of soil- and soil-policy-related issues in Europe, three policy briefs, logistic and organisational support for six workshops and the organisation and provision of content to the European website and the wiki platform on soil-related policy instruments.

The work is performed by: Deltares, The Netherlands (coordinator); IUNG Institute of Soil Science and Plant Cultivation, Poland; UFZ - Helmholtz Centre for Environmental Research, Germany; IAMZ - Mediterranean Agronomic Institute of Zaragoza, Spain; CSIC-EEAD Spanish National Research Council - Estación Experimental de Aula Dei, Spain.

Document Information

Title	The impact of soil degradation on human health
Lead Author	Agnieszka Klimkowicz-Pawlas
Contributors	Grzegorz Siebielec, Beata Suszek-Łopatka, Linda Maring
Distribution	DG ENV
Report Number	1.7

Disclaimer

The information and views set out in this report are those of the authors and do not necessarily reflect the official opinion of the European Commission. The European Commission does not guarantee the accuracy of the data included in this study. Neither the European Commission nor any person acting on the Commission's behalf may be held responsible for the use which may be made on the information contained therein.

Document History

Date	Version	Prepared by	Organisation	Approved by	Review by
15.05.2019	0.1	Agnieszka Klimkowicz-Pawlas	IUNG-PIB		
8.08.2019	0.2	Agnieszka Klimkowicz-Pawlas Beata Suszek-Łopatka Grzegorz Siebielec	IUNG-PIB		
23.09.2019	0.3	Agnieszka Klimkowicz-Pawlas	IUNG-PIB		
13.12.2019	0.4	Agnieszka Klimkowicz-Pawlas Grzegorz Siebielec	IUNG-PIB		DG ENV
16.12.2019	1.0	Agnieszka Klimkowicz-Pawlas	IUNG-PIB	Henriette Otter	Linda Maring

Contents

List of abbreviations ... 4
Executive summary ... 6
1 Introduction ... 8
1.1 Types of soil degradation ... 8
1.2 General characteristics of the relationship between soil and human health ... 9
2 Impact of soil contamination on human health ... 12
2.1 Methodologies applied for evaluating the impact of soil contamination on human health ... 14
2.1.1 Toxicological approach ... 14
2.1.2 Epidemiological approach ... 17
2.2 Pathways of exposure and possible health effects ... 18
2.2.1 Routes of exposure ... 18
2.2.2 Overview of health effects and factors influencing these effects ... 20
2.3 Measures to reduce health risk ... 25
3 Impact of soil sealing on human health ... 27
3.1 Methodologies applied for evaluating the impact of soil sealing on human health ... 27
3.2 Pathways of exposure and possible health effects ... 28
3.3 Measures to reduce health risk ... 30
4 Impact of erosion processes on human health ... 32
4.1 Methodologies applied for evaluating the impact of erosion on human health ... 32
4.2 Pathways of exposure and possible health effects ... 33
4.3 Measures to reduce health risk ... 34
5 Impact of other degradation processes (flooding and landslides) on human health ... 36
5.1 Methodologies applied for evaluating the impact of floods and landslides on human health 37
5.2 Pathways of exposure and possible health effects ... 37
5.3 Measures to reduce health risk ... 41
6 Links between losses of soil biodiversity and human health ... 43
6.1 Pathogens versus bacteria, potential negative effect ... 46
6.2 Antimicrobial resistance ... 47
6.3 Hampered nutrient cycling and degradation processes ... 49

7 The positive role of soil on human health 51
7.1 Use of soil biodiversity for pharmaceuticals 51
7.2 Sorption of pathogens and contaminants 52
7.3 Positive impact on physical and mental health 54
8 Conclusions and recommendations 56
9 References 59
Annex I Case studies on epidemiological effects 73
Annex II Characteristics of the main pollutants 76
Annex III Characteristics of emerging contaminants 81

List of abbreviations

Abbreviation	Explanation
ADI	acceptable daily intake
ARG	antibiotic resistance genes
BARGE	Bioavailability Research Group of Europe
BTEX	Benzene, Toluene, Ethylbenzene and Xylenes
BMD	benchmark dose
CBD	Convention on Biological Diversity
CEV	critical exposure value
CLEA	Contaminated Land Exposure Assessment Tool
DALY	disability adjusted life years
DBP	disinfection by-products
DDE	dichlorodiphenyldichloroethylene
DDT	dichlorodiphenyltrichloroethane
EDC	endocrine disrupting compounds
EEA	European Environment Agency
EM-DAT	International Disaster Database
EP	emerging contaminants/pollutants
E-PRTR	European Pollutant Release and Transfer Register
EU	European Union
FAO	Food and Agriculture Organization of the United Nations
GBD	Global Burden of Diseases
GHG	greenhouse gas
GMO	genetically modified organism
GSP	Global Soil Partnership
HI	hazard index
HHRA	human health risk assessment
HM	heavy metals
IARC	International Agency for Research on Cancers
ICSHNet	Industrially Contaminated Sites and Health Network
IPCS	International Programme on Chemical Safety
IRIS	Integrated Risk Information System

Abbreviation	Explanation
ITPS	Intergovernmental Technical Panel on Soils
LOAEL	lowest-adverse-effect-level
NPCS	National Priority Contaminated Sites
NOAEL	no-observed-adverse-effect-level
OR	odds ratio
PAH	polycyclic aromatic hydrocarbons
PCB	polychlorinated biphenyls
PCDD	polychlorinated dibenzodioxin
PCDF	polychlorinated dibenzofuran
PFAS	perfluoroalkyl and polyfluoroalkyl substances
PFC	perfluorinated compounds
PM	particulate matter
POP	persistent organic pollutants
PTSD	post-traumatic stress disorders
RAIS	Risk Assessment Information System
RAR	Risk Assessment Report
RfC	reference concentration
RfD	reference dose
SMR	standardised mortality ratios
TDI	tolerable daily intake
TEQ	toxicity equivalents
TOXNET	TOXicology Data NETwork
VP	veterinary pharmaceuticals
WHO	World Health Organization

Executive summary

The soil condition can affect human health either directly or indirectly. Although links between soil and human health have been known for thousands of years (since Hippocrates), scientists from various fields (e.g., soil science, agronomy, biology, medicine, etc.) are still performing research to better understand the impact of soil quality on human health.

This report provides a comprehensive overview of impacts driven by different types of soil degradation on human health. The relationships between soil contamination and human health and the reported case studies that document the linkage between epidemiologic and environmental data are described in Chapter 2. Assessment of the impact of soil pollution on human health refers to two approaches: toxicological and epidemiological. Different exposure models or biomonitoring approaches are applied within European countries to predict exposure to soil contamination. Risk assessment methods are usually based on the single pollutant assessment, do not include the interaction between chemical and non-chemical stressors and depend on toxicological data that requires the use of arbitrary safety factors. The majority of available data on epidemiological effects caused by soil contamination is derived from scientific papers. The most frequently investigated health impacts are mortality, hospital admission and morbidity, all based on periodically collected statistics and hospitalisation records. One challenge is to investigate the risks of emerging pollutants (EP) because most of their potential ecological and health impacts are still not identified. The only available information on health effects of these compounds is based on toxicological data.

The report also provides an overview of the recent research on how other types of soil degradation (e.g., soil sealing, erosion, loss of biodiversity) affect human health.

Uncontrolled urbanisation and soil sealing might strongly affect human health (Chapter 3). Improper land management might affect humans through urban heat islands, contaminant toxicity, air pollution (dust, particulate matter), disconnection of citizens from green areas, limited recreation opportunities, the risk of flooding, etc. An increase in mortality due to heat waves has been reported during summer in densely sealed urban areas e.g., France and the Netherlands (Filluel et al., 2006). Limited access to green areas might lead to deterioration of mental health (increase in nervousness and depression), social cohesion and physical condition.

There is limited information available in Europe on the links between soil erosion (both water and wind) and human health (Chapter 4). Recently, an analysis of long-range transport of Saharan dust indicated risks for human health. Transboundary soil degradation may adversely affect human health through the transport of particulate matter, anthropogenic pollutants or microorganisms. Exposure to airborne dust particles causes respiratory and cardiopulmonary disorders and leads to increased mortality, mainly among the elderly. However, it is very difficult to clearly evaluate health effects of soil erosion due to the many confounding factors (e.g., heat waves, humidity or flu epidemic weeks) that occur simultaneously with the dust episodes.

The literature on the health impacts of flooding and landslides in Europe is limited, and it is often difficult to quantify the effects or link these specifically to floods or landslides (Chapter 5). The adverse human health consequences of flooding are relatively well documented in the UK. Direct effects result from the exposure to the water or flooded environment and include mortality, injuries, intoxication and increased incidence of mental disorders. Indirect consequences are those associated with the damage caused by the water to the natural and built environment and include infectious diseases, malnutrition, poverty-related diseases and diseases associated with displaced populations. There is a very limited set of data with regards to links between landslides and human health outcomes. Only the number of fatalities due to landslides has been reported in some European countries (e.g., Turkey, Italy, Portugal).

Soil biodiversity can have both direct and indirect effects on human health (Chapter 6). Biodiversity loss can lead to the emergence and transmission of infectious diseases and cause malnutrition through the loss of food. Furthermore, as a consequence of global environmental changes and associated losses in soil biodiversity, we can lose a possible source of antibiotics, medicines and the biological controls needed to prevent human, animal and plant disease. On the other hand, a rapid increase in antimicrobial resistance has been observed; it threatens the prevention and treatment of diseases caused by bacteria, fungi and parasites.

Apart from the negative effects on human health, some authors underline numerous benefits that soils provide to human health and well-being (Chapter 7).

There are still many knowledge gaps, mainly on the human health impact of soil sealing, erosion and decline in biodiversity. So far, scientists and environmental managers have focused their attention mainly on recognising the state of soil degradation and risk prevention. Analysis of the soil degradation impact on human health is usually based on the probability of occurrence of the harmful effects, while there is little high quality epidemiological data that refers to the actual health consequences.

1 Introduction

Soil has a profound impact on the health and well-being of humans; this effect can be either positive or negative and direct or indirect (Brevik et al., 2018a; Steffan et al., 2018). Soil affects human health directly through the ingestion or inhalation of soil or its constituents. There are also myriad pathways of indirect impact, e.g., through the quantity and quality of food. Some effects can be detrimental to health if toxic substances or pathogenic organisms enter the food chain and are passed to humans (Oliver and Gregory, 2015). Indirect soil degradation effects on human health might be connected to loss of soil or its coverage by impermeable materials that results in deterioration of human living conditions in cities. The role of soils for human health is recognised and underlined by international authorities, such as the United Nations, the World Health Organization (WHO), the Convention on Biological Diversity (CBD) and the Global Soil Partnership (GSP) of the FAO.

1.1 Types of soil degradation

Soil is a three-dimensional natural body of Earth's surface that is essential to numerous ecosystem functions, including production of biomass and net primary productivity, climate change mitigation, storage and purification of water, biodegradation of pollutants and provision of plant nutrients. Soil quality refers to the capacity of the soil to perform several of these ecosystem functions (Lal, 2009, 2015). Soil degradation implies the decline in the quantity and quality of soil through natural and anthropogenic perturbations. According to the FAO (2019), soil degradation is defined as "a change in the soil health status resulting in a diminished capacity of the ecosystem to provide goods and services for its beneficiaries. Degraded soils (…) do not provide the normal goods and services of the particular soil in its ecosystem." Soil is under continuous pressure due to an increasing global population, which will only become more and more dependent on the availability and fertility of soil. The types of soil degradation differ and include physical, chemical, biological and ecological processes (Figure 1.1).

Soil physical degradation results in a reduction in structural attributes (soil pore geometry and continuity) and thus increases the soil susceptibility to compaction, reduces water infiltration, accelerates wind and water erosion, amplifies soil temperature fluctuations and hastens desertification. Soil chemical degradation includes acidification, salinisation, nutrient depletion, reduced cation exchange capacity, increased aluminium (Al) and manganese (Mn) toxicities, calcium (Ca) and magnesium (Mg) deficiency, leaching of essential plant nutrients, contamination by wastes or by-products from industry or agriculture (Brevik and Burgess, 2013; Lal, 2009, 2015). Soil biological degradation reflects depletion of soil organic matter, reduction in the activity and species diversity of soil organisms and a decline in soil carbon (C) sink capacity. Ecological degradation is a function of the other three types of soil degradation and leads to disruption in ecosystem functions (e.g., elemental cycling, water infiltration and purification; Lal, 2015).

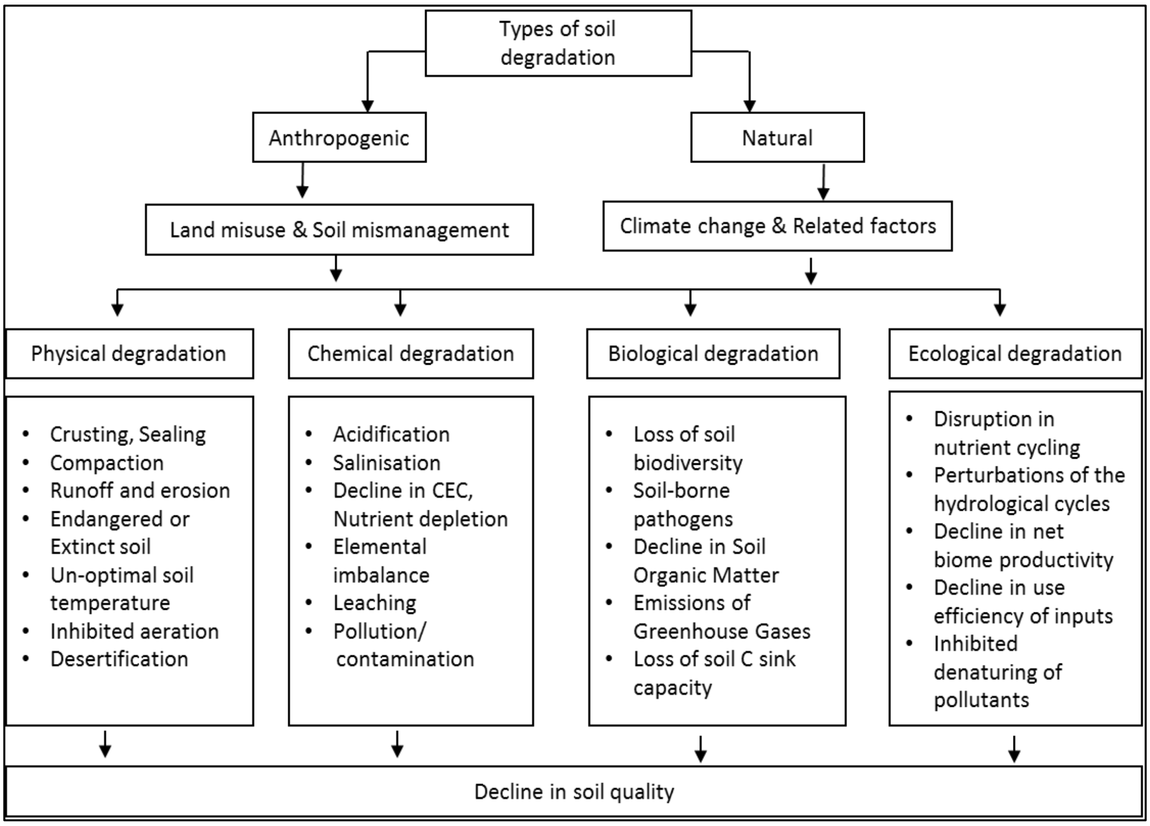

Figure 1.1: Types of soil degradation (modified, based on Lal, 2015).

1.2 General characteristics of the relationship between soil and human health

Medical practitioners and the public often define human health as the absence of disease (Sandifer et al., 2015). However, the WHO defines health more broadly as: "a state of physical, mental and social well-being and not merely the absence of disease or infirmity" (WHO Constitution, 1946).[1]

Soils contribute directly and indirectly to human health. The negative impact of soil degradation on human health might be attributed to deterioration of a range of ecosystem services provided by soil. For example, soil **regulating** services relevant to human health include greenhouse gas (GHG) cycling and pathogen regulation through biological control (Aerts et al., 2018; Brevik et al., 2018b). Soil contributes to GHG balance, which affects global climate change trends and regulates local climate extremes, such as heat waves and dryness. Soils can positively influence human health by acting as a filter to inactivate or remove hazardous materials and pathogens. Worsening of some soil properties (acidification, organic matter level and soil structure) limits the filtering ability of the soil, a phenomenon that might lead to excessive release of contaminants or nutrients into water or the food chain. One of the main ecological services provided by soil is filtering, buffering and transforming inorganic and organic contaminants, all of which ensure the good quality of groundwater and safe food production and thus protect human health.

[1] WHO Constitution, available at: https://www.who.int/about/who-we-are/constitution

Soil is a dynamic reservoir of biodiversity. Many beneficial organisms have been isolated from soil: plant growth promoting and disease suppressive microbes used as inoculants and foliar inoculants and inoculants used for bioremediation of toxic compounds in the environment. Soil biodiversity is highly recognised as an important feature of healthy soil, and imbalances advantage harmful over beneficial organisms (Sandifer et al., 2015; Wall et al., 2015). The positive role of soils on human health is associated with prevention of immunity-related disorders and a lower incidence of allergic diseases as a result of exposure to soil microorganisms.

Protecting soil quality, manifested by proper soil pH and balanced organic matter and nutrients, and increasing landscape biodiversity through habitat conservation or restoration have positive effects on several aspects of well-being, including physical and mental health benefits associated with spending time in nature (Clark et al., 2014; WHO, 2015; Kilpatrick et al., 2017; Aerts et al., 2018). Short-term exposure to green spaces (e.g., forests, urban parks and gardens) may reduce stress and depressive symptoms, restore attention fatigue, increase positive emotions (i.e., vitality, energy, pleasure) and improve self-esteem and mood (Clark et al., 2014; Sandifer et al., 2015; Aerts et al., 2018). Long-term exposure to biodiverse natural environments can diminish all-cause, respiratory, cardiovascular and cancer mortality and positively affects respiratory and mental health (Aerts et al., 2018).

Regulating soil services involves its capability to retain water, both in terms of providing enough water to grow crops and green areas and regulating local climate or protecting citizens against flooding.

Provisioning services ensure a sufficient supply of nutritious plant- and animal-based foods, micronutrients and medications, particularly antibiotics and anticancer drugs (Huynen et al., 2004). Soil provides essential nutrients through plants to animals and humans. Thus, insufficient nutrients in soil may cause malnutrition and deficiencies in humans. Soil also plays a key role in ensuring food safety and security. Therefore, protecting a sufficient pool of good quality soil, which enables the production of sufficient amounts of high-quality and nutritious food, will be fundamental to the environmental, social and economic stability of the world.

Provisioning services also include building supplies that allow the construction of housing that protects human health from inclement weather, fibres for clothing that aid in body heat regulation and fuel to heat houses during cold weather (Brevik et al., 2018b).

Exposure to soil microorganisms is thought to be important in the prevention of allergies and other immunity-related disorders. Soil is also a natural source of medicines: antibiotics and other pharmaceuticals. On the other hand, overuse of antibiotics in treating human and animal illness may cause the spread of antibiotic resistance genes in the environment.

Cultural services are important for healing, stress reduction, recreation and contribute to the maintenance of mental health and cognitive development (Huynen et al., 2004; Brevik et al., 2018b).

Human health problems can be caused by soil degradation through various exposure routes: toxic levels of trace elements and hazardous organic contaminants or disease-causing organisms can enter the food chain from the soil; direct encounters with pathogenic organisms; production of nutrient-deficient crops from soils, contributing to malnutrition; mass movement events that can injure or kill humans and direct inhalation of airborne soil particles causing respiratory problems (Brevik et al., 2018a). Indirect effects include the discovery and development of antibiotics and other pharmaceuticals from soil, mobilisation of chemicals with eroded soil particles and water purification as it infiltrates the soil (Brevik et al., 2018a).

In our report, the impact of soil degradation on human health is discussed for a range of soil degradation types (threats to soil): contamination with special attention to emerging contaminants, erosion, sealing, loss of soil biodiversity, flooding and landslides. Focus is given to more direct effects of soil degradation on human health, through local exposure pathways, in order to raise awareness on the need to protect soils. Therefore, soil threats like the loss of organic matter or compaction are not discussed in detail because they might impact humans less directly or through global processes, such as contribution to GHG emissions, lower crop productivity or contamination of surface water resulting from soil compaction-induced nutrient run-off. The mechanisms and scale of the impact, methodologies used to evaluate the impact and measures to counteract the soil degradation-related threats are discussed in the following chapters.

2 Impact of soil contamination on human health

Soil contamination is described as one of the main threats to soil in the Thematic Strategy for Soil Protection (COM(2006) 231). The Intergovernmental Technical Panel of Soil (ITPS) identified soil pollution as the third most important threat (after soil sealing and salinisation) to soil functions in Europe (FAO and ITPS, 2015). Until recently, the term “contamination” was used as a synonym for “pollution”. Recently, the ITPS under the Global Soil Partnership (GSP) has formalised definitions of these two terms (FAO and ITPS, 2015; Rodríguez-Eugenio et al., 2018). Soil contamination refers to the presence of a chemical or substance at a higher-than-normal concentration but does not cause harm to target organisms, whereas soil pollution occurs when the chemical or substance is present at a higher concentration and leads to adverse effects on organisms.

Soil contamination can be local or diffuse. Diffuse soil contamination does not have a single or easily identifiable sources and usually covers large areas (FAO and ITPS, 2015; Stolte et al., 2016). Diffuse soil contamination occurs where emission, transformation and dilution of contaminants in other media have occurred prior to their transfer to soil. Sources of diffuse soil contamination include the long-distance transport of dust, car emissions, long-term use of low-quality fertilisers, intensive application of agrochemicals and manure that contain residues of veterinary drugs and uncontrolled sewage sludge application (Carre et al., 2015; FAO and ITPS, 2015; Stolte et al., 2016; Rodríguez-Eugenio et al., 2018). Due to the complexity of diffuse sources, the magnitude and real extent of diffuse contamination at the EU level is difficult to evaluate (Rodríguez-Eugenio et al., 2018).

Local soil contamination (related with a point-source) may result from intensive industrial activities, inadequate waste disposal, mining and smelting, military activities or accidents as well as application of pesticides and fertilisers (Stolte et al., 2016; Pérez and Rodríguez-Eugenio, 2018). It was estimated (Panagos et al., 2013) that municipal and industrial waste treatment and disposal contribute most to soil contamination in Europe (37%), followed by the industrial/commercial sector (33%). Large volumes of waste and the intense use of chemicals during past decades have resulted in numerous contaminated sites across Europe. These sites are important sources of pollution and may pose significant hazards to ecosystem and human health (Panagos et al., 2012; Rodríguez-Eugenio et al., 2018). In Europe, there are almost 3 million sites where potentially polluting activities took/are taking place. These activities have caused contamination in more than 362,000 sites, of which 150,000 require remediation (Pérez and Rodríguez-Eugenio, 2018).

To deal with soil contamination, the first WHO Collaborating Centre on Environmental Health in Contaminated Sites was established in 2013, in Italy in the National Institute of Health. In 2015, the COST Action IS1408 “Industrially Contaminated Sites and Health Network” (ICSHNet) was created. This COST Action set up a European Network of experts and relevant institutions and developed a common framework for research and response with the production of information for decision makers who have deal with contaminated sites (Shaddick et al., 2018). The network aims to clarify knowledge gaps and research priorities, support the collection of relevant data, stimulate development of harmonised methodology and develop guidance on risk and health impact assessment in contaminated sites. ICSHNet currently involves the WHO, EU and European Community bodies and public environmental health institution from 33 countries (Hoek et al., 2018; Shaddick et al., 2018).

Soil contaminants occur in various forms, such as organic (e.g., polycyclic aromatic hydrocarbons [PAH], pesticide residues and antibiotics), inorganic (metals and metalloids) and particulate contaminants (Stolte et al., 2016). Heavy metals (HM) and mineral oil are the contaminants that occur most often: They occur in 60% of contamination cases (Panagos et al., 2013). In Europe, HM contamination concerns mainly regions with the presence of historical mining and heavy industry (Tóth et al., 2016). Mineral oil (a complex mixture of numerous hydrocarbons) is spread to the environment as a result of the refining, transport and storage of petroleum or by accidents (Gallego et al., 2001; Becker et al., 2006; Panagos et al., 2013).

More than 900 emerging pollutants (EP) have been found in the European environment (Geissen et al., 2015; NORMAN, 2019[2]). EP are divided into more than 20 groups related to their origin, e.g., perfluorinated compounds (PFC), disinfection by-products (DBP), gasoline additives, man-made nanomaterials, human pharmaceuticals, veterinary pharmaceuticals (VP), endocrine-disrupting compounds (EDC), cyanotoxins and rare earth elements (see Table A3 in Annex III). Currently, little is known about their fate, occurrence and hazardous effects to the natural environment and human health (Gavrilescu et al., 2014). Therefore, improving knowledge about EP is a crucial and necessary challenge. Future EP research should consider their substitution and entire lifecycle, from the source of emission to their removal through treatment and remediation techniques, including the impacts and risks they may pose to the environment and human health.

According to the WHO, approximately a quarter of diseases that humans face today occur due to prolonged exposure to pollution, of which 70% are non-communicable diseases (Landrigan et al., 2018). Different types of pollutants can impact various organs or systems (Valentin et al., 2013). Exposure to polychlorinated biphenyls (PCB) and dioxins may result in immunotoxicity, reproductive diseases and neurotoxicity (Bornman et al., 2018). Some contaminants (e.g., lead, DDT and PAH) may be carcinogenic, mutagenic or teratogenic (Rodgers et al., 2018). The presence of emerging contaminants in soil (e.g., perfluoroalkyl and polyfluoroalkyl substances [PFAS]) may result in endocrine disruption, metabolic diseases and different types of cancer (Lei et al., 2015, Winkens et al., 2017). Chronic exposure to lead is an established risk factor for hypertension, renal failure and cardiovascular diseases (Landrigan et al., 2018). The Global Burden Diseases (GBD) study indicates that deaths in 2015 that were attributable to lead are as follows: cardiovascular diseases (465,000), ischaemic heart disease (240,000), cerebrovascular disease (155,000), ischaemic stroke (68,000), haemorrhagic stroke (87,000), hypertensive heart disease (47,000) and chronic kidney disease (28,000; after Landrigan et al., 2018).

[2] NORMAN - Network of reference laboratories, research centres and related organisations for monitoring of emerging environmental substances (https://www.norman-network.net/?q=node/19)

2.1 Methodologies applied for evaluating the impact of soil contamination on human health

Human health risk assessment (HHRA) can be defined as a procedure for characterising the nature and magnitude of health risk to human beings of chemical contaminants and other stressors that may be present in the environment (WHO, 2013a). Two main approaches are used to assess the impact of pollution on human health: risk assessment and epidemiology (WHO, 2013a; de Sario et al., 2018; Hoek et al., 2018).

2.1.1 Toxicological approach

The first approach is the **traditional toxicological** approach, based on the multistage risk assessment procedure, which leads to theoretical estimates of the potential health risk (Figure 2.1).

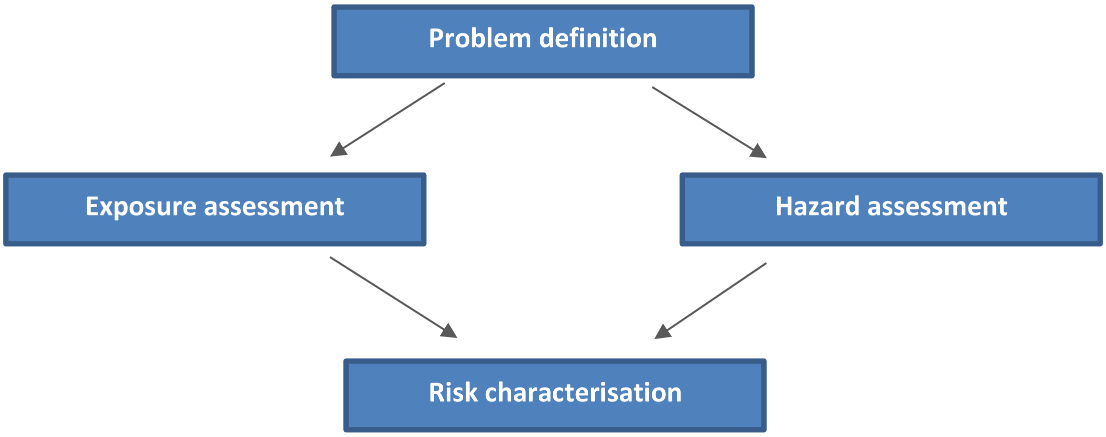

Figure 2.1: Generic risk assessment framework (Swartjes, 2015).

The toxicological approach is based on the evaluation of a single substance, often considers the worst-case scenario and is commonly applied for chemical safety issues. A conceptual exposure scenario is developed in which the contamination sources, exposure pathways and receptors are identified. Available data on the level of soil contamination with a specific chemical agent are compared to the tolerable dose as a reference value, and finally the potential health risk is quantified (Pirastu et al., 2013; WHO, 2013a).

HHRA with regard to contaminated sites includes two elements: exposure assessment and hazard assessment (Swartjes et al., 2012). **Exposure** can be measured directly, estimated using models or generalised from existing data. The most direct way to assess human exposure from soil contamination is to measure the actual body burden through biomonitoring. In practice, this action requires sampling and measuring body fluids (blood or urine) or body tissue (e.g., nails, hair or skin tissue; Henríquez-Hernández et al., 2011; Santonen et al., 2017; Bornman et al., 2018; Pérez et al., 2019). However, the exposure assessment is mainly based on measuring the concentration of pollutants in the edible parts of vegetables and indoor air (Swartjes, 2015).

A very useful and practical methodological for possibly assessing human exposure is modelling human exposure by combining contaminant distribution over the soil phases, contaminant transfer from the soil and calculation of direct and indirect exposure of humans.

Different exposure models are used within European countries e.g., Cetoxhuman in Denmark, Contaminated Land Exposure Assessment Tool (CLEA) in the UK (Hosford, 2009), CSOIL in the Netherlands (Swartjes et al., 2012; Cachada et al., 2016), RISKNET in Italy, S-Risk in Belgium and RAIS[3] (developed in the USA but used in Europe; Hoek et al., 2018).

Hazard assessment combines hazard identification (which defines the type and nature of the adverse effects of a contaminant) and hazard characterisation, where adverse effects are quantified and dose-response is assessed with the use of the critical exposure value (CEV; Langley, 2011; Cachada et al., 2016). Several international databases contain CEV values, such as the International Programme on Chemical Safety of the WHO (IPCS)[4], the International Agency for Research on Cancer (IARC)[5], European Risk Assessment Reports (RARs)[6], Integrated Risk Information System (IRIS)[7] and TOXicology Data NETwork (TOXNET)[8].

Given the lack of epidemiological data, most CEV values are derived from experimental studies with test animals. Different types of studies are reported in literature: acute toxicity, sub-chronic toxicity, reproductive toxicity, developmental toxicity and genotoxicity (Langley, 2011). In animal toxicity studies, the no-observed-adverse-effect-level (NOAEL) and lowest-adverse-effect-level (LOAEL) are determined and then transferred into effect measurements applicable to humans through the use of assessment factors. A more advanced method is the calculation of a benchmark dose (BMD) from dose-response data. In scientific papers, different names are used for the toxicity reference value, e.g., tolerable daily intake (TDI), acceptable daily intake (ADI), reference dose (RfD) or reference concentration (RfC; Briggs, 2003; Langley, 2011; Swartjes and Cornelis, 2011; Augustsson et al., 2018). For carcinogenic compounds, acceptable excess cancer risk values are used in risk assessment for human health. With regards to contaminated land, these values range from 1 in 10^4 and 1 in 10^6 lifelong exposed individuals (Cachada et al., 2016; Qu et al., 2019).

One-tier single exposure assessments have many limitations, and thus the best way to deal with human health is a **tiered approach based on multiple lines of evidence** (Swartjes, 2015). An example of a tiered approach is the methodology developed in the Netherlands (Figure 2.2).

---

[3] Risk Assessment Information System (RAIS, https://rais.ornl.gov/), sponsored by the US Department of Energy, Oak Ridge Operations Office, is a source of data for chemicals and radionuclides e.g., Toxicity Profiles, Chemical Tools, Radionuclides Tools, etc.
[4] The International Programme on Chemical Safety (IPCS, https://www.who.int/ipcs/en/), a programme of the WHO that implements activities related to chemical safety and management of chemicals.
[5] The International Agency for Research on Cancer (IARC, https://www.iarc.fr/), a specialised cancer agency of the WHO, promotes international collaboration in cancer research. IARC Monographs identify environmental factors that can increase the risk of human cancer.
[6] Risk Assessment Reports (RARs, https://echa.europa.eu) published by the European Chemical Bureau of the EU Commission
[7] Integrated Risk Information System (IRIS, https://www.epa.gov/iris) developed by the United States Environmental Protection Agency (USEPA) to identify and characterise the health hazards of chemicals found in the environment
[8] The TOXicology Data NETwork (TOXNET, https://toxnet.nlm.nih.gov/) is managed by the United States National Library of Medicine (NLM). TOXNET combines databases covering chemicals and drugs, diseases and the environment, environmental health, occupational safety and health, poisoning, risk assessment and regulations, and toxicology.

This approach generates data in sequential steps with increasing complexity and lower uncertainties. In each consecutive tier, more site-specific information is considered. It is advised to move to the next tier only if results from the previous tier are not satisfactory. This procedure starts with a preliminary qualitative evaluation of the possibility for adverse human health effects, comparison of the actual total soil concentration with a critical soil concentration and if a risk is unacceptable, site-specific risk calculation (Swartjes, 2015).

Many risk assessment procedures have focused on the measurement of the impact on human health of individual chemicals or on the individual pathways of exposure. Recent studies are now also focusing on a **multipollutant approach** and evaluation of health effects of pollutant mixtures. Examples of such research are the EU projects INTARESE[9] and HEIMSTA,[10] both of which integrate epidemiological models that provide an overall qualitative/quantitative assessment of the impact on health of environmental stressors by considering multiple risk factors, combined exposure pathways and cumulative health outcomes (Briggs, 2008). A simplified method to deal with combined exposure is application of different hazard indexes (HI; ratio between estimated exposure and critical exposure value). If the sum of these indices exceeds the value of 1, there is the possibility of unacceptable human health risk (Cachada et al., 2012; Swartjes, 2015; Qu et al., 2019). Another way of assessing combined exposure is the use of toxicity equivalents (TEQ). TEQ can be applied for contaminants with the same mode of toxicity, and they often are used to assess human health risk from dioxins, dioxin-like compounds and PAH. The so-called disability adjusted life years (DALY) characterise the severity of exposure by accounting for both mortality and morbidity (Briggs, 2003; Swartjes, 2015).

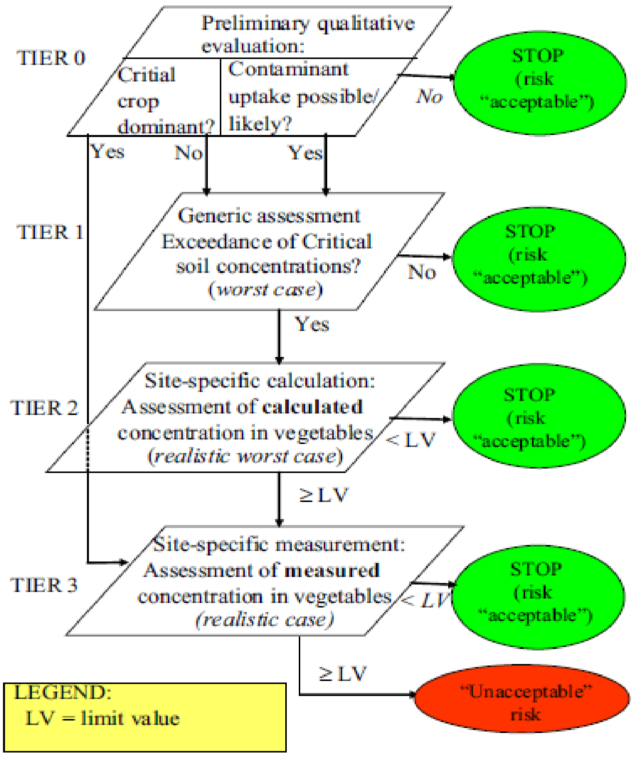

Figure 2.2: An example of a tiered approach for assessing risks, as used in the Netherlands (Swartjes, 2015).

[9] INTARESE - Integrated Assessment of Health Risk of Environmental Stressors in Europe
[10] HEIMSTA - Health and Environment Integrated Methodology and Toolbox for Scenario Development

For new industrial chemicals, human data is usually unavailable. In this situation, toxicological hazard may be predicted from the chemical's properties or the characteristics of structurally related chemicals, given that factors such as volatility, solubility and molecular weight can indicate the likely extent of absorption across biological membranes. Studies of possible health effects caused by, for example, EP, are based on estimation of animal health or extrapolation of data from testing chemicals with a similar structure. Application of animal-based toxicological studies in the assessment of human health risk have some limitations: they are based on the single pollutant assessment, performed under strictly controlled laboratory conditions and do not include the interaction between chemical and non-chemical stressors.

2.1.2 Epidemiological approach

The second approach used to evaluate human health consequences of soil pollution is **epidemiological**—based on epidemiological evidence and dose-response functions—that involves estimation of the environmental burden of diseases. In this approach, health statistics on mortality and morbidity of the resident population is used (WHO, 2013a; de Sario et al., 2018; Hoek et al., 2018). Two distinct approaches are used to investigate the potential effects of soil on human health in the epidemiological studies: aggregate and individual (Hough, 2007; Hoek et al., 2018). The **individual-level approach** comprises the relationship between health outcome and exposure to the stress factor at an individual level; it includes prospective or retrospective studies (Hough, 2007). In prospective studies, the health status of individuals in a cohort is monitored over time. Such studies are very expensive and time consuming. In retrospective studies, case-control designs are applied in which new cases of health outcomes are selected and matched to a control (Hough, 2007). **Aggregate-level approaches** relate spatial soil characteristics to geographic incidence of disease.

Epidemiological studies can be grouped into three categories (Langley, 2011; Pasetto et al., 2016; de Sario et al., 2018; Hoek et al., 2018):

1. **Description of the health profile of residents**. These epidemiological methods use routinely collected health data, analysed at a small-area level. Such a method does not require *ad hoc* data collection. Health effects are estimated at the group level and exposure to contaminants is attributed to each group using aggregate measures or qualitative indicators.
2. **Analysis of the casual associations between environmental exposures and health effects** based on an analytical epidemiological study, using individual data on exposure and health outcomes. To evaluate the effects from the long-time exposure to pollution, the most suitable studies are: ecological (at a small-area level), cross-sectional (e.g., biomonitoring), cohort and case-control. For short-term effect time series, case-crossover and panel studies can be adapted.
3. **Planning epidemiological surveillance of the population's health profile**. Such studies can be used to evaluate the effectiveness of remediation and include a descriptive approach based on routinely collected data at small-area level, analytical longitudinal studies (e.g., residential cohort studies) and cross-sectional studies based on biomonitoring.

The **SIENTIERI project** is worth mentioning with regards to methods that describe a population health profile at the **aggregate level**. The SENTIERI approach was developed in Italy; it is based on routinely collected health data on mortality and morbidity at the small-area level and involves two phases (Pirastu et al., 2013; Pasetto et al., 2016). The first phase includes identification and classification of contaminated sites and additional definition of populations affected by contamination (WHO, 2013a). In the second phase, small-area statistics for all causes with a possible environmental aetiology are computed for each contaminated site by using mortality and morbidity indicators (WHO, 2013a).

Most of the epidemiological studies on the human health effects in contaminated sites used an ecological design, based on standardised mortality or morbidity ratios, hospitalisation, cancer in adult population and congenital anomalies and birth outcomes (Table 2.1; Marra et al., 2012; Pirastu et al., 2013; Fernández-Navarro et al., 2017). Some studies used a case-control design (Maifredi et al., 2011), a cross-sectional design (Donato et al., 2008) or a retrospective cohort design that focused on mortality ratios, cancer in adults or hospitalisations (Martin-Olmendo et al., 2018). A detailed description of several case-studies is provided in Annex I.

Table 2.1: Routine health data used in epidemiological studies conducted in industrially contaminated sites (Martin-Olmendo et al., 2018).

Health outcomes	Primary source of health data	Study design
Mortality	Routinely collected vital statistics, cancer registries	Ecological Retrospective cohort
Morbidity	Specific morbidity registries	Descriptive; study specific collection
Hospitalisations	Routinely collected hospitalisation records	Ecological Retrospective cohort
Cancer incidence (childhood)	Cancer registers and National Birth registries	Population based case-control
Cancer incidence (adults)	Cancer registries and routinely vital statistics	Ecological Retrospective cohort
Congenital anomalies and birth outcomes	Routinely collected vital statistics, congenital anomalies registries	Ecological Retrospective cohort Multicentre case-control

2.2 Pathways of exposure and possible health effects

2.2.1 Routes of exposure

The routes of human exposure to soil contaminants depend on the type of contaminant, the soil properties, the type of the receptor and the conditions and activities at a particulate site (Hosford, 2009; Hoek et al., 2018; Rodríguez-Eugenio et al., 2018). Humans are exposed to soil via three common pathways: oral (ingestion), respiration and skin absorption or penetration (Table 2.2).

Soil ingestion can occur incidentally (during hand-to-mouth contact or consumption of plants or animals that have accumulated large amounts of soil), or intentionally, known as geophagy[11] (Sing and Sing, 2010; Brevik and Burgess, 2013; Science Communication Unit, University of the West of England, 2013; Steffan et al., 2018). Ingested soil can potentially supply essential nutrients, but it can also lead to exposure to heavy metals, organic chemicals or pathogens. In large quantities, it can cause an intestinal obstruction (Steffan et al., 2018). It is commonly believed that direct ingestion is the most important pathway for human exposure to soil contamination (Science Communication Unit, University of the West of England, 2013). The exposure pathway through incidental ingestion of soil and dust is particularly important for children (< 3 years of age) because their behavioural patterns expose them to a higher rate of ingestion compared to adults (Juhasz et al., 2011; Landrigan et al., 2018; Qu et al., 2019).

Respiration involves inhaling soil-derived dust and volatile compounds, but it is a less significant source of human exposure compared to ingestion (Hosford, 2009; Steffan et al., 2018). Inhaling BTEX-polluted air is the greatest hazard to humans by these compounds (Annex II). BTEX chemicals are water-soluble compounds, and thus improper handling can also cause groundwater contamination (Valentin et al., 2013; Godambe and Fulekar, 2017).

Table 2.2: Overview of the possible exposure pathways for contaminated sites (Swartjes and Cornelis, 2011).

	Indoor exposure		**Outdoor exposure**	
	Pathway	***Contact medium***	***Pathway***	***Contact medium***
Oral	Dust digestion Drinking water consumption Meat consumption Milk consumption Egg consumption	Dust Groundwater/drinking water Meat Milk Eggs	Soil ingestion Vegetable consumption	Soil vegetables
Inhalation	Air inhalation Airborne dust inhalation Water vapour inhalation during showering	Indoor air Indoor airborne dust Drinking water	Air inhalation Airborne dust inhalation	Outdoor air Outdoor airborne dust
Dermal	Dust contact Water contact during bathing	Dust Drinking water	Soil contact	Soil

Dermal contact via skin absorption or penetration can expose an individual to soil chemicals and pathogens. Absorption through the skin is important in the case of volatile organic compounds (Science Communication Unit, University of the West of England, 2013; Steffan et al., 2018).

[11] Direct ingestion of soil is not a common dietary practice and is considered to be a pathological psychological behaviour, called pica, that involves consumption of non-food materials.

Humans may also be affected as a result of secondary contamination of water supplies and from deposition of air contaminants (Rodríguez-Eugenio et al., 2018). Soil contaminants may enter the human food chain through consumption of plants and animals that have accumulated large amounts of soil pollutants. Some of the effects may be significant, as in the case of dioxins that accumulate up the food chain, or large quantities of cadmium (Cd) in crops grown in contaminated sites (Science Communication Unit, University of the West of England, 2013; Augustsson et al., 2018).

2.2.2 Overview of health effects and factors influencing these effects

Hazard identification in a risk assessment is the process to identify the specific chemical hazard(s) and determine whether exposure to this chemical has the potential to harm human health. The potential hazard(s) of the chemical can be determined from the available scientific data on the chemical, generally data from toxicological or epidemiological studies. A chemical may be associated with one or more hazards to human health. In general, chemicals are classified according to human health hazards that they pose, such as neurological, developmental, reproductive, respiratory, cardiovascular and carcinogenic effects.

Potential health effects

A detailed review of health effects associated with different types of pollutants is provided in Table 2.3, Table A2 (Annex II) and Table A4 (Annex III). Health effects caused by contaminants may be divided into local (the effect on specific organs at the place of contact or intake) and systemic (effects in the whole body after systemic circulation and absorption and distribution in the human body). Soil pollutants can impact various organs (e.g., lungs, skin, gut, liver and kidneys) or systems (e.g. immune, reproductive, nervous and cardiovascular), see Table 2.3 and Table A4 (Annex III).

According to the WHO (Science Communication Unit, University of the West of England, 2013), 10 pollutants are the cause for major public concern, of which eight can also occur in soil: arsenic (As), Cd, lead (Pb), mercury (Hg), fluoride, dioxins, hazardous pesticides and asbestos. They are known for their human health effects, like carcinogenicity (As, asbestos and dioxins), neurological defects and lower IQ effects (As and Pb), kidney disease (Pb, Hg and Cd) and skeletal and bone diseases (Pb and fluoride; Science Communication Unit, University of the West of England, 2013). General characteristics of the main pollutants listed by the WHO are presented in Annex II, Table A1.

According to the Stockholm Convention[12], the presence of persistent organic pollutants (POP) in the soil leads to serious health effects. POP compounds like organochlorine pesticides, e.g., DDT and DDE, PCBs, dioxins and PAH, can accumulate in human tissues. Even very low concentrations in the environment can cause adverse health effects. Long-term exposure to PCBs and dioxins may result in an elevated risk for infectious diseases in children (Valentin et al., 2013; Rodgers et al., 2018). The development of cancers, including breast cancer, leukaemia and thyroid cancer, was observed in humans exposed to elevated concentrations of DDT/DDE and dioxins (Bornman et al., 2018; Rodgers et al., 2018). Exposure to these compounds may result in immunotoxicity, reproductive diseases and neurotoxicity (see Table A2, Annex II, for details).

[12] The Stockholm Convention on Persistent Organic Pollutants (http://chm.pops.int/TheConvention/Overview/TextoftheConvention/tabid/2232/Default.aspx).

Additionally, some PAH are toxic compounds towards soil organisms and carcinogenic to humans (Abdel-Shafy and Mansour, 2016), and 16 PAH compounds are on the US Environmental Protection Agency (EPA) priority pollutant list (Zelinkova and Wenzl, 2015).

Some contaminants may be carcinogenic, mutagenic or teratogenic (Rodgers et al., 2018). Mutagenic contaminants change the DNA in the cell cores, cause cancer or result in miscarriages due to gross chromosomal changes. A common mutagenic contaminant frequently found in soil and groundwater is benzene. Lead is an example of a teratogenic contaminant, which causes birth defects, that is frequently found in soils (Table A1, Annex II).

The International Agency for Research on Cancers (IARC) classifies chemicals into five categories[13] based on the strength of evidence that an agent could alter the age-specific incidence of cancer in humans. An example of compounds included in the first group of carcinogens for which there is sufficient evidence of a carcinogenic effect on humans is provided in Table 2.3.

Table 2.3: Some IARC "Group 1" carcinogens, with geochemical exposure routes and their mechanisms of carcinogenesis (according to Middleton et al., 2019).

IARC Group I carcinogens	Cancer sites*	Geochemical exposure routes	Carcinogenic mechanisms
As and inorganic As compounds	<u>Lung, skin, urinary bladder,</u> kidney, liver and bile duct, prostate	Groundwater consumption, soil ingestion	Alter DNA repair/cause genome instability Induce epigenetic alterations Induce oxidative stress Modulate receptor mediated effects
Asbestos (all forms)	<u>Larynx, lung, mesothelium (pleura ad peritoneum), ovary</u>, colon and rectum, pharynx, stomach	Soil inhalation	Induce oxidative stress Induce chronic inflammation
Radon-222 and its decay products	<u>Lung</u>, leukaemia and lymphoma	Indoor air exposure (bedrock derived)	Genotoxic Induce oxidative stress Alter cell division, death or nutrient supply
Cd and Cd compounds	<u>Lung</u>, kidney, prostate	Soil ingestion, groundwater consumption	Alter DNA repair/cause genome instability Induce epigenetic alterations Induce oxidative stress

*Sites underlined – sufficient evidence; in regular font – sites with limited evidence

[13] IARC categorisation of chemicals regarding their carcinogenicity: Group 1 – carcinogenic to humans; Group 2A – probably carcinogenic to humans; Group 2B – possibly carcinogenic to humans; Group 3 – not classifiable as to its carcinogenicity to humans; Group 4 – probably not carcinogenic to humans.

Beyond the physical effects, researchers have reported the impact of soil contamination on mental health (van Wezel et al., 2008). These authors conducted a study with cost benefit analysis for soil remediation in the Netherlands and reported that the most disturbing health effects are stress related to people's perception of the risk of living on or close to a contaminated site as well as the fear of value loss of their property.

EP, released to the environment from many anthropogenic sources, are spread to the different environmental compartments (water, air, soil and organisms). Due to their persistence, bioaccumulative nature and toxicity, they can affect human health (Figure 2.3; Lei et al., 2015; Naidu et al., 2016; Snow et al., 2017).

Environmental or public health risks from EP and their interaction and toxicological impacts on receptors should be better studied. Specifically, there is a huge knowledge gap about the effects of chronic exposure to low EP levels (Lei et al., 2015). The main adverse human health effects caused by different emerging contaminants are presented in Annex III, Table A4.

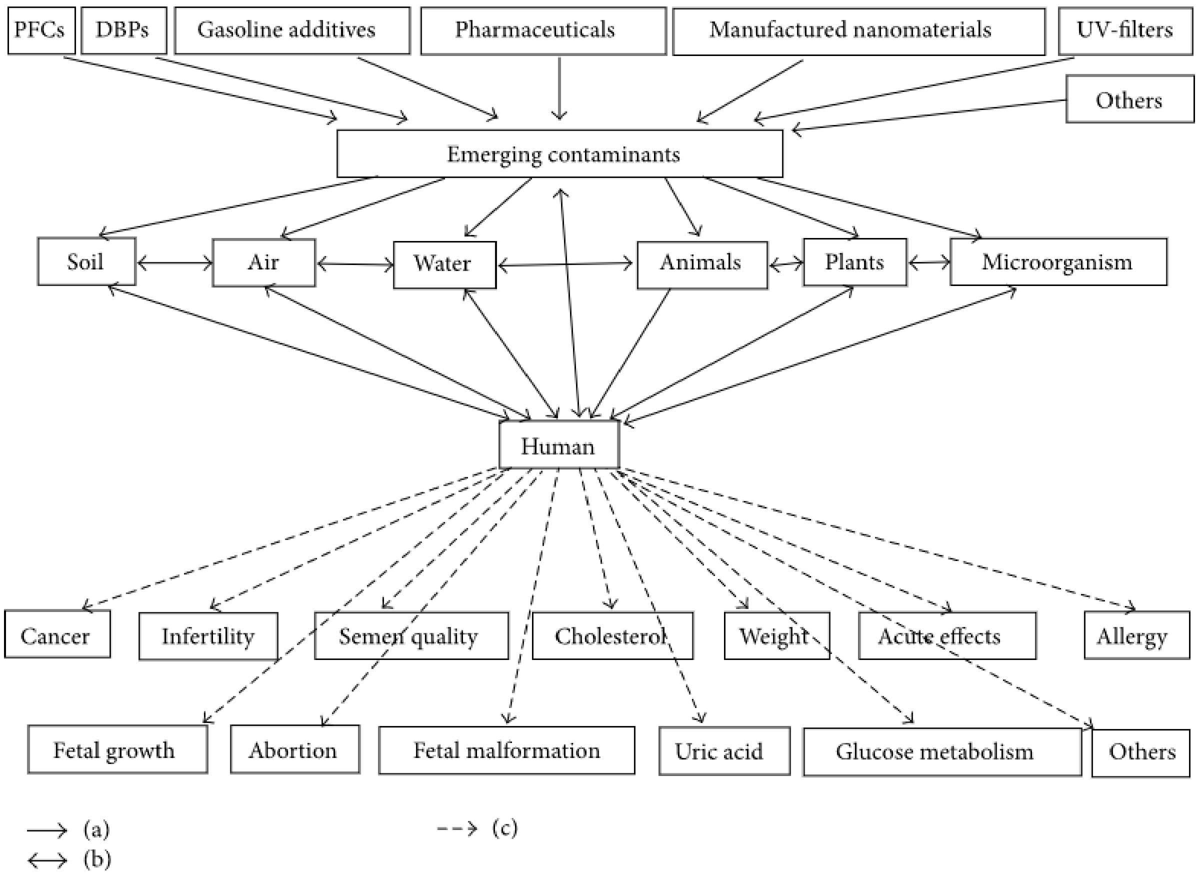

Figure 2.3: (a) Different groups of emerging contaminants; (b) interactions between soil, air, water, animals, plants, microorganisms, humans and emerging contaminants; (c) human exposure to emerging contaminants may have potential adverse effects (according to Lei et al., 2015).

Endocrine disrupting compounds (EDC), for example, bisphenol A, affect normal functioning of the endocrine system and can lead to diabetes, problems with the cardiovascular system and even obesity, among other complications (Valentin et al., 2013; WHO, 2013b; Naidu et al., 2016; Siddique et al., 2016). EDC are identified under REACH[14] as substances of very high concern. Moreover, substances that are persistent, bioaccumulative and toxic or very persistent and very bioaccumulative are covered with restrictions. These compounds do not degrade near emission sources but may be gradually transported into remote areas, have a high potential to accumulate in biota and their long-term effects for human health are rarely predictable. PFAS compounds may, due to their persistence and bioaccumulative nature, result in changes in the endocrine system, metabolic diseases, reproductive dysfunction and different types of cancer (Lei et al., 2015; Concawe, 2016; Winkens et al., 2017). Widespread use of antibiotics may contribute to increased antimicrobial resistance (Lei et al., 2015; Hashmi et al., 2017; Grenni et al., 2018).

Factors that influence effects

The health impacts of contaminants are influenced by several factors: exposure time, the exposure level of the hazardous substances, the soil properties that determine chemical bioavailability, the type and properties of the contaminant and the efficiency of the human detoxification system (Science Communication Unit, University of the West of England, 2013; Carre et al., 2015). The health impact of soil pollution also depends on the vulnerability of people within communities. The most susceptible groups are the elderly, children and pregnant women (Landrigan et al., 2018). The health effect also relies on substances' potency for dispersion, solubility in water or fat, bioavailability and carcinogenicity. Some chemicals that cause severe health damage at high doses may be innocuous or even essential at low doses. Selenium (Se) and zinc (Zn) are examples of geochemically derived micronutrients that play a crucial role in antioxidant mechanisms and immunity (Steinnes, 2009; Middleton et al., 2019).

The main soil properties that affect leaching, transport and bioavailability of inorganic contaminants in soil are: pH, cation exchange capacity, type and content of clay, organic matter content, redox potential, iron (Fe)/Mn oxides and diversity of soil microorganisms (Cave et al., 2011; Biswas et al., 2018). Fate and transport of organic compounds is regulated by soil properties (clay and organic matter content), soil moisture and temperature, soil biological activity and the physicochemical properties of these chemicals. Potential absorption of organic compounds in the food chain depends on a distribution coefficient, volatility, water solubility, persistence and half-life and bioconcentration factor (Carre et al., 2015).

[14] Regulation (EC) No 1907/2006 of the European Parliament and of the Council of 18 December 2006 concerning the Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH), establishing a European Chemicals Agency, amending Directive 1999/45/EC and repealing Council Regulation (EEC) No 793/93 and Commission Regulation (EC) No 1488/94 as well as Council Directive 76/769/EEC and Commission Directives 91/155/EEC, 93/67/EEC, 93/105/EC and 2000/21/EC (available: https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:02006R1907-20140410&from=EN)

A few defence mechanisms protect the human body against harmful effects of pollution: physical barriers, enzymes and vitamins that neutralise toxic contaminants and the ability to remove contaminants through the liver and kidneys. The efficiency of the human detoxification system is dependent on genetic features, life style, dietary habits and age (Langley, 2011).

Until recently, most risk assessment models assumed that the target chemical is 100% available. Therefore, soil pollution was often assessed by the total concentrations rather than the bioavailable fraction, an assumption that may overestimate the health risk and increase remediation costs (Li et al., 2018). This potential has generated an increasing interest in the incorporation of bioavailability and bioaccessibility methods and measures into both ecological and human risk assessment procedures (Juhasz et al., 2011; Rostami and Juhasz 2011; Li et al., 2019).

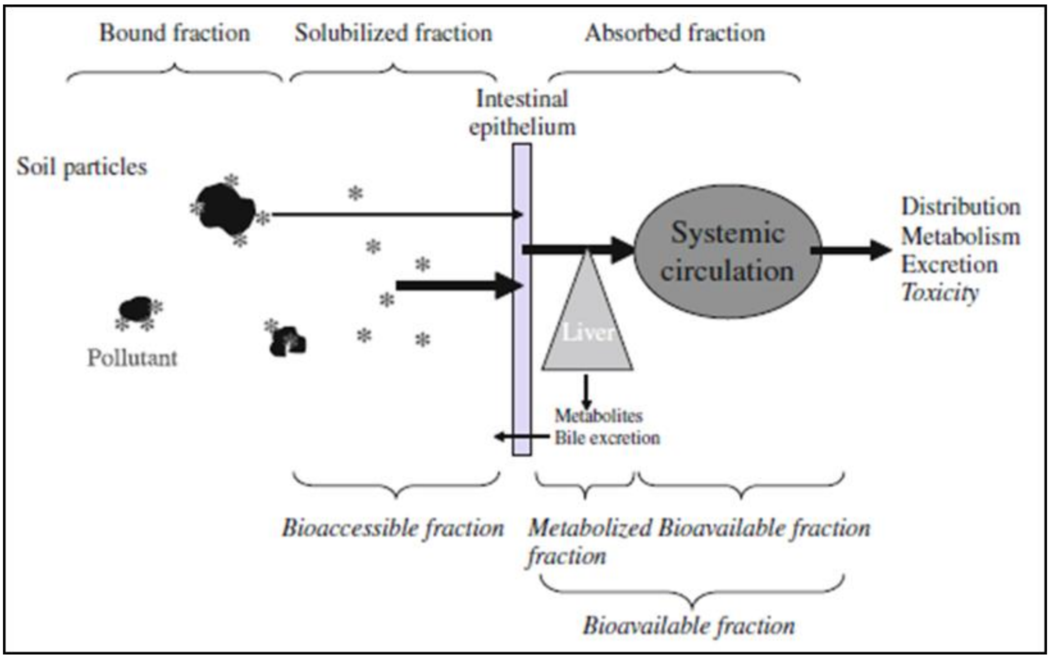

Figure 2.4: Steps involved in oral bioavailability (Cave et al., 2011).

In terms of human health risk assessment, bioavailability may be simply defined as the fraction of a dose (obtained via ingestion, inhalation or dermal pathways) that reaches systemic circulation (Figure 2.4), where it may then cause adverse health effects (Cave et al., 2011; Rostami and Juhasz, 2011). Bioaccessibility is the fraction of a compound that is soluble in gastrointestinal tract (Figure 2.4) and is therefore available for absorption (Hosford, 2009; Science Communication Unit, University of the West of England, 2013). Estimates of bioaccessibility and overall bioavailability (i.e., bioaccessibility plus absorption) can be determined from experimental studies: *in vitro* systems that mimic biological conditions for bioaccessibility estimates and *in vivo* models for bioavailability (Naidu et al., 2008; Li et al., 2019). Bioaccessibility and absorption affect the bioavailability of all chemicals from the environment but are of special importance for metals, which can exist in a variety of chemical and physical forms. Different forms of a given metal can be absorbed to a different extent.

To study human bioaccessibility of priority contaminants (e.g., As, Pb and Cd) via the gastrointestinal tract, the Bioaccessibility Research Group of Europe (BARGE[15]) was launched. The activities of this network include development, validation and harmonisation of bioaccessibility methodology and provide robust and defensible data on bioaccessibility that can be used in human health risk assessment and policy making.

2.3 Measures to reduce health risk

Reducing the health risk related to polluted soil can be achieved by measures and management that rely on limiting inputs of contaminants to soil or contamination isolation and cleaning up contaminated sites. Prevention strategies include reduction in emissions of pollutants, transition to non-polluting, renewable sources of energy, adoption of non-polluting technologies for production and transportation and limiting the application and better control of fertilisers, pesticides and herbicides in agriculture (Landrigan et al., 2018). To prevent excessive accumulation of contaminants and transmission in the food chain, introducing alternative crops (e.g., cropping industrial plants or energy biomass), as well as alternative land uses (e.g., afforestation), are recommended. Regulation is an essential tool; both the polluter-pays principle and an end to subsidies and tax breaks for polluting industries must be integral components of pollution control programmes (Landrigan et al., 2018).

Soil remediation in Europe is still a huge challenge. The first step to take effective action is identification of the sites and their risk assessment. Currently, approximately 3 million places are estimated to be potentially polluted. During the last three decades, remediation has been performed on about 80,000 sites (EEA, 2017)[16]. Numerous approaches and methods of soil remediation have been applied in European countries (Pérez et al., 2015; Sarwar et al., 2017; Khan et al., 2018). Examples of the methods utilised for remediation of soils contaminated with heavy metals are presented in Table 2.4.

Table. 2.4: Heavy metal remediation methods (Liu et al., 2018).

Ex situ	Physical	Landfilling
	Chemical	Soil washing
		Solidification
	Thermal	Vitrification
In situ	Physical	Surface capping
		Encapsulation
	Electrical	Electrokinetics
		Vitrification
	Chemical	Soil flushing
		Immobilisation
	Biological	Phytoremediation
		Bioremediation

[15] Available at: https://www.bgs.ac.uk/barge/home.html
[16] EEA, 2017: https://www.eea.europa.eu/themes/soil/soil-threats

In practice in Europe, the most commonly used remediation method (approximately 30% of cases) is "dig and dump" (EEA, 2019)[17], based on the excavation of the soil from the contaminated site, transport and deposition on the hazardous waste landfill (Khan et al., 2018).

Soil contamination can be remediated *ex situ* (soil removal and subsequent treatment by using physical, chemical, electrical, thermal or biological methods) or *in situ* (soil treatment on the contaminated site). Conventional remediation techniques include *in situ* vitrification, soil incineration, excavation and landfilling, soil washing, soil flushing, solidification and stabilisation of electrokinetic systems. Generally, the physical and chemical methods suffer from limitations like high costs, intensive labour, irreversible changes in soil properties and disturbance of native soil microflora. Chemical methods can also create secondary pollution problems. Therefore, only cost effective, efficient and environmentally friendly gentle remediation techniques are recommended.

Gentle remediation encompasses a number of technologies, including the use of plants and associated soil microbes (like fungi and bacteria) for reducing exposure of local receptors to contaminants by *in situ* stabilisation, extraction, degradation or transformation of contaminants. Gentle remediation comprises such methods as *in situ* immobilisation, phytovolatilisation, phytostabilisation, rhizofiltration, rhizodegradation, phytodegradation and phytoextraction (Sarwar et al., 2017; Khan et al., 2018; Liu et al., 2018). *In situ* methods are in general cost effective and can be used at a large scale (Pérez and Rodríguez-Eugenio, 2018). They are also less invasive and safer for the workers (Olexsey and Parker, 2006). However, *in situ* techniques are time consuming (Pérez and Rodríguez-Eugenio, 2018), and their application is bound to additional conditions, e.g., depth of the contamination and potential deep leaching of chemicals (Olexsey and Parker, 2006). Many gentle remediation options do not clean the soil; rather, they only reduce risks related to the contamination. In some cases, it is possible to deal with the risks without treating the soil (e.g., using fencing to prevent access to a site). Sustainable remediation is a term that describes actions that eliminate unacceptable risks in a safe and timely manner and that maximise the overall environmental, social and economic benefits of the remediation (SuRF-UK).

[17] EEA, 2019: https://www.eea.europa.eu/data-and-maps/indicators/progress-in-management-of-contaminated-sites-3/assessment/view

3 Impact of soil sealing on human health

Soil sealing is a process of covering soils by buildings, constructions and layers of completely or partly impermeable artificial material (asphalt, concrete, etc.). It is a very intense form of land degradation that is rarely subjected to restoration or de-sealing. Sealing can be considered to be a land take process, which is an increase in artificial surfaces, usually at the expense of agricultural or natural areas. The term "artificial surface" is used in the CORINE Land Cover nomenclature and refers to "continuous and discontinuous urban fabric (housing areas), industrial, commercial and transport units, road and rail networks, dump sites and extraction sites, but also green urban areas" (Prokop et al., 2011).

As indicated by Zhang (2016), rapid urbanisation has been observed in many parts of the world. In 1800, only 2% of the world's population lived in urban areas, in 1900 it was 15%, whereas the pace of urban population growth accelerated very rapidly after the 1950s. In 1950, more than 70% of people worldwide lived in rural settlements. In 2014, more than half of the world's population lived in urban areas. This number is expected to increase from 3.6 billion to 6.3 billion by 2050.

There is an intensive and often uncontrolled increase in artificial surfaces. According to the CORINE land cover spatial databases, an 8.8% increase of artificial surface in the EU was recorded between 1990 and 2006. In the same period, the population increased by only 5% (Prokop et al., 2011). Between 2006 and 2012, the annual land take in the European countries (EEA-39) was approximately 107,000 ha/year. The conversion of land into artificial areas in the EU has been accelerating over the years; the growth from 2012 to 2015 was approximately 6% higher than from 2009 to 2012. In the EU, artificial surfaces are on average sealed by 51%, but this share varies strongly among Member States, depending on dominant settlement structures and the intensity of the interpretation of artificial surfaces (Prokop et al., 2011).

Soil sealing is driven by demand for new housing, business locations and road infrastructure related to economic development of cities. These processes are influenced by spatial planning practices, but the importance of soils and their role in the urban environment and quality of life still receives limited attention. Most intensive sealing occurs in urban areas. Therefore, it can be assumed that the impact of soil sealing on human health is the highest in cities, especially in developed countries, where approximately 75% of the population lives in urban environments.

3.1 Methodologies applied for evaluating the impact of soil sealing on human health

There is a lack of methods that evaluate the direct relationship between soil sealing and human health. However, surveys, participatory or data-driven types of evaluations might lead to new insights on the impact of sealed or artificial surface density on human health.

For example, in their recent study, Cox et al. (2018) collected data from 3,000 survey respondents from across the UK. They used a nature-dose framework to determine whether (a) increasing urbanisation is associated with a decrease in the frequency, duration and intensity of the nature dose and (b) differences in nature exposure associated with urbanisation affect four population health outcomes (depression, self-reported health, social cohesion and physical activity). Respondents were requested to provide a full UK postcode so that their neighbourhood was known with sufficient spatial resolution (one UK postcode covers approximately 20 households).

The study revealed that people in urban areas had reduced exposure to nature across all three dimensions of the nature dose compared to their rural counterparts. The people with reduced access to nature also tended to have worse health across multiple domains.

Participatory assessments help to semi-quantify the general impact of soil sealing on human health at the level of a city. The approach, applied within the Urban SMS project, involved participation of local stakeholders, who were led through steps of impact assessment in order to collect their opinions on possible urbanisation impacts (Morris et al., 2011). Three scenarios, which represent different soil protection approaches, were assessed for their impacts on the soil functions in the long-term, including the health provision function (Siebielec et al., 2011). The baseline scenario assumed that nothing would change in regulations concerning soil protection. The stakeholders' opinions led to the conclusion that only much stronger soil protection during urbanisation would reduce its negative impacts on human health.

There have been studies reported in the literature linking the incidence of disease or deaths with heat extremes or air pollution, especially during summer hot periods in densely urbanised areas (Piver et al., 1999; Fischer et al., 2004; Filleul et al., 2006; Ho et al., 2017; Depietri et al., 2012).

3.2 Pathways of exposure and possible health effects

While the impacts of soil sealing on humans are not exclusive to urban areas, they are most pronounced in these locations. The population density is highest and multiple pathways exist for human exposure to effects of dense soil sealing. Uncontrolled urbanisation and related soil degradation likely strongly affect human health because 72.5% of the population in Europe (535 million) lives in urban areas. Improper land management exhibited by unsustainable soil sealing might affect humans through urban heat islands, contaminant toxicity, air pollution (dust, particulate matter [PM]), disconnection of citizens from green areas, limited recreation opportunities, impact on mental health, the risk of flooding, etc. However, there is very limited understanding of the relationships between soil management in urban areas and human health or life quality.

Sealed surfaces have higher surface temperatures compared to green surfaces and alter the micro-climate, particularly in highly sealed urban areas (EEA, 2010). Recent surface temperature surveys in Budapest (Hungary) and Zaragoza (Spain) revealed that temperatures in highly sealed areas can be up to 20°C higher compared to green shaded surfaces (Prokop et al., 2011).

An interaction between heat waves and ozone concentration reportedly increases the risk of mortality during summer heat extremes in densely sealed urban areas. For example, in France during August 2003, when record high temperatures were observed across Europe, the excess risk of deaths linked to ozone and the extreme temperatures ranged from 10.6% in Le Havre to 174.7% in Paris. During the same period, a significant portion of excess mortality was attributed to PM_{10} and ozone in the air (Filleul et al., 2006).

In the Netherlands, 1,000–1,400 deaths were attributed to the hot temperatures that occurred during the 2003 summer period, including 400–600 deaths attributable to the ozone and PM_{10} concentrations from June to August (Fischer et al., 2004).

According to the Dutch Central Bureau of Statistics (CBS), almost 400 extra deaths compared to an average week in the summer period were reported during the heat wave in week 30 of 2019[18].

As reported by Ho et al. (2017), excess mortality due to prolonged heat has been widely observed in an urban environment, including cities in tropical and subtropical areas and temperate regions. The heat during such extreme hot weather is worsened by the compact urban settings. They also found that consecutive hot nights contribute more significantly to higher mortality risk than a number of consecutive hot days.

Air quality can cause illness during heat waves. Hot days are often followed by hot nights because of the heat island effect. These conditions can produce a combination of heat and air pollution stress, especially for people with cardiovascular and respiratory disorders (Piver et al., 1999; Depietri et al., 2012).

Soil sealing patterns affect air quality. Poorly planned patterns might increase traffic or travel time and the connected emissions of contaminants and gases to the air. Furthermore, deterioration of air quality in densely sealed areas is probably also linked to reduced soil function as sink and diluter for pollutants, accelerated wind erosion and loss of green areas adsorbing pollutants and dusts from the air.

Loss of water retention areas, concomitant with increased surface water runoff, leads to additional flood risk and in some cases to catastrophic floods (Siebielec et al., 2015).

The density of soil sealing affects human health through reduced access to green areas. Cox et al. (2018) reported, based on an extensive survey study in the UK, negative exponential relationships between nature dose and the degree of urbanisation. There were weak but positive associations between frequency and duration of dose across all four studied population health outcomes (depression, self-reported health, social cohesion and physical activity). The study showed that people in urban areas with a low nature dose tend to have worse health across multiple domains, but they have the potential for the greatest gains from spending more time in green areas. There was also evidence that citizens with a greater orientation to nature have better mental health, social cohesion and physical behaviour. Therefore, access to green areas in the urban environment is key to connect to nature. Cox et al. (2018) also suggested that decreasing experiences with nature, often associated with an urbanising population, can result in a reduced knowledge of, and support for, environmental issues, and hence further worsened living conditions in cities.

The DG for Regional and Urban Policy of the European Commission in its report “Cities of tomorrow” raised that urban sprawl hampers the efficient organisation of services: health care for the elderly, primary and secondary education for the youngest, etc. This problem increases the risk of social isolation. Sprawl often takes place outside local administrative areas, and thus public services may not coincide with the territorial distribution of its users (European Commission, 2011).

[18] https://www.cbs.nl/nl-nl/nieuws/2019/32/hogere-sterfte-tijdens-recente-hittegolf

The density and compactness of cities reduce energy needs for heating and mobility. However, the density of the sealed surface raises important questions about the environment's capacity to accommodate the concentration of waste and pollution. Natural ecocycles, especially those for water, are being disrupted by a lack of natural soil and wetlands (European Commission, 2011). The report also draws attention to noise pollution, amplified by the concentration of activities, in particular transport, and the use of hard, sound-reflecting materials, as a cause for health problems.

3.3 Measures to reduce health risk

Measures to prevent excessive soil sealing and its consequences can be grouped into limitation, compensation and mitigation categories, as presented in the EC guidelines on best practice to limit, mitigate or compensate soil sealing (European Commission, 2012). The guidelines provide a range of good practice examples among European cities, including setting land take targets, soil quality in urban planning, regulations against conversion of agricultural land, brownfield regeneration and others. It also provides recommendations for tackling the problem of soil sealing. The greatest potential to reduce health risks related to soil sealing is in improved spatial planning (Maring, 2019). This activity refers to both limiting soil sealing, its spatial pattern and density and connectivity or access of inhabitants to green areas. Challenges cannot be addressed individually; their interrelations and trade-offs need to be properly understood and appreciated in spatial planning. Therefore, urban spatial planning must treat a city as an organism where all issues, namely transport, air quality, temperatures, green areas and recreation, are balanced. Grunewald et al. (2018) reviewed the effects of better exposure of humans to green areas, e.g.:

- positively affects mental health by reducing stress and mental distress, rates of anxiety and depression;
- raises positive emotions;
- improves cognitive functioning and mood;
- improves physical health and longevity;
- improves life satisfaction in general;
- improves air quality, including through removing ozone and storing carbon dioxide,
- buffers anthropogenic noise;
- decreases the urban heat island effect;
- improves the immune system through microbial input from the environment to drive immunoregulation;
- fosters social cohesion.

Spatial planning that involves soil quality (e.g., texture and contamination) would enable further reduction of exposure through the selection of soils with lower capacity to fulfil various soil functions as a location for new constructions.

Soil cover improves water storage in the urban landscape. Urban environments, due to the high contribution of built-up and sealed areas, are particularly susceptible to temperature extremes. The potential of soil types to mitigate heat waves depends greatly on the soil profile texture and structure. Therefore, soil sealing—at the expense of high-water storage soils—has consequences for the local climate and the risk of flooding. The quality of soil cover left in urban and peri-urban areas also influences health risks through controlling wind erosion. Wind erosion is generating local sand and dust storms, particularly on dry soils covered by scarce vegetation.

Sustainable spatial planning has the greatest potential for new expansions of cities. In existing densely sealed districts, the available measures are of a different nature. Brownfield regeneration primarily limits conversion of agricultural land into newly constructed land, but it also solves problems related to the unused brownfield areas, which are often contaminated and susceptible to erosion.

Mitigating soil sealing effects may involve the use of permeable materials in construction work, in order to protect soil retention and buffering functions and provide at least partly a cooling effect of soil (European Commission, 2012). Permeable materials enable water evaporation, which is important for avoiding the heat island effect, while also decreasing the cost of water treatment and reducing the risk of flooding.

Proper management of green areas that already exist in cities is crucial for maintaining air quality and temperatures at acceptable level.

4 Impact of erosion processes on human health

Soil erosion can be defined in general as a three-phase process: the detachment of soil particles from the land surface, their subsequent transport and deposition. The main erosive agents are water and wind (Stolte et al., 2016). Soil erosion may affect water runoff, soil water holding capacity, soil organic matter, nutrients, soil depth and soil biota. The loss of soils from land surfaces by erosion diminishes soil quality and reduces the productivity of natural, agricultural and forest ecosystems. Moreover, the diversity of plants, animals and microorganisms in the soil is adversely affected (Pimentel, 2006; Sterk and Goossens, 2007; Pimentel and Burgess, 2013; Stolte et al., 2016). Soil erosion changes the physical, chemical and biological characteristics, all of which result decrease agricultural productivity and may raise concerns in the context of food security. Soil erosion by water accounts for the greatest loss of soil in Europe compared to other erosion processes (e.g., wind erosion). Land degradation due to wind erosion locally affects the Mediterranean region, as well as the temperate climate areas of the northern European countries: Northern Germany, Eastern Netherlands and Eastern England (Stolte et al., 2016). Wind erosion is the main cause of sand-dust storms in drylands and downwind regions. These storms adversely influence human well-being, such as air quality, transportation safety and human health.

4.1 Methodologies applied for evaluating the impact of erosion on human health

The studies about the links between soil erosion by wind and water and direct human health effects are rather scarce. Given the lacking information on the impacts of water erosion on human health, this chapter focuses on wind erosion. In recent years, more attention has been focused on the impact of long-range transport of soil particles (e.g., desert dust) on human health (Griffin et al., 2001; Jiménez et al., 2010; Grineski et al., 2011; Zhang et al., 2016; Diaz et al., 2017; Middleton, 2017). The most frequently reported effects in the scientific literature were caused by Sahara dust episodes. They were identified by the following methods: back-trajectory analysis using NRL,[19] SKIRON[20] dust maps and satellite images provided by the National Aeronautics and Space Administration (NASA) and the combination of Lidar[21] observations with operational models and the changes of the ratio PM_{10} (see section 4.2) to NO_2 (Karanasiou et al., 2012; Stafoggia et al. 2016). The available data are based mainly on epidemiological studies and less frequently on toxicological investigations (Karanasiou et al., 2012; Stafoggia et al. 2016; Zhang et al., 2016). To assess the links between dust outbreaks and human mortality or morbidity, regression models or case-crossover designs were used in the epidemiological studies (Perez et al., 2008; Grineski et al., 2011). Human exposure on a case day (when mortality or morbidity occurs) was compared with exposure on control days on which the outcome does not occur. The epidemiological data also included a time series analysis on daily mortality and morbidity (Jiménez et al., 2010; Diaz et al., 2017).

[19] NRL – the Naval Research Laboratory (available at: https://www.nrlmry.navy.mil/aerosol/). NRL has developed a multi-component aerosol analysis and modelling capability system, which provides aerosol and dust maps.
[20] SKIRON – the weather prediction model developed by the Atmospheric Modelling and Weather Forecasting Group of the University of Athens (available at: http://forecast.uoa.gr/).
[21] Lidar –light detection and ranging, an active remote sensing technique that measures distance to a target by illuminating the target with a pulsed laser and measuring the reflected pulses with a sensor.

4.2 Pathways of exposure and possible health effects

All wind-erosion-prone areas potentially produce dust, but the quantity depends on the soil texture. Soil-derived (mineral) dust particles can be a major part of the atmospheric particulate matter (PM) concentrations, both fine (smaller than 2.5 µm – $PM_{2.5}$) and coarse (< 10 µm – PM_{10}). The health effects of dust generated from soil sources have raised interest recently. These effects could be either direct (aspiration of dust via respiratory system) or indirect (exposure to dust pathogens or pollutants; Table 4.1).

Dust particles are transported via air, inhaled through the nose or mouth and passed via the trachea to the lung tissues and finally cause respiratory or cardiopulmonary diseases (Schenker, 2000; Sterk and Goossens, 2007; Ginoux et al., 2012; Zhang et al., 2016). Inhalation of mineral dust may lead to an interstitial lung disease called silicosis (also called desert lung syndrome). The main symptoms include shortness of breath, fever and fatigue, and in some cases may lead to lung cancer (Morman and Plumlee, 2013; Goudie, 2014). Mineral dust exposure can cause chronic bronchitis, pulmonary fibrosis, interstitial fibrosis—depending on the exposure levels—and pulmonary tuberculosis. Dust can also lead to chemical and pathogen exposure that negatively impact human health (Karanasiou et al., 2012). Cessna et al. (2006) and Bento et al. (2017) revealed the high risk of the transport of herbicides (e.g., trifluralin and glyphosate) during wind erosion events in agricultural regions to off-target areas. The highest content of these herbicides was found in the smallest soil fractions (PM_{10} and less), which are easily inhaled and contribute to human exposure.

Table 4.1: Influence of soil erosion on human health (based on Goudie, 2014; Stafoggia et al. 2016; Zhang et al., 2016).

	Health effects via contact with dust	
Direct effects	Mortality	due to respiratory and cardiovascular diseases
	Respiratory disorders	asthma, tracheitis, pneumonia, chronic obstructive pulmonary disease, allergic rhinitis, silicosis
	Cardiovascular disorders	stroke, arrhythmia, ischaemic heart disease, cerebrovascular disease
	Dermatological disorders	skin allergy, reddish skin, skin irritation and itching
	Conjunctivitis	
	Reproductive disorders	
Indirect effects	Infectious diseases	bacterial meningitis, coccidioidomycosis
	Exposure to chemicals	herbicides, pesticides, heavy metals, dioxins, radioactive isotopes
	Deterioration of drinking water quality	
	Economic damages	muddy flooding of homes, villages and infrastructure

There is a limited range of data on the links between soil erosion (both water and wind) and human health effects at the European level. The transport of Saharan dust in southern Europe is an example of a transboundary impact of soil degradation on human health (Perez et al., 2008; Jimenez et al., 2010; Stafoggia et al. 2016; Diaz et al., 2017). The Sahara is one of the dominant sources of dust globally; approximately 12% of Saharan dust moves northwards to Europe (Karanasiou et al., 2012).

Saharan dust outbreaks may adversely impact the human health due to the high level of particulate matter and the transport of anthropogenic pollutants and possible transport of microorganisms. Epidemiological studies from Italy and Spain found evidence of increased respiratory mortality among the elderly exposed to Saharan dust events. Jimenez et al. (2010) and Staffogia et al. (2016) found that PM_{10} concentrations significantly correlated with the daily mortality and hospital admissions the residents of 13 Southern European cities (e.g., Madrid, Barcelona, Turin, Rome, Athens) on days with Saharan dust, while there was no statistical effect attributable to PM_{10} exposure on non-Saharan dust days. In contrast, Perez et al. (2008) did not observed an increased daily mortality of Barcelona residents due to $PM_{2.5}$ during Saharan dust episodes.

Desert dust was associated with increased odds of hospitalisation for asthma and bronchitis, mainly in children (Grineski et al., 2011). Various fungal diseases, including coccidioidomycosis and aspergillosis, may be a result of exposure to dust storms (Ginoux et al., 2012).

The health effects caused by soil materials like dust are related to distinct factors: intensity and the time of exposure, physicochemical properties of the dust, presence of the pathogens and toxicants in the dust, the bioaccessibility, biodurability and bioreactivity of dust in body fluids, the physiological processes in the human body that control absorption, distribution, excretion of toxicants and confounding factors (Morman and Plumlee, 2013). A number of various confounding factors can affect the results of the epidemiological studies (Karanasiou et al., 2012). The most frequently reported are: temperature, humidity, seasonality, flu epidemic weeks, heat waves and ambient pollutants (chemical, biotic, acoustic; Jimenez et al., 2010; Diaz et al., 2017; Middleton and Kang, 2017). Susceptibility of the human population to short-term effects of suspended particulates is also associated with age and the coexistence of other diseases. Jimenez et al. (2010) reported that the most sensitive group of residents are the elderly (due to their lower immunological capacity and deterioration in their general health due to the ageing process) and people affected by chronic cardiopulmonary disorders.

4.3 Measures to reduce health risk

Soil conservation techniques to prevent soil erosion, and thus reduce the impact on environmental and human health, include the use of biomass mulches, crop rotations, no-tillage, ridge-tillage, added grass strips, shelterbelts and contour row-crop planting (Table 4.2; Pimentel, 2006).

Table 4.2: Types of measures to reduce soil wind erosion (according to Middleton and Kang, 2017).

Type of measure	Measure
Agronomic	Maintaining a sufficient vegetative cover Contour cropping Mulching and residue management Strip contour cropping Crop rotation
Soil management	Reduce or eliminate tillage Reduce cultivated fallow Avoid overgrazing
Protective barriers	Plant and maintain field shelterbelts Establishment fences or walls as windbreaks

The most effective method to prevent erosion is to increase vegetative cover, which reduces surface runoff and soil loss due to wind and increases water infiltration. Soil stabilisation can be achieved by introducing plant species with a rich root system (grasses, shrubs and trees). Traditional planting methods, such as mixed-cropping and crop rotation, can significantly reduce erosion rates (Middleton and Kang, 2017; Siebielec et al., 2019). An important measure to reduce erosion of arable land is contour cropping, namely growing crops in rows and strips perpendicular to the slope.

Soil mulching through the use of crop residues plays a role in mitigating soil erosion mediated by water and wind. Leaving crop residues on the land promotes water retention in soil and decreases the susceptibility of soil particles to water and wind energy (Middleton and Kang, 2017; Siebielec et al., 2019). Wind erosion can be successfully combated through vegetation practices, such as planting shelterbelts or windbreaks perpendicularly to the prevailing wind directions. The application of permanent grasses, crop rotations with legumes and conservation tillage also help to manage wind erosion (Siebielec et al., 2019). Erosion can also be reduced through avoiding land use changes such as deforestation and conversion of grassland to cropland.

Dust forecasts play a crucial role in reducing health risks associated with the soil dust (e.g., Saharan dust intrusion). Alert messages and general recommendations to the public and especially the elderly such as to minimise time outdoors and physical activity during dust storms could improve public health. Street sweeping and cleaning could be efficient in reducing Saharan dust particles available for resuspension. Recent studies provide evidence that these methods reduce resuspension from paved road surfaces (Karanasiou et al., 2012; Middleton and Kang, 2017).

5 Impact of other degradation processes (flooding and landslides) on human health

Floods and landslides are major natural hazards; they cost millions of euros in property damage and claim many lives each year in almost all areas in Europe (Hajat et al., 2005; Jakubicka et al., 2010; Menne and Murray, 2013; Stolte et al., 2016). Both the probability and consequences of floods and landslides are expected to increase in the coming decades as a result of climate change and increased vulnerabilities, especially in urban areas (Andersson-Sköld and Nyberg, 2016).

Flooding can be defined as the overflowing by water of the normal confines of a watercourse or water body and/or the accumulation of drainage water over areas that are not normally submerged. Small and large-scale temporary flooding of soil can cause significant soil degradation, e.g., erosion, mudflows, nutrient leaching, decline in soil biodiversity and changes in soil chemical properties (Stolte et al., 2016). Soil degradation as a result of intensive land take and soil sealing leads to a loss of water retention areas, an increase in surface water runoff and in some cases extreme flooding (Pistocchi et al., 2015; Siebielec et al., 2015). Heavy rainfall, melting snow and dam break flows are the cause of inland flooding. There are two main types of river floods that affect Europe: flash floods and slow-rising riverine floods (Hajat et al., 2005; Stolte et al., 2016). A third category is coastal floods. Flash floods are mainly local or regional and are associated with intense thunderstorm activity, when the excess of water overwhelms the drainage capacity of river basins. Slow rise events, during which water accumulates over longer periods of time, are characteristic for the large rivers of northern Europe, e.g., the Rhine, Vistula, Thames, Loire and Rhone (Hajat et al., 2005; Jakubicka et al., 2010).

A **landslide** is defined as the movement of a mass of rock, debris, artificial fill or earth down a slope, under the force of gravity, causing a deterioration or loss of one or more soil functions (Kennedy et al., 2015; Stolte et al., 2016). Landslides in Europe are dominantly considered to be a local soil threat in mountainous, hilly and coastal regions. Hazards posed by landslides are accidental and dynamic (Haque et al., 2016; Stolte et al., 2016).

Natural disasters such as floods and landslides will continue to increase the global burden of disease, morbidity, mortality and social and economic disruptions and will continue to place stress on health services, especially in low-resource countries (Alderman et al., 2012). The literature on the health impacts of flooding and landslides in Europe is limited (Hajat et al., 2005; Guzzetti et al., 2005; Tunstall et al., 2006; Jakubicka et al., 2010; Kennedy et al., 2015; Haque et al., 2016), and it is often difficult to link the health effects separately to floods or landslides. The adverse human health consequences of flooding are complex; these include drowning, injuries and an increased incidence of common mental disorders (Hajat et al., 2005; Tunstall et al., 2006; Menne and Murray, 2013). They are relatively well documented only for the UK (Tunstall et al., 2006; Waite et al., 2017). There is very limited data with regards to the links between landslides and human health outcomes (Guzzetti et al., 2005; Kennedy et al., 2015; Haque et al., 2016).

5.1 Methodologies applied for evaluating the impact of floods and landslides on human health

The research methods used to assess **health impact of floods** vary from observational studies that describe trends in routine health data in flood-affected areas to analytical epidemiological studies that link detailed exposure data to health data, as well as community survey, emergency response and outbreak investigations (Keshishian, 2014). Most studies were cross-sectional, usually describing the prevalence of symptoms in the preceding period (Tunstall et al., 2006; Waite et al., 2017). In the majority of research, the exposure is based on geographical units and the assumption that the entire population living in an administrative area had the same exposure. Community surveys and interviews aim to assess the incidence of chronic diseases where good routine health data were not available, such as mental health, coughs and mild injuries (Tunstall et al., 2006; Keshishian, 2014). A summary of the main health outcomes after flooding and the sources of data is presented in Table 5.1.

Some methodological limitations and difficulties have been identified for epidemiological studies of flooding (Ahern et al., 2005). The majority of available data were based on retrospective analyses and case studies and thus may to be prone to recall bias. Another point is the lack of clinical diagnosis of health outcomes and good baseline pre-flood data. It is also difficult to assess the duration of symptoms and diseases and theirs causes without longitudinal data.

No specific research methods were found to assess the **health impact of landslides**.

5.2 Pathways of exposure and possible health effects

The range of health consequences that may result from **flooding** is broad. These health impacts may occur in different time periods (immediate and long-term effects) and may be categorised as both direct and indirect. The summary of possible human health effects of flooding most frequently reported in EU member States are given in Table 5.1.

Table 5.1: Human health consequences of floods from the literature, sources of health data and health effects reported during and after floods in European countries.

	Health effects[1]	Sources of health data [2]	No of countries[3]	Countries[3]
Direct - short term	Mortality	Routine data: death certificates/mortality data Emergency response databases		
	Injuries	Community surveys	during floods: 10	Czech Republic, Georgia, Hungary, Malta, Republic of Moldova, Slovenia, Tajikistan, Turkey, Ukraine, United Kingdom
			after floods: 11	Georgia, Hungary, Malta, Republic of Moldova, Slovenia, Tajikistan, Turkey, Ukraine, United Kingdom
	Chemical exposure	Routine data: poisoning calls to help lines Routine data: hospitals visits		
	Hypothermia			
Indirect - short term	Communicable diseases		during floods: 4	Bosnia and Herzegovina, Hungary, Ukraine, Tajikistan
	Water-borne diseases gastrointestinal diseases hepatitis A and E respiratory infections skin infections leptospirosis	Emergency outbreak investigations Community surveys Routine data: hospital visits Routine data: laboratory reports		
	Vector-borne diseases	Emergency outbreak investigations Routine data: hospital visits Routine data: laboratory reports		
Direct - long term	Non-communicable diseases			
	Mental health	Community surveys Routine data: prescription data	during floods: 3	Poland, Slovenia, United Kingdom
			after floods: 4	Poland, Slovenia, Spain, United Kingdom
	Birth outcomes			

	Health effects[1]	Sources of health data [2]	No of countries[3]	Countries[3]
Indirect - long term	Malnutrition			
	Health outcome associated with damage to:			
	health care infrastructure		Disruption of routine hospital care during floods: 3	Republic of Moldova, Ukraine, United Kingdom*
	water and sanitation infrastructure		safe water shortages during floods: 11	Armenia, Bosnia and Herzegovina, Croatia, Georgia, Hungary, Republic of Moldova, Slovenia, Tajikistan, Macedonia, Ukraine, United Kingdom
			safe water shortages after floods: 12	Armenia, Bosnia and Herzegovina, Georgia, Hungary, Poland, Republic of Moldova, Slovenia, Tajikistan, Macedonia, Turkey, Ukraine, United Kingdom
	crops and disruption of food supplies		food shortages during floods: 2	Poland, Turkey
	property (lack of shelter)			
	population displacement			

[1]Human health consequences of floods reported in the literature (compilation based on Du et al., 2010; Alderman et al., 2012).
[2]Health outcomes studied after floods and sources of health data (according to Keshishian, 2014).
[3] Health effects reported during and after floods in European countries (Menne and Murray, 2013).
** data for the United Kingdom only for England and Wales.*

Direct effects are the result of direct exposure to the water or flooded environment and mainly include mortality, injuries from debris, chemical contamination and hypothermia (Du et al., 2010; Jakubicka et al., 2010). Indirect consequences are those associated with the risk of damage caused by the water to the natural and built environment and include infectious diseases, malnutrition, poverty-related diseases and diseases associated with displaced populations (Few et al., 2004; Du et al., 2010).

Different types of floods affect human health in distinct ways: physical and mental (Hajat et al., 2005; Du et al., 2010). There is limited quantitative evidence of the health effects of floods, particularly in relation to morbidity (Ahern et al., 2005; Jakubicka et al., 2010). Different kinds of injuries (sprains and contusions) can occur during and after the flood, throughout the clean-up phase and finally during repopulation (Hajat et al., 2005; Alderman et al., 2012). The main health impacts are death and mental health illness during the flood event itself, during the restoration process or from knock-on effects from damage to major infrastructure including displacement of populations. The main reason for mortality is drowning, heart attack, trauma and vehicle-related accidents (Few et al., 2004; Alderman et al., 2012).

Floods may trigger the release of chemicals (e.g., pesticides, agricultural chemicals, dioxins and HM) that are already stored in the environment. Exposure to these chemical agents may result in different diseases (Du et al., 2010; Alderman et al., 2012). Furthermore, Few et al. (2004) and Ahern et al. (2005) reported that European floods are associated with an increased risk of communicable diseases, which include gastrointestinal infections, skin irritations and respiratory infections.

Long-term impacts of floods manifest in the occurrence of non-communicable, chronic diseases and mental health effects as a result of exposure to human and animal viruses during evacuation, or substantial psychological or physical stress at the time of flooding (Few et al., 2004; Du et al., 2010; Alderman et al., 2012; Stephenson et al., 2014; Waite et al., 2017). A cohort study conducted by Waite et al. (2017) in England revealed a significant increase in psychological morbidity amongst respondents directly exposed to floods. Depression (20%), anxiety (28.3%) and post-traumatic stress disorder (PTSD; 36.2%) were the most frequently reported mental disorders.

The health consequences of floods depend upon various factors, including geographic and socioeconomic factors, the characteristic of the flood (i.e., its scale and duration, the suddenness of the onset, the velocity and depth of the water and the lack of warning) and the baseline vulnerability of the affected populations. Floods with the largest mortality have occurred where infrastructure is poor and the population at risk has limited economic resources (Ahern et al., 2005; Tunstall et al., 2006; Du et al., 2010; Alderman et al., 2012). The groups within communities that are most vulnerable to the health impacts of flooding are the elderly, disabled, children, women, ethnic minorities and those on low incomes. People with limited physical capacity and restricted mobility, who rely on medication, who require home care or regular visits to health care facilities and who have weak social networks and little access to flood warnings are at particularly high risk (Menne and Murray, 2013; WHO, 2017). The persistence of the flood-related health effects is directly related to flood intensity (Hajat et al., 2005).

The literature on **landslides** is rich with detailed technical and geological papers that cover spatial and temporal distributions of landslides, the processes causing landslides and the approaches to mitigate their effects (EEA, 2010; Geertsema and Highland, 2011; Andersson-Sköld and Nyberg, 2016; Stolte et al., 2016), while the published literature on the health impacts of these disasters in Europe is exceptionally limited (Guzzetti et al., 2005; Kennedy et al., 2015; Haque et al., 2016). Only one paper (Haque et al., 2016) refers to the number of fatalities due to landslides in European countries. Haque et al. (2016) analysed landslide events that occurred in Europe between 1995 and 2014. They found 1,370 deaths and 784 injuries resulting from 476 landslides. The most frequently affected country was Turkey (with 335 landslide-induced deaths), followed by Italy (283), Russia (169), and Portugal (91; Haque et al., 2016). The authors highlighted that data on the links between landslides and human health in the International Disaster Database are underestimated because most of the landslide casualties are registered under the main events that trigger them, such as an earthquake, storm and/or flood (Haque et al., 2016).

5.3 Measures to reduce health risk

The population at risk, policy makers and emergency responders should undertake activities to reduce health risk caused by flooding and landslides. Mitigation measures may reduce but not eliminate major damage. The harmful effects of flooding can be reduced by building codes, legislation to relocate structures away from flood-prone areas and planning appropriate land use (Menne and Murray, 2013; WHO, 2017). All action plans on flood are closely linked to a local context. However, because it was recommended by the European core group on flood protection, they should combine four types of measures: information, prevention, protection and emergency (Table 5.2; Lamothe et al., 2005).

Table 5.2: Types of measures to reduce impact of floods (modified from Lamothe et al., 2005).

Type of measure	Measure
Information	Flood risk mapping and communication Flood forecasting and early warning system Public awareness on best practices Establishment of emergency plan
Prevention	Limit the use of floodplains Increasing retention capability of soils Increasing retention capability of floodplains and wetlands
Protection	Measures to reduce pick run-off (flood control, emergency reservoir) Reduce level of flooding for give run-off (dikes) Storage of valuable or environmentally harmful goods
Emergency	Implementation of emergency plan

Table 5.3: Types of measures to reduce landslides (according to the RECARE project[22]).

Type of measure	Measure
Vegetative	Preserving vegetation, grasses and trees
Structural	Constructing piles and retention walls Improving surface and subsurface drainage Excavating head and buttressing toes Rock-fall protection

Landslides can be prevented by soil reinforcements and erosion-prevention measures in the most landslide-prone areas (Table 5.3). Important nonphysical measures are education, information campaigns, communication with landowners, increased preparedness, recommendations and restrictions in master plans and in detailed spatial planning (Andersson-Sköld and Nyberg, 2016).

There is a need to shift the emphasis from disaster response to risk management and prevention, to improve flood forecasting, to establish early warning systems and to include health actors in the communication flow (Stephenson et al., 2014). Risk management in this area must cover a broad field, including health impact assessment of flood structural measures, regulations concerning building in flood- or landslide-prone areas and insurance policies.

[22] https://www.recare-hub.eu/soil-threats/soil-erosion

6 Links between losses of soil biodiversity and human health

Biodiversity is defined in the Millennium Ecosystem Assessment as "the variability among living organisms in terrestrial, marine, and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within and between species, and the diversity of ecosystems". Soil biodiversity is the diversity of living organisms in the soil (Jeffery et al., 2010; WHO, 2015). Soil organisms may be divided into macro-, meso-, and microfauna. Bacteria, fungi, protozoa and algae are grouped as microorganisms. Biological activity in soils is largely concentrated in the topsoil, consists of plant roots and soil organisms, is a small fraction of the total soil volume (< 0.5%) and makes less than 10% of the total soil organic matter (Breure, 2004). All processes that regulate nutrient cycles and decomposition of organic residues are responsible for almost 80% on the proper functioning of soil microorganisms.

In general, the state of soil biodiversity has been well described in the European Atlas of Soil Biodiversity (Jeffery et al., 2010). Soil biodiversity management has also been promoted in the context of the Convention on Biological Diversity (CBD) and the 2030 Agenda for Sustainable Development by relating the soil function "biodiversity pool, such as habitats, species and genes" to the topics "ensure healthy lives and promote well-being for all at all ages" and "protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification and halt and reverse land degradation and halt biodiversity loss" (Keesstra et al., 2016). In the Thematic Strategy for Soil Protection (COM(2006) 231), a decline in soil biodiversity was identified as one of the eight soil threats. Biodiversity loss is considered to be the reduction of life forms in soils, both in terms of quantity and variety of related soil functions (Aksoy et al., 2017).

Soil biodiversity is threatened by several factors (Figure 6.1) associated with human activity, including soil erosion, salinisation, compaction, sealing, contamination, organic matter decline, climate change, land use change, habitat fragmentation, intensive soil exploitation, use of genetically modified organisms (GMO) in agriculture and introduction and diffusion of invasive species (Gardi et al., 2013; Orgiazzi et al., 2016; Aksoy et al., 2017).

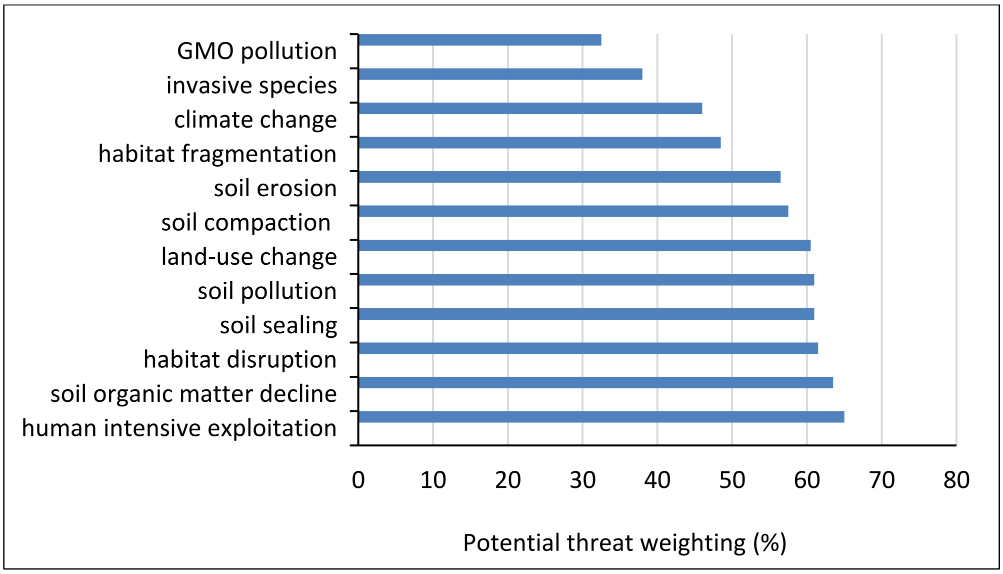

Figure 6.1: The potential weighting of possible threats to soil biodiversity (provided by the Soil Biodiversity Expert Workgroup of the European Commission and modified from Jeffery et al., 2010; Gardi et al., 2013).

In agricultural soils, the main drivers of soil biodiversity loss are intensification of land use, simplification of crop rotation and cropping in monocultures, excessive application of pesticides and fertilisers (Breure, 2004). All these threats impair soil biodiversity and functioning and may ultimately decrease their ability to deliver ecosystem services (Table 6.1), including provisioning (food, fibre, timber and fuel), regulating (climate, disease and natural hazards), supporting (nutrient cycling and soil formation) and cultural services (Aerts et al., 2018). The consequence of declining soil biodiversity is a deterioration in the soil health and productivity, which may translate into human health effects.

Table 6.1: Essential ecosystem services provided by soil biota (modified from Jeffery et al., 2010).

Ecosystem services	Soil biota groups that provide a service
Decomposition and cycling or organic matter	Bacteria, fungi and actinomycetes; meso- and macrofauna such as various saprophytic and litter that feed invertebrates including earthworms, ants, Collembola and mites
Regulation of nutrient availability and uptake	Mostly microorganisms like mycorrhizae, actinomycetes, nitrogen fixing and nitrifying bacteria, some soil and litter that feed invertebrates (ants and earthworms)
Suppression of pests and diseases	Bacteria, fungi, nematodes, Collembola, earthworms and decomposers, as well as predators
Maintenance of soil structure and regulation of soil hydrological processes	Bioturbation by invertebrates such as earthworms, ants, termites and plant roots, mycorrhizae and some other organisms (microstructure)
Gas exchanges and carbon sequestration	Mostly microorganisms and plant roots, some organic carbon protected in biogenic aggregates made by earthworms, ants or termites
Soil detoxification	Mostly bacteria and fungi
Plant growth control	Plant roots, rhizobia, mycorrhizae, actinomycetes, pathogens, phytoparasitic nematodes, rhizophagous insects, plant growth promoting rhizosphere organisms, biocontrol agents
Pollination of horticultural crops	Soil-nesting insects such as solitary bees

As shown in Figure 6.2, poor land management and climate change may influence the composition and structure of soil biodiversity, which in turn directly affect ecosystem functioning and reduce the services the ecosystem provides towards human health (Huynen et al., 2004; Wall et al., 2015).

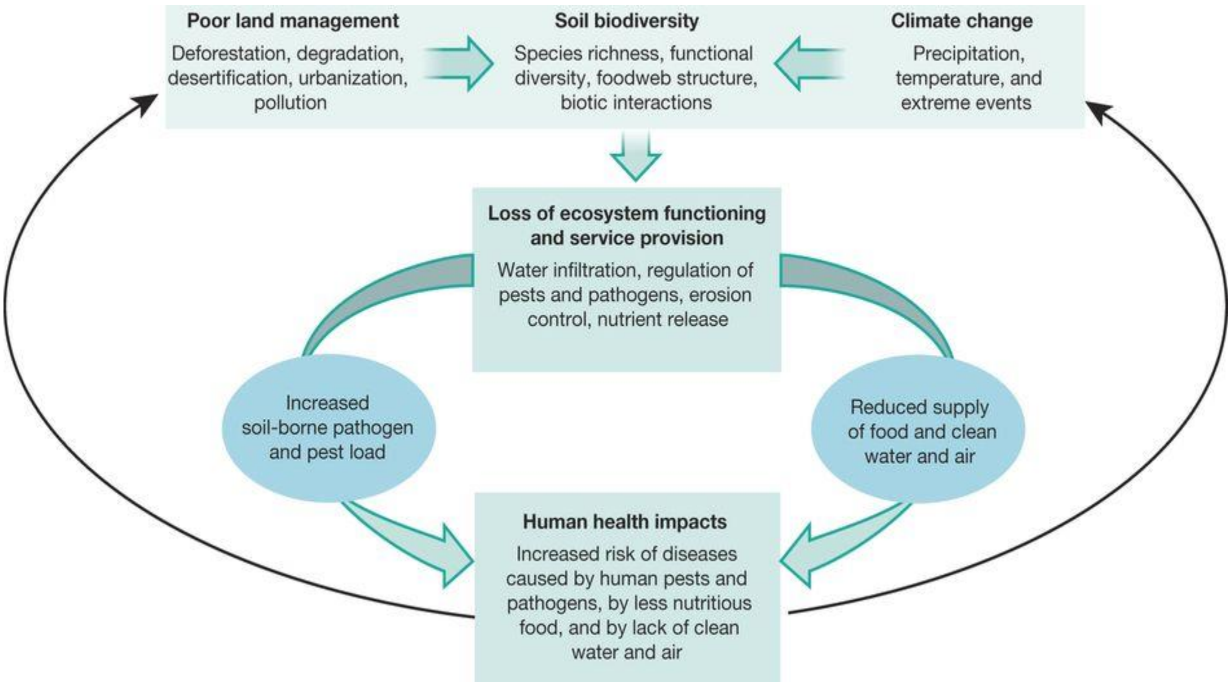

Figure 6.2: Soil biodiversity and its relationships to human health (Wall et al., 2015).

Soil biodiversity can have both direct and indirect effects on human health (Clark et al., 2014). Soil biodiversity loss can lead to the emergence and transmission of infectious diseases and cause health effects through the loss of food and nutritional diversity. Other potential consequences of biodiversity decline include diminished supplies of raw materials for drug discovery and biotechnology, as well as threats to food production and water quality (Huynen et al., 2004). Biodiversity loss may affect cultural values and cause significant consequences for human well-being by generating anxiety, frustration and stress (Clark et al., 2014). Furthermore, as a consequence of global environmental changes and associated losses in soil biodiversity, we could lose a possible source of antibiotics, medicines and the biological controls needed to prevent human, animal and plant disease. On the other hand, a rapid increase in antibiotic resistance to microbial-derived medicines has been observed recently. This phenomenon threatens the prevention and treatment of diseases caused by bacteria, fungi and parasites (Wall et al., 2015).

The vast majority of data on biodiversity decline address the state (Jeffery et al., 2010; WHO, 2015; Orgiazzi et al., 2016; Aksoy et al., 2017), indicators and measures of soil biodiversity (Breure, 2004; Stone et al., 2016). A comprehensive overview of the links between biodiversity and human health in the term of human infectious diseases and soil borne diseases was provided by Zaghi et al. (2010) and Jeffery and van der Putten (2011). Apart from the negative effects on human health, some authors underscored the many benefits that biodiversity creates to human well-being (Wagg et al., 2014; Civitello et al., 2015; Wall et al., 2015; Sandifer et al., 2015).

6.1 Pathogens versus bacteria, potential negative effect

Soil is a reservoir for many microorganisms both non-pathogenic and pathogenic. Soil-borne pathogens and parasites that cause human diseases represent a minority of the species that live in soils. Most soil organisms pose no risk to human health (Brevik and Sauer, 2015; Wall et al., 2015; Steffan et al., 2018). Soil microbes interact with one another and also adapt to extreme conditions of temperature, pH and moisture in soil and, consequently, genotypic and phenotypic modifications can occur that alter their virulence (Pepper et al., 2009; Oliver and Gregory, 2015). Pathogenic organisms present in the soil enter humans by three pathways: ingestion (e.g., *Clostridium botulinum*), inhalation (e.g., *Aspergillus fumigatus*) and through the skin and skin lesions (hook worms and *Clostridium tetani*, respectively).

Soil pathogens can be described as geo-indigenous, geo-transportable or geo-treatable (Pepper et al., 2009; Oliver and Gregory, 2015; Wall et al., 2015). Geo-indigenous pathogens are native to soil, capable of metabolism, growth and reproduction, occur in almost all soils and include a variety of prokaryotic and eukaryotic organisms (Table 6.2).

Table 6.2: Soil pathogens and parasites of humans (compilation based on Jeffery and Van der Putten 2011; Pepper 2013; Wall et al., 2015).

Type of organism	Disease caused
Euedaphic pathogens	
Bacteria	
Bacillus antracis	Anthrax
Listeria monocytogenes	Listeriosis
Legionella spp.	Legionnaire's disease
Clostridium perfringens	Minor infections and gas gangrene
Fungi	
Aspergillus spp.	Aspergillosis
Protozoa	
Naegleria fowleri	Brain encephalitis
Soil-transmitted pathogens	
Bacteria	
Escherichia coli	Diarrhoea
Salmonella spp.	Salmonellosis
Shigella dysenteriae	Shigellosis
Pseudomonas aeruginosa	Shigellosis
Protozoa	
Toxoplasma gondii	Toxoplasmosis
Entamoeba histolytica	Amoebiasis
Giardia lamblia	Giardiasis
Helminths (Nematoda)	
Ascaris lumbricoides	Ascariasis
Ancylostoma duodenale	Hookworm
Strongyloides stercoralis	Strongyloidiasis
Trichuris trichiura	Trichuriasis
Taenia saginata	Taeniasis

Of the geo-indigenous pathogens, only the spore-forming ones are known to be consistently problematic to human health due to their long-time surviving in soils and ease in which they can be disseminated as aerosols via airborne transmission (Pepper et al., 2009; Brevik and Burgess, 2013). Geo-transportable pathogens can be transported from soil via water or dust. Geo-treatable pathogens are viruses, bacteria, protozoa and helminths introduced into soil deliberately or accidentally via anthropogenic activities. Such pathogens can get into the soil as a result of land application of manure and biosolids and are normally rapidly inactivated in the soil by biological and abiotic factors (Pepper et al., 2009; Oliver and Gregory, 2015; Wall et al., 2015). Land application of manure or biosolids may be a source of bacteria (*Salmonella*, *Listeria* and *Escherichia coli*), protozoa (*Giardia lamblia*, *Toxoplasma gondii*) and helminths (*Ascaris lumbricoides* and *Ascaris suum*). Geo-treatable viruses and bacteria rarely survive longer than a few weeks or months, while geo-treatable helminths are very resistant and can survive several years (Pepper et al., 2009; Pepper 2013).

Some soil-borne pathogens (e.g., the bacterial genera *Pseudomonas* and *Enterobacter*) play a crucial role in the soil food web as antagonists against plant root pathogens, promoters of plant growth and decomposers, but they are opportunistic species they can infect and cause diseases in humans (Berg et al., 2005; Pepper et al., 2009; Wall et al., 2015). A good example is rhizosphere bacteria from *Burkholderia*, *Stenotrophomonas* or *Ochrobactrum* genera, which may cause bacteremia, endocarditis and respiratory tract infections (LiPuma et al., 2002; Berg et al., 2005). The most susceptible group of patients to the infections caused by opportunistic pathogens are older patients with chronic diseases, patients with long-term antibiotic therapy and immune-suppressed patients (Berg et al., 2005).

6.2 Antimicrobial resistance

The excessive utilisation of antimicrobial pharmaceuticals to treat and prevent human and animal diseases has generated antimicrobial resistance[23] against these compounds. WHO (WHO, 2015; Rodríguez-Eugenio et al., 2018) highlighted that antibiotic resistance[24] in any environment can pose serious threats to public health and the environment. The soil resistome[25] includes genes that confer antibiotic resistance to pathogenic and non-pathogenic species present in the environment, as well as to proto-resistance genes that serve as a source of resistant elements. Recent scientific studies focused on the identification of genes for antimicrobial resistance by a metagenomic[26] approach (de Castro et al., 2014; Nesme and Simonet, 2015).

Soil microorganisms, which are the main source of antibiotics, have developed natural resistance mechanisms to prevent themselves from detrimental influence from the compounds they produce. There is evidence for the resistance of soil bacteria to antibiotics such as ß-lactams, aminoglycoside and tetracycline (Thiele-Bruhn, 2003; Schmieder and Edwards, 2012; Hashmi et al., 2017).

[23] Antimicrobial resistance occurs when microorganisms (such as bacteria, fungi, viruses and parasites) change when they are exposed to antimicrobial drugs (antibiotics, antifungals, antivirals and anthelmintics), according to WHO: https://www.who.int/en/news-room/fact-sheets/detail/antimicrobial-resistance.

[24] Antibiotic resistance occurs when bacteria change in response to the use of these medicines, according to WHO: https://www.who.int/en/news-room/fact-sheets/detail/antibiotic-resistance.

[25] The soil resistome is the collection of all the antibiotic resistance genes and their precursors in both pathogenic and non-pathogenic bacteria (Nesme and Simonet, 2015).

[26] Metagenomics is application of modern genomic techniques (DNA extraction and sequencing) to the study of communities of microbial organisms directly in their natural environments, bypassing the need for isolation and lab cultivation of individual species (de Castro et al., 2014).

Generally, bacterial resistance to antibiotics is the result of genetic changes or spontaneous mutation in the target gene (de Castro et al., 2014; Mullany, 2014). The resistance strategy includes the production of specific enzymes and inactivation or degradation of antibiotics, changes in bacterial cell wall permeability preventing the entrance of antibiotics, active transportation systems like the efflux pump that exports antibiotics out of the bacterial cell (Mullany, 2014; Grenni et al., 2018; Li et al., 2018).

Many anthropogenic factors, such as the application of manure and sludge to soil as fertilisers, antibiotic use in humans and livestock and irrigation with reclaimed water, can contribute to dissemination of antibiotics and antibiotic resistance genes (ARG) in soil (Schmieder and Edwards, 2012; Hashmi et al. 2017; Steffan et al., 2018). ARG are emerging contaminants that pose a potential human health risk worldwide. Soil is a rich source of ARG because of the large microbial community and diversity of antibiotic-producing microorganisms in soil (Grenni et al., 2018; Li et al., 2018; Rodríguez-Eugenio et al., 2018). Microorganisms, which are not susceptible to some antibiotics, are capable of developing resistance over time. Some microorganisms can grow even when exposed to different antibiotics and even use some of the antibiotics as a food source (Schmieder and Edwards, 2012). Antibiotic resistance occurs because antibiotics provide an evolutionary pressure on a given population whereby those organisms with natural resistance can survive and reproduce. Once a resistance factor has developed, it can spread rapidly within a population or a community through horizontal gene transfer by passing genetic material (small molecules of DNA) from one bacterium to another, even when the latter is phylogenetically distant (Jeffery et al., 2010; Grenni et al., 2018; Xie et al., 2018). Increasing the number of antibiotic resistant bacteria in the environment may increase the transfer of resistance to organisms, which can cause disease in humans (Pepper et al., 2009; Jeffery and Van der Putten, 2011). As stated by de Castro et al. (2014), there is a link between resistance genes in human pathogens and those found in commensal microorganisms, with several common bacteria resistance taxa such as *Staphylococcus aureus*, *Pseudomonas aeruginosa* and *Klebsiella pneumoniae*, all of which are ubiquitous in soils.

Antimicrobial resistance can also develop through co-resistance, specifically as a result of the presence of biocides and heavy metals. ARG and the resistance genes for metals are often located together on the same genetic element (Grenni et al., 2018). Application of sewage sludge that contains bacteria with class 1 integrons (the resistance genes for almost all antibiotic families) to soil can increase the reservoir of antibiotic-resistant bacteria in humans. Exogenous bacteria are mixing with antibiotic-producing bacteria that occur naturally in soil, a phenomenon that can result in horizontal gene transfer that produces antibiotic-resistant genes. Class 1 integrons are genetic elements that carry antibiotic and quaternary ammonium compound resistance genes and resistance to detergents and biocides (Oliver and Gregory, 2015; Xie et al., 2018).

To diminish the impact of antibiotics used in agriculture on a selection of resistant strains among human pathogens, in 2006[27] the EU banned the use of antibiotics as growth promoters in livestock.

[27] EU ban on antibiotics as growth promoters in animal feed https://ec.europa.eu/commission/presscorner/detail/en/IP_05_1687

Administration of antibiotics at low doses (below the minimum inhibitory concentration) modulates the metabolism of commensal bacterial flora and can promote the spread of the antimicrobial resistance (Hashmi et al. 2017; Grenni et al. 2018).

6.3 Hampered nutrient cycling and degradation processes

Soil biota perform many beneficial activities that improve soil quality and hence soil productivity (Table 6.1). Soil biota recycle basic nutrients required by plants for their growth; the tunnelling and burrowing activities of earthworms and other soil biota enhance productivity by increasing water infiltration into the soil. Soil microorganisms play a crucial role in organic matter turnover, nutrient release and stabilisation of the soil structure and ensure soil fertility. Moreover, many organisms act as biological control agents by inhibiting the growth of pathogens. Soil homeostasis may be disturbed by biotic and abiotic factors, such as bacteriophages, predation, competition, pesticides, HM, toxic hydrocarbons and antibiotics, all of which reduce microbial biodiversity. Xenobiotics can affect soil ecological functions through changes in nitrogen transformation, methanogenesis, sulphate reduction, nutrient cycling and organic matter degradation (Brevik and Burgess, 2013; Grenni et al., 2018).

A decline in soil biodiversity can impair numerous ecosystem functions, including plant uptake of nutrients, nutrient cycling and retention. Wagg et al. (2014) demonstrated that soil biodiversity loss and changes of soil community composition in grassland communities adversely affects plant diversity, decomposition, nutrient retention and nutrient cycling. Simplification of soil communities increases phosphorus leaching and nitrogen losses via N_2O (an important greenhouse gas) emissions. Naveed et al. (2014) reported that a loss of soil biodiversity as a result of soil contamination with copper (Cu) adversely affects natural soil bioturbation, aggregate formation and stabilisation, decomposition and mineralisation processes and promotes compacted soil with narrow pore size distribution and lower total porosity, restricted air and water storage flow and impedes C, N, and P cycling.

Soil fertility determines the quantity and quality of food that can be grown on a given area of land, and its productivity indirectly determines our health (Brevik and Sauer, 2015; Oliver and Gregory, 2015; Steffan et al., 2018). The nutritional value of foods is affected by soil quality. Mineral or nutrient malnutrition results from crops produced on soil with poor phytoavailability of the elements essential to human nutrition. A nutrient deficit is caused by prevalence of extractive farming practices, including removal of crop residues, lack or low rate of application of inorganic and organic fertilisers and uncontrolled grazing (Lal, 2009). Alkaline and calcareous soils have limited availabilities of Fe, Zn and Cu; coarse-textured, calcareous or strongly acidic soils contain little Mg, and soil derived from igneous rocks contains minimal Se. Nutrient deficiencies can have significant consequences for human health (Lal, 2009; Steffan et al., 2018). They can result in anaemia, diseases of the immune system, mental retardation and cardiovascular diseases, and have a greater impact during pregnancy, lactation and periods of rapid growth, such as early childhood. Nutrient deficiencies might be an underlying factor in many diseases in developed countries, in particular Se and Zn. To correct Zn deficiency and prevent yield loss in crops, Zn is applied to deficient soil as $ZnSO_4$ at rates of 5–25 kg Zn ha^{-1} (Cakmak and Kutman, 2018). The main advantages of soil biofortification with Zn are better yields, improved vigour of seedlings and reduced root uptake and shoot/grain accumulation of Cd, which is beneficial for human health. Symbiotic soil microbes are essential for nutrient supply and can contribute to biofortification of plants for important micronutrients such as Zn (Wall et al., 2015).

The vast majority of soil microorganisms are decomposers; they can degrade different types of organic substances. Readily degradable compounds (e.g., carbohydrates and amino acids) are susceptible to decomposition by a wide range of soil microbial groups. Complex substrates like lignin, cellulose and hemicellulose are highly recalcitrant and can only be degraded by a selective group of soil microbes, such as white rot fungi and some bacteria (Jeffery et al., 2010). Soil microbial communities are key players in several processes controlling the quality of soil and water ecosystems and regulating the fate of pollution released into the environment. Microorganisms are involved in ecosystem self-purification processes; they can degrade different contaminants by metabolic or co-metabolic pathway to smaller, non-toxic molecules. Most xenobiotics are removed from the soil environment during biodegradation depending on the presence of microbial population capable of degrading specific pollutants and resistant to their harmful effects. The xenobiotic biodegradation rate is related to abiotic factors such as temperature, water content, soil properties and co-occurrence of other contaminants. Generally, if bacterial community diversity is high, the probability of biodegradation of a compound is also high (Grenni et al., 2018).

7 The positive role of soil on human health

Soils provide many benefits to human health and well-being (see section 1.2). This chapter describes the most important positive effects of soils on human health.

7.1 Use of soil biodiversity for pharmaceuticals

The most important medical application of soil has been the isolation of antibiotics from indigenous organisms (rhizosphere bacteria and endophytic microbes, actinomycetes and fungi). Antibiotics are produced in the soil by microorganisms as secondary metabolites at concentrations lower that those used in medicine (Hashmi et al., 2017). They are generally classified according to their effect on the competing microorganisms as bactericidal (killing bacteria) and bacteriostatic (impair microbial growth; Grenni et al., 2018). Soil microorganisms are the main organisms that produce antibiotics (Table 7.1); over 60% of the entire production of antibiotics in soil is attributed to microbes. Among soil microorganisms, more than 50% of *Actinobacteria* isolated from soil can synthesise antibiotics, which are found mainly in the soil rhizosphere with concentrations of up to 5 µg g^{-1} (Thiele-Bruhn, 2003). The most popular antimicrobial pharmaceuticals produced by *Actinobacteri*a (*Streptomyces*, *Micromonospora* and *Saccharopolyspora* genera) are tetracyclines, gentamycin and erythromycin. Fungi generate approximately 18% of all antibiotics, with imperfect fungi accounting for 12%, while *Basidiomycota* and *Ascomycota* contribute around 6% (Hashmi et al., 2017).

Eighty percent of antibiotics in clinical use today have their origin in soil bacteria, either directly as natural products or as their semi-synthetic derivatives (Schmieder and Edwards, 2012). The first antibiotics isolated from soil organisms were penicillin, actinomycin, neomycin and streptomycin (Jeffery et al., 2010). Penicillin was isolated from the soilborne fungus *Penicillium* by Sir Alexander Fleming in 1929, and streptomycin from actinomycete *Streptomyces griseus* by Selman Waksman in 1943 (Pepper et al., 2009).

Rhizosphere bacteria and endophytic microorganisms are rich sources of metabolites, namely natural products with novel biological activity. Endophytes are bacterial and fungal microbes that colonise plant roots without pathogenic effects; they produce metabolites to improve and protect plant roots and may also improve human health. Endophytes can produce novel antibiotics, antimycotics, immunosuppressants and anticancer agents (Pepper, 2013). The latter are a group of pharmaceuticals that were historically isolated from natural products in soil (Oliver and Gregory, 2015). Paclitaxel, the world's first anti-cancer drug, is produced by many endophytic fungi associated with the yew (*Taxus* L.) species. Other beneficial endophytic natural products include pestacin, with antioxidant activity, bioinsecticides, insect repellents and antidiabetic agents that act as an insulin mimetic and immunosuppressive drugs (Pepper et al., 2009). Many rhizobacteria, like the fluorescent pseudomonads and *Streptomyces* species, also produce antibiotics and can prevent infections by plant pathogens. Antibiotic production can protect rhizobacteria against grazing by protozoa (Berg et al., 2005). The strong biocontrol activity of *Burkholderia* is a result of the production of antifungal compounds (Berg et al., 2005).

Table 7.1: Soil microorganisms that produce antibiotics (modified from Hashmi et al., 2017).

Soil microorganisms		**Antibiotic**
Bacteria		
Streptomyces spp.	*Streptomyces griseus*	Streptomycin
	Streptomyces spectabilis	Spectinomycin
	Streptomyces erythreus	Erythromycin
	Streptomyces aureofaciens	Tetracycline
	Streptomyces venezuelae	Chloramphenicol
	Streptomyces orientalis	Vancomycin
	Streptomyces teichomyceticus	Teicoplanin
Micromonospora spp.	*Micromonospora purpurea*	Gentamycin
Bacillus spp.	*Bacillus licheniformis*	Bacitracin
	Bacillus brevis	Gramicidin
	Bacillus polymyxa	Polymyxin
Fungi		
Penicillium spp.	*Penicillium notatum*	Penicillin
Cephalosporium spp.		Cephalosporin

Different clay minerals have a long history of medicinal use due to their capacity for adsorption and absorption and surface charge (Sing and Sing, 2010; Oliver and Gregory, 2015). Iron-rich smectite and illite clays exhibit natural antibacterial properties, while non-swelling clays like kaolin and attapulgite have been used in diarrhoea medicines and kaolin in ointments as an emollient (Oliver and Gregory, 2015).

7.2 Sorption of pathogens and contaminants

Soils can positively influence human health by acting as a filter to remove hazardous materials and pathogens (Zhao et al. 2012; Brevik and Sauer, 2015). Soils are very effective in inactivating introduced pathogens through competition and biotic and abiotic stresses (Figure 7.1; Pepper, 2013). Populations of soil pathogens are limited by predators and parasites, such as protozoa and bacteriophages, as well as by antagonistic microorganisms and competition for space in an appropriate ecological niche (Johnson and Thieltges, 2010; Civitello et al., 2015).

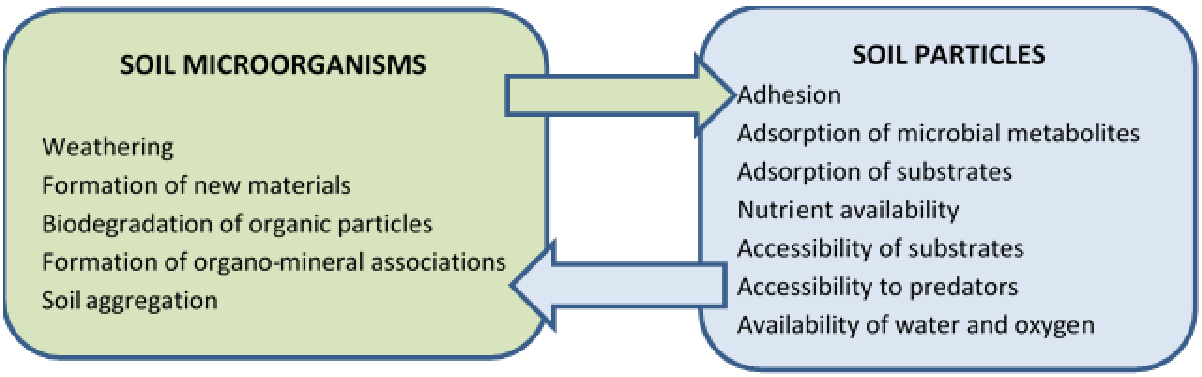

Figure 7.1: Possible interactions between microorganisms and soil particles (modified from Chenu and Stotzky, 2002).

Pathogen virulence and survival in soils is affected by soil properties (pH, organic matter content and soil particle size), soil moisture, temperature and exposure to sunlight (Guzman et al., 2012; Zhao et al., 2015). Pathogens in soils can be transported by surface run-off and underground percolation in the form of free cells or in association with soil particles. Bacterial adsorption on soil is one of the main factors that controls bacterial transport to water bodies (Kraemer et al., 2013). Retention of microbes in soil includes sorption on the external surfaces of soil aggregates and physical entrapment (straining) in small pore spaces (Zhao et al., 2015; Sasidharan et al., 2016). This process may be modified by application of exogenous organic matter like biochar. Sasidharan et al. (2016) demonstrated that biochar influences transport and retention of *E. coli* and bacteriophages through straining by alteration of pore size distribution. As reported by Guzman et al. (2012), soil clay minerals and organic matter are fundamental properties for controlling the transport of faecal bacteria *E. coli* after application of swine effluent into the soil. Retention of these bacteria is primarily controlled by sorption of the bacteria substrate or bacteria aggregates rather than the bacterium itself (Guzman et al., 2012).

Zhao et al. (2012) conducted a study to explain the sorption behaviour of *Streptococcus suis*, a serious zoonotic pathogen that can cause meningitis, septicaemia and pneumonia in pigs and humans. The study revealed that clays with the largest specific surface area (high sorption ability) are more effective in binding *S. suis* compared to silts or sands. Sorption of these bacteria is also inhibited by organic matter and increase of pH from 4.0 to 9.0 (Zhao et al., 2012).

Soil rhizosphere bacteria from the genera *Burkholderia*, *Enterobacter*, *Herbaspirillum*, *Ochrobactrum*, *Pseudomonas*, *Staphylococcus* and *Stenotrophomonas* promote plant growth, have antagonistic properties against plant pathogens and were utilised for the development of biopesticides (Berg et al., 2005; Jeffery et al., 2010).

Some of the main ecological services provided by soil are filtering, buffering and transforming inorganic and organic contaminants, all of which ensure good groundwater quality and safe food production and, consequently, protects human health. Sorption and desorption reactions are the main factors that control the retention and fate of pollutants in soil and their uptake by plants. Sorption may be chemical (as with ionic and hydrogen binding) or purely physical (as with the van der Waals forces) in nature (Rodríguez-Eugenio et al., 2018). The rate of sorption is a function of soil and contaminant properties and environmental factors, like moisture level and temperature. Among the soil parameters, the most important for the retention of contaminants are soil texture (soil mineralogy and clay content), soil pH and the amount and fractional composition of soil organic matter. Relevant contaminant properties include the size, shape, molecular structure, solubility, charge distribution and acid-base nature of the molecule. Generally, a low content of clay and organic matter promotes the bioavailability of contaminants due to limited sorption sites and thus increases the leaching and distribution of the contaminants to underground and surface water. Therefore, pollutants may impair soil functions and pose risks to soil organisms. Sequestration and sorption of pollutants in soil reduce their bioavailability and risk for the environment (Riding et al., 2013; Umeh et al., 2017).

7.3 Positive impact on physical and mental health

Soils contribute directly and indirectly to human health (Figure 7.2). Increasing biodiversity through habitat conservation or restoration may have positive effects on several aspects of well-being, including physical and mental health benefits associated with spending time in nature (Clark et al., 2014; WHO, 2015; Kilpatrick et al., 2017; Aerts et al., 2018). Short-term exposure to green spaces (e.g., forests, urban parks and gardens) can reduce stress and depressive symptoms, restore attention fatigue, increase positive emotions (i.e., vitality, energy and pleasure) and improve self-esteem and mood (Clark et al., 2014; Sandifer et al., 2015; Aerts et al., 2018). Long-term exposure to biodiverse natural environments can reduce all-cause, respiratory, cardiovascular and cancer mortality and positively affects respiratory and mental health (Aerts et al., 2018).

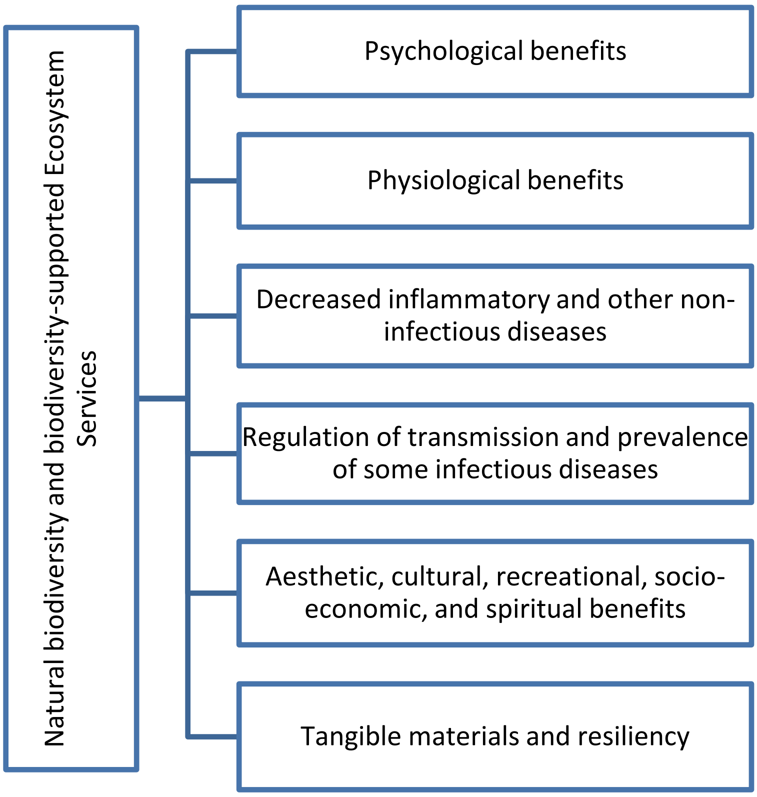

Figure 7.2: Major pathways through which biodiversity may provide health and well-being benefits to humans (modified from Sandifer et al., 2015).

Exposure to plants reportedly accelerates the healing process for patients after surgery. Indeed, walking through a garden helps to restore mental clarity, reduce stress and recover from mental fatigue. Soils are also important for recreation (Sandifer et al., 2015; Brevik et al., 2018a). The most direct link is gardening, where gardeners work intensively with soil. Gardening improves mental and physical health and encourages healthy eating through increased consumption of fresh fruits and vegetables. Recreational activities such as hiking, camping and biking also improve mental and physical health (WHO, 2015; Schram-Bijkerk et al., 2018).

Green spaces in urban areas can limit the negative impact of other factors that affect human health, including poor air quality and heat stress (Gunawardena et al., 2017). The creation of green spaces in urban areas contributes to cooling during hot periods.

Air temperature in tree-dominated greenspace (parks) is lower compared to an open area (Bowler et al., 2010; Claessens et al., 2014). Green spaces encourage people (mainly children) to increase physical activity, a phenomenon that may be translated to the potential reduction of obesity and type 2 diabetes (Claessens et al., 2014; Aerts et al., 2018). The positive effects of green spaces on human health was widely investigated during the EU project PHENOTYPE[28] (Nieuwenhuijsen et al., 2014; Dadvand et al., 2016; Ruijsbroek et al., 2017). Dadvand et al. (2016) conducted a cross-sectional study based on the population of adults in Barcelona (Spain) and found that residencies surrounded by greenness were associated with better general human health. Another cross-sectional study by Petraviciene et al. (2018) reported the significantly higher risk of obesity for 4–6 year-old children living in areas of Kaunas (Lithuania) with less exposure to green spaces. The study of Ruijsbroek et al. (2017), conducted in four European cities—Barcelona (Spain), Stoke-on-Trent (United Kingdom), Doetinchem (the Netherlands) and Kaunas (Lithuania)—revealed a link between human mental health (reduction of nervousness and feelings of depression) and the presence of green space only in Barcelona. In the other cities, these links were related to the social environment.

The positive role of soils on human health is associated with the prevention of immune-related disorders and a lower incidence of allergic diseases as a result of exposure to soil microorganisms. According to the biodiversity hypothesis, the human immune system must be exposed to possible pathogens that reside in soils in order to develop tolerance and improve the immune system (Brevik and Sauer, 2015; Wall et al., 2015; WHO, 2015; Liddicoat et al. 2016; Tasnim et al., 2017; Aerts et al., 2018). Exposure to beneficial environmental microbiota reduces the prevalence of a wide variety inflammation-mediated illnesses, including allergies and asthma, inflammatory bowel disease, cardiovascular disease, some cancers, potentially some neurodegenerative diseases, types 2 diabetes, inflammatory-associated depression and obesity (Sandifer et al., 2015; Aerts et al., 2018). Some results (Hough, 2014; Li et al., 2018) suggest that allergies might result from a lack of exposure to microbes. This deficiency means that the human microbial community gets "poor training" and hence hyper-responsiveness to bioparticles.

Biodiversity may affect the emergence and transmission of infectious diseases, mainly those vector-borne diseases (Zaghi et al. 2010; Jeffery and van der Putten, 2011; Sandifer et al., 2015; WHO, 2015). Biodiversity can be protective of human health through regulation of infectious diseases by two principal mechanisms: regulation of host pathogens population by direct predatory and competitive interactions and the reduction of pathogen success by the dilution effect (Keesing et al., 2010; Hough, 2014; Civitello et al., 2015; Aerts et al., 2018). The dilution effect predicts that infection rates among vectors, and ultimately human infection risk, will be lower in highly diverse host communities. This diversity inhibits the abundance of parasites through regulation of the population of susceptible hosts or interfering with the transmission process (Johnson and Thieltges, 2010; Kilpatrick et al., 2017). Diverse communities can inhibit the proliferation of parasites and promote ecological communities and ecosystem services (i.e., nutrient cycling or carbon sequestration; Johnson and Thieltges, 2010; Civitello et al., 2015).

28 PHENOTYPE – the EU project "Positive health effects on the natural outdoor environment in typical populations of different regions in Europe", http://www.phenotype.eu/en/

8 Conclusions and recommendations

The report analyzed the impact of various types of soil degradation on human health based on the available scientific literature, databases and project reports, and identified the gaps in knowledge and availability of data. The focus was on the impact of soil pollution, erosion, sealing, flooding, landslides and reduced soil biodiversity on human health.

The health consequences of human exposure to contamination are assessed usually by using risk assessment methods that compare measured or modelled human intake of specific chemicals with toxicologically based guideline values for pollutant doses. Usually, such analysis focusses on the assessment of the risk of one single chemical, resulting in a potential underestimation of the real risks because humans are in real life often exposed to multiple pollutants. Another approach involves conducting epidemiological studies to investigate the relationship between exposure and health directly within the affected population. The most frequently investigated health effects of soil pollution are mortality and hospital admissions/morbidity based on routinely collected statistics and hospitalisation records. Epidemiological data on cancer incidence are based on data gathered from national or regional cancer registries. Epidemiological studies often apply simplifications in exposure modelling, because not all pathways are known or can be adequately assessed. It is a huge challenge to deal with EP, which are identified in substantial amounts in the environment and due to their properties can exert adverse effects on human health. Only a limited amount of data is available on the epidemiological consequences of these compounds. Their fate and transport still remains largely unknown, and for the vast majority of EP, the potential ecological and health effects are not yet identified. Moreover, mixtures of EP and other contaminants enter the environment through different sources, but their combined effects have not yet been determined.

The available literature on the effect of soil sealing on human health or well-being is very limited. There are only a few scientific papers available that link the incidence of disease or death with heat extremes or air pollution, especially during hot summers in densely urbanised areas. There is currently a lack of methods and data to evaluate the direct effect of soil sealing intensity and spatial distribution on human health.

There are also few data and reports available addressing the effect of soil erosion on human health. Some scientific papers specifically focus on the long-distance transport of soil dust caused by wind erosion. These studies were conducted in the Mediterranean region, where residents were exposed to Saharan dust intrusion. The epidemiological conclusions are based on regression models, case-crossover design and time series analysis of daily mortality and morbidity. However, the health consequences of soil erosion are very difficult to assess due to the many influencing factors that occur simultaneously during dust intrusion events.

The assessment of the human health impact of natural disasters such as floods and landslides in Europe is particularly weak, although floods are the most common natural disaster worldwide. In Europe, researchers and decision makers tend to focus more on early warning systems, physical impact, risk analysis, infrastructure and population vulnerability, instead of studying the health effects. Some methodological limitations and difficulties (e.g. the lack of pre-flood data, control groups and clinical diagnosis of health effects) have been identified for epidemiological studies on flooding.

Data on the flood events and their health effects are reported in the EM-DAT (the International Disaster Database). A disaster is entered into this database when it fulfils specific criteria (e.g. 10 or more people reportedly killed, 100 people reportedly affected, a call for international assistance or declaration of a state of emergency), meaning that only large events are studied and small and local events tend to be ignored. There is a lack of good quality quantitative epidemiological data. Most of the available data result from retrospective analyses based on medical records. Access to such data varies from country to country and is strongly related to the structure of the health system and privacy regulations.

More research is needed on the relationship between soil biodiversity and human health. The majority of research focuses on single pathogens and their interaction with host organisms and the correlation with diseases. Most studies on the link between biodiversity and human health evaluate the short-term (positive) effects of exposure to biodiversity. Such research often lacks proper experimental design. To validate the long-term benefits of biodiversity on human health, there is a need for longitudinal studies, such as birth cohort studies, rather than cross-sectional research.

Recommendations for research:

- There is a need for multidisciplinary research that involves scientists and experts from various disciplines such as soil science, agronomy, biology, geology, medicine, toxicology and epidemiology, and that uses holistic system thinking (soil-sediment-water-atmosphere).
- Sufficient project funding should be made available to better understand the impact of soil degradation on human health.
- Additional harmonisation and improvement of risk assessment procedures and models is necessary. Multi-pathway exposure, the impact of various pollutant mixtures and bioavailability/bioaccessibility methods should be integrated better in both ecological and human risk assessment.
- Knowledge on fate, occurrence and hazardous effects of emerging pollutants and their interactions with other chemicals and abiotic factors should be improved. Future research on emerging pollutants should consider the entire lifecycle of the pollutants, from the source of emission to their removal from the environment through treatment and remediation techniques, as well as the impacts and risks they may pose to human health.
- More research is needed on the magnitude and current effects of diffuse contamination on human health, including pesticides and fertilizers. The environmental risks of the circular economy should be better investigated (e.g. waste-derived fertilisers, soil improvers, sewage sludge, nutrient recycling, etc.).
- More epidemiological case studies with quantitative data are needed to assess the long-term effects of various types of soil degradation on human health. Most of the reports now focus on the impact on human health from soil contamination. There is a need to investigate also the magnitude of complex effects of soil erosion, sealing and biodiversity loss on human health. Epidemiological studies should cover ideally different regions across the EU and take into account all the factors that potentially influence the human health risk.

Recommendations for preventing soil degradation and reducing health risk:

- Limit and better control the application of fertilisers and pesticides in agriculture, prevent and reduce pollutant emissions, adopt non-polluting technologies for production and transportation, implement sustainable soil management practices.
- Limit soil sealing through responsible spatial planning procedures, reduce land take and decrease the conversion of the most valuable soils, provide incentives for land recycling, promote green infrastructure, use permeable materials in the construction sector.
- The harmful effects of natural disasters (floods and landslides) on human health can be reduced by adopting stringent building codes and legislation to relocate infrastructure away from prone areas, improving flood forecasting, establishing early warning systems and insurance policies.

Recommendations for increasing awareness:

- Increase public awareness on the relation between the health of soils and human health through education, informational campaigns and communication with landowners. There is a need to convey clear and simple messages to relevant stakeholders. More emphasis should be placed on solutions, best practices and the benefits of soil.

9 References

Abdel-Shafy, H.I., Mansour, M.S.M., 2016. A review on polycyclic aromatic hydrocarbons: Source, environmental impact, effect on human health and remediation. Egyptian Journal of Petroleum, 25, 107–123. Doi:10.1016/j.ejpe.2015.03.011.

Aerts, R., Honnay O., Van Nieuwenhuyse, A., 2018. Biodiversity and human health: mechanisms and evidence of the positive health effects of diversity in nature and green spaces. British medical Bulletin, 127, 5-22.

Ahern, M., Kovats, R.S., Wilkinson, P., Few, R., Matties, F., 2005. Global Health Impacts of Floods: Epidemiological Evidence. Epidemiologic Reviews, 27, 36-46.

Aksoy, E., Louwagie, G., Gardi, C., Gregor, M., Schröder, C., Löhnertz, M., 2017. Assessing soil biodiversity potentials in Europe. Science of the Total Environment, 589, 236-249.

Alderman, K., Turner, L.R., Tong, S., 2012. Floods and human health: A systematic review. Environment International, 47, 37-47.

Alias, C., Benassi, L., Bertazzi, L., Sorlini, S., Volta, M., Gelati, U., 2019. Environmental exposure and health effects in a highly polluted area of Northern Italy: a narrative review. Environmental Science and Pollution Research, 26, 4555-4569.

Almeida, S., Raposo, A., Almeida-González, M., Carrascosa, C., 2018. Bisphenol A: food exposure and impact on human health. Comprehensive Reviews in Food Science and Food Safety, 17, 1503-1517. Doi:10.1111/1541-4337.12388.

Andersson-Sköld, Y., Nyberg, L., 2016. Effective and Sustainable Flood and Landslide Risk Reduction Measures: An Investigation of Two Assessment Frameworks. International Journal of Disasters Risk Science, 7, 374-392.

Augustsson, A., Uddh-Söderberg, T., Filipsson, M., Helmfrid, I., Berglund, M., Karlsson, H., Hogmalm, J., Karlsson, A., Alriksson, S., 2018. Challenges in assessing the health risks of consuming vegetables in metal contaminated environments. Environment International, 113, 269-280.

Becker, R., Bremser, W., Nehls, I., et. al., 2006. Mineral oil content in sediments and soils: Comparability, traceability and a certified reference material for quality assurance. Analytical and Bioanalytical Chemistry, 385, 645–651.10.1007/s00216-006-0423-4.

Bento, C.P.M., Goossens, D., Rezaei, M., Riksen, M., Mol, H.G.J., Ritsema, C.J., Geissen, V., 2017. Glyphosate ad AMPA distribution in wind-eroded sediment derived from loess soil. Environmental Pollution, 220, 1079-1089.

Berg, G., Eberl, L., Hartmann, A., 2005. The rhizosphere as a reservoir for opportunistic human pathogenic bacteria. Environmental Microbiology, 7:11, 1673-1685.

Biswas, B., Qi, F., Biswas, J.K., Wijayawardena, A., Khan, M.A.I., Naidu, R., 2018. The Fate of Chemical Pollutants with Soil Properties and Processes in the Climate Change Paradigm—A Review. Soil Systems, 2:51; doi:10.3390/soilsystems2030051, 1-20.

Bornman, M., Delport, R., Farias, P., Aneck-Hahn, N., Patrick, S., Millar, R.P., de Jager, C., 2018. Alterations in male reproductive hormones in relation to environmental DDT exposure. Environment International, 113, 281-289.

Bowler, D.E., Buyung-Ali, L., Knight, T.M., Pullin, A.S., 2010. Urban greening to cool towns and cities: A systematic review of the empirical evidence. Landscape and Urban Planning. 97, 147-155.

Breure, A.M., 2004. Soil biodiversity: measurements, indicators, threats and soil functions. In: Mat. of I International Conference "Soil and compost eco-biology", 15-17 September 2004, León, Spain, https://www.researchgate.net/publication/228751484.

Brevik, E.C., Burgess, L.C. (eds.), 2013. Soils and Human Health. Boca Raton: CRC Press.

Brevik, E.C., Sauer, T.J., 2015. The past, present, and future of soils and human health studies. Soil, 1, 35-46.

Brevik, E.C., Steffan, J.J., Rodrigo-Comino, J., Neubert, D., Burgess, L.C., Cerdà, A., 2018a. Connecting the public with soil to improve human health. European Journal of Soil Science, doi: 10.1111/ejss.12764.

Brevik, E.C., Pereg, L., Steffan, J.J., Burgess, L.C., 2018b. Soil ecosystem services and human health. Current Opinion in Environmental Science & Health, 5, 87-92.

Briggs, D., 2003. Environmental pollution and the global burden of disease. British Medical Bulletin, 68,1–24, DOI: 10.1093/bmb/ldg019.

Briggs, D.J., 2008. A framework for integrated environmental health impact assessment of systemic risks. Environmental Health, 7:61. DOI: 10.1186/1476-069X-7-61.

Cachada, A., Ferreira da Silva, E., Duarte, A.C., Pereira, R., 2016. Risk assessment of urban soils contamination: the particular case of polycyclic aromatic hydrocarbons. Science of the Total Environment, 551-552, 271-284.

Cakmak, I., Kutman, U.B., 2018. Agronomic biofortification of cereals with zinc: a review. European Journal of Soil Science, 69, 172-180.

Cao, Q., Steinman, A.D., Wan, X., Xie, L., 2018. Bioaccumulation of microcystin congeners in soil-plant system and human health risk assessment: A field study from Lake Taihu region of China. Environmental Pollution, 240, 44-50. Doi:10.1016/j.envpol.2018.04.067.

Carre, F., Caudeville, J., Bonnard, R., Bert, V., Boucard, P., 2015. Soil contamination and human health: A major challenge for global soil security. Global Soil Security Symposium, May 2015, College Station, United States. pp.275-295, 10.1007/978-3-319-43394-3_25. ineris- 01864711.

Cave, M.R., Wragg, J., Denys, S., Jondreville, C., Feidt, C., 2011. Oral bioavailability. In: In: Swartjes FA (ed) Dealing with contaminated sites. From theory towards practical application. Dortrecht: Springer publishers, 287-324.

Cessna, A.J., Larney, F.J., Kerr, L.A., Bullock, M.S., 2006. Transport of trifluralin on wind-eroded sediment. Canadian Journal of Soil Science, 86:3, 545-554.

Chen, Y.-Y., Kao, T.-W., Wang, C.-C., Chen, Y.-J., Wu, C-J., Ching-Huang Lai, C.-H., Wei-Liang Chen, W.-L., 2019. Exposure to polycyclic aromatic hydrocarbons and risk of disability among an elderly population. Environmental Science and Pollution Research, 26, 10719–10726.

Chenu, C., Stotzky, G., 2002. Interactions between microorganisms and soil particles: an overview. In: Interactions between soil particles and microorganisms, Huang P.M., Bollag J.-M., Senesi N. (eds), John Wiley&Sons, Ltd.

Civitello, D.J., Cohen, J., Fatima, H., Halstead, N.T., Liriano, J., McMahon, T.A., Ortega, C.N., Sauer, E.L., Sehgal, T., Young, S., Rohr, J.R., 2015. Biodiversity inhibits parasites: Broad evidence for the dilution effect. PNAS, 112:28, 8667-8671, doi:10.1073/pnas.1506279112.

Claessens, J., Schram-Bijkerk, D., Dirven-van Breemen, L., Otte, P., van Wijnen, H., 2014. The soil-water system as basis for a climate proof and healthy urban environment: Opportunities identified in a Dutch case-study. Science of The Total Environment, 485-486, 776-784.

Clark, N.E., Lovell, R., Wheeler, B.W., Higgins, S.L., Depledge, M.H., Norris, K., 2014. Biodiversity, cultural pathways, and human health: a framework. Trends in Ecology & Evolution, 29:4, 198-204.

Concawe, 2016. Environmental fate and effects of poly-and perfluoroalkyl substances (PFAS). Report no. 8/2016, pp 121.

COM 231, 2006. 23, communication from the commission to the council, the European Parliament, the European Economic and Social Committee and the Committee of the Regions. Thematic Strategy for Soil Protection, SEC (2006)620, SEC (2006)1165. European Commission, Bruxelles, pp. 12.

Comba, P., Ricci, P., Iavarone, I., Pirastu, R., Buzzoni, C., Fusco, M., Ferretti, S., Fazzo, L., Pasetto, R., Zona, A., Crocetti, E., 2014. Cancer incidence in Italian contaminated sites. Annali dell'Istituto Superiore di Sanità, 50: 2, 186-191.

Corrales, J., Kristofco, L.A., Baylor Steele, W., Yates, B.S., Breed, C.S. et. al., 2015. Global assessment of bisphenol A in the environment: review and analysis of its occurrence and bioaccumulation. Dose-Response: An International Journal July-September 2015, 1-29.10.1177/1559325815598308.

Cox D.T.C., D.F. Shanahan, H.L. Hudson, R.A. Fuller, K.J. Gaston. 2018. The impact of urbanisation on nature dose and the implications for human health. Landscape and Urban Planning 179: 72-80

Dadvand, P., Bartoll, X., Basagaña, X., Dalmau-Bueno, A., Martinez, D., Ambros, A., Cirach, M., Triguero-Mas, M., Gascon, M., Borrell, C., Nieuwenhuijsen, M.J., 2016. Green spaces and General Health: Roles of mental health status, social support, and physical activity. Environment International, 91, 161-167.

de Castro, A.P., Fernandez, G.R., Franco, O.L. 2014. Insights into novel antimicrobial compounds and antibiotic resistance genes from soil metagenomics. Frontiers in Microbiology, 5:489, doi: 10.3389/fmicb.2014.00489.

Depietri Y., Renaud, F.G., Kallis, G., 2012. Heat waves and floods in urban areas: a policy-oriented review of ecosystem services. Sustain Sci. 7: 95–107.

de Sario, M., Pasetto, R., Vecchi, S., Zeka, A., Hoek, A., Michlozzi, P., Iavarone, I., Fletcher, T., Bauleo, L., Ancona, C., 2018. A scoping review of the epidemiological methods used to investigate the health effects of industrially contaminates sites. Epidemiologia & Prevenzione, 42:5-6, Suppl 1, 59-68.

Diaz, J., Linares, C., Carmona, R., Russo, A., Ortiz, C., Salvador, P., Trigo, R.M., 2017. Saharan dust intrusions in Spain: Health impacts and associated synoptic conditions. Environmental Research, 156, 455-467.

Donato, F., Zani, C., Magoni, M., Gelatti, U., Covolo, L., Orizio, G., Speziani, F., Indelicato, A., Scarcella, C., Bergonzi, R., Apostoli, P., 2008. Polychlorinated biphenyls and thyroid hormone serum concentrations among people living in a highly polluted area: A cross-sectional population based study. Environmental Research, 108, 380-386.

Drobac, D.D., Tokodi, N., Simeunović, J., Baltić, V., Svirčev, Z., 2013. Human exposure to cyanotoxins and their effects on health. Archives of Industrial Hygiene and Toxicology, 64, 305-316. Doi:10.2478/10004-1254-64-2013-2320.

Du, W., FitzGerald, G.J., Clark, M., Hou, X.Y., 2010. Health impacts of flood. Prehospital and disaster Medicine, 25:3, 265-272.

European Commission — Directorate General for Regional Policy, 2011. Cities of tomorrow - Challenges, visions, ways forward Luxembourg: Publications Office of the European Union ISBN: 978-92-79-21307-6 doi:10.2776/41803.

European Commission, 2012. Guidelines on best practice to limit, mitigate or compensate soil sealing, SWD(2012) 101 final/2.

European Environmental Agency, EEA, 2010. Mapping the impacts of natural hazards and technological accidents in Europe. An overview of the last decade. Technical Report No 13, Publications Office of the European Union, Luxemburg, doi:10.2800/62638.

FAO and ITPS, 2015. Status of the World's Soil Resources (SWSR) – Main Report. Food and Agriculture Organization of the United Nations and Intergovernmental Technical Panel on Soils, Rome, Italy, pp. 648.

FAO, 2019. FAO Soil Portal. Soil degradation. Available at: http://www.fao.org/soils-portal/soil-degradation-restoration/en (11/07/2019).

Fernández-Navarro, P., García-Pérez, J., Ramis, R., Boldo, E., López-Abente, G., 2017. Industrial pollution and cancer in Spain: An important public health issue. Environmental Research, 159, 555-563.

Few, R., Ahern, M., Matthies, F., Kovats, S., 2004. Floods, health and climate change: a strategic review. Technical Report. Tyndall Centre for Climate Change Research, Working Paper No. 63, pp. 117.

Filleul, L., Cassadou, S., Médina, S., Fabres, P., Lefranc, A., Eilstein, D., Le Tertre, A., Pascal, L., Chardon, B., Blanchard, M., Declercq, C., Jusot, J.F., Prouvost, H., Ledrans, M., 2006. The relation between temperature, ozone, and mortality in nine French cities during the heat wave of 2003. Environ Health Perspective, 114(9): 1344-1347.

Fischer, P.H., Brunekreef, B., Lebret, E., 2004. Air pollution related deaths during the 2003 heat wave in the Netherlands. Atmospheric Environment 38(8): 1083-1085.

Flint, S., Markle, T., Thompson, S., Wallace, E., 2012. Bisphenol A exposure, effects, and policy: A wildlife perspective. Journal of Environmental Management, 104, 19-34. Doi:10.1016/j.jenvman.2012.03.021.

Gallego, J. L. R., Loredo, J., Llamas, J. F., Vázquez, F., Sánchez, J., 2001. Bioremediation of diesel-contaminated soils: Evaluation of potential in situ techniques by study of bacterial degradation. Biodegradation, 12, 325–335.

Gardi, C., Jeffery, S., Saltelli, A., 2013. An estimate of potential threats levels to soil biodiversity in EU. Global Change Biology, 19, 1538-1548.

Gavrilescu, M., Demnerowá, K., Aamand, J., Agathos, S., Fava, F., 2014. Emerging pollutants in the environment: present and future challenges in biomonitoring, ecological risks and bioremediation. New Biotechnology, http://dx.doi.org/10.1016/j.nbt.2014.01.001.

Gebreyesus, S.T., 2014. Heavy metals in contaminated soil: sources & washing through chemical extractants. American Scientific Research Journal for Engineering, Technology, and Sciences (ASRJETS), 10, 54-60.

Geertsema, M., Highland, L.M., 2011. Landslides: Human Health Effects. In: Encyclopedia of Environmental Health, Second edition, 380-395. Doi: 10.1016/B978-0-444-52272-6.00550-X.

Geissen, V., Mol, H., Klimpp, E., Umlauf, G., Nadal, M., van der Ploeg, M., van de Zee, S.E.A.T.M., Ritsema, C.J., 2015. Emerging pollutants in the environment: a challenge for water resource management. International Soil and Water Conservation Research, 3, 57-65.

Ginoux, P., Prospero, J. M., Gill, T. E., Hsu, N. C., Zhao, M., 2012. Global-scale attribution of anthropogenic and natural dust sources and their emission rates based on MODIS Deep Blue aerosol products. Reviews of Geophysics, 50, RG3005, doi:10.1029/2012RG000388, 1-36.

Godambe, T., Fulekar, M., 2017. Bioremediation of petrochemical hydrocarbons (BTEX) – Review. Journal of Environmental Science and Pollution Research, 3, 189–199.

Goudie, A.S., 2014. Desert dust and human health disorders. Environment International, 63, 101-113.

Grenni, P., Ancona, V., Caracciola, A.B., 2018. Ecological effects of antibiotics on natural ecosystems: A review. Microchemical Journal, 136, 25-39.

Griffin, D.W., Kellogg, C.A., Shinn, E.A., 2001. Dust in the wind: Long range transport of dust in the atmosphere and its implications for global public and ecosystem health. Global Change & Human Health, 2:1, 20-33.

Grineski, S.E., Staniswalis, J.G., Bulathsinhala, P., Peng, Y., Gill, T.E., 2011. Hospital admissions for asthma and acute bronchitis in El Paso, Texas: Do age, sex, and insurance status modify the effects of dust and low wind events? Environmental Research, 111, 1148-1155.

Grunewald, K., Li, J., Xie, G., Kűmper-Schlake, L., 2018. Towards green cities. Urban biodiversity and ecosystem services in China and Germany. Springer International Publishing AG, pp.212.

Gunawardena, K.R., Wells, M.J., Kershaw, T., 2017. Utilising green and bluespace to mitigate urban heat island intensity. Science of the Total Environment, 584-585, 1040-1055.

Guzman, J.A., Fox, G.A., Penn, C.J., 2012. Sorption of Escherichia coli in agricultural soils influenced by swine manure constituents. American Society of Agricultural and Biological Engineers, 55:1, 61-71.

Guzzetti, F., Stark, C.P., Salvati, P., 2005. Evaluation of flood and landslide risk to the population of Italy. Environmental Management, 36:1, 15-36.

Gwenzi, W., Mangori, L., Danha, C., Chaukura, N., Dunjana, N., Sanganyado, E., 2018. Sources, behaviour, and environmental and human health risks of high technology rare earth elements as emerging contaminants. Science of the Total Environment, 636, 299-313.

Hajat, S., Ebi, K.L., Kovats, R.S., Menne, B., Edwards, S., Haines, A., 2005. The Human Health Consequences of Flooding in Europe: a Review. In: Extreme Weather Events and Public Health Responses, Springer-Verlag Berlin Hiedelberg, 185-196.

Haque, U., Blum, P., da Silva, P.F., Andersen, P., Pilz, J., Chalov, S.R., Malet, J.-P., Auflič M. J., Andres, N., Poyiadji, E., Lamas, P.C., Zhang, W., Peshevski, I., Pétursson, H.G., Kurt, T., Dobrev, N., García-Davalillo, J.C., Halkia, M., Ferri, S., Gaprindashvili, G., Engström, J., Keellings, D., 2016. Fatal landslides in Europe. Landslides, 1-11, DOI 10.1007/s10346-016-0689-3.

Hashmi, M.Z., Strezov, V., Varma, A., (eds), 2017. Antibiotics and antibiotics resistance genes in soils. Monitoring, toxicity, risk assessment and management. Springer International Publishing AG, pp. 436.

Henríquez-Hernández, L.A., Luzardo, O.P., Almeida-González, M., Álvarez-León, E.E., Serra-Majem, L., Zumbado, M., Boada, L.D., 2011. Background level of polychlorinated biphenyls in the population of the Canary Islands (Spain). Environmental Research, 111, 10-16.

Ho, H.C., Lau, K.K.-L., Ren, C., Ng, E., 2017. Characterizing prolonged heat effects on mortality in a sub-tropical high-density city, Hong Kong. International Jouirnal of Biometeorology, 61, 1935-1944.

Hoek, G., Ranzi, A., Alimehmeti, I., Ardeleau, E.-R., Arrebola, J.P., Avila, P., et al. 2018. A review of exposure assessment methods for epidemiological studies of health effects related to industrially contaminated site. Epidemiologia & Prevenzione, 42:5-6, Suppl 1, 21-36.

Hosford, M., 2009. Human health toxicological assessment of contaminants in soil: using science to create better place. Bristol, Environment Agency of Great Britain, pp. 156.

Hough, R.L., 2007. Soil and human health: an epidemiological review. European Journal of Soil Science, 58, 1200-1212.

Hough, R.L., 2014. Biodiversity and human health: evidence for causality? Biodiversity conservation, 23, 267-288.

Huynen, M.M.T.E., Martens, P., De Groot, R.S., 2004. Linkages between biodiversity loss and human health: a global indicator analysis. International Journal of Environmental Health Research, 14:1, 13-30.

Jakubicka, T., Vos, F., Phalkey, R., Marx, M., Guha-Sapir, D., 2010. Health impacts of floods in Europe: Data gaps and information needs from a spatial perspective. Centre for Research on the Epidemiology and Disasters – CRED, Universitätsklinikum Heidelberg, pp. 43.

Jeffery, S., Gardi, C., Jones, A., Montanarella, L., Marmo, L., Miko, L., Ritz, K., Peres, G., Römbke, J., van der Putten, W.H., 2010. European Atlas of Soil Biodiversity, European Commission, Publications Office of the European Union, Luxemburg, pp. 128.

Jeffery, S., van der Putten, W.H., 2011. Soil borne human diseases. Joint Research Centre Scientific and Technical Reports, Institute for Environment and Sustainability, European Commission, EUR 24893 EN, 1-56.

Jiménez, E., Linares, C., Martínez, D., Díaz, J., 2010. Role of Saharan dust in the relationship between particulate matter and short-term daily mortality among the elderly in Madrid (Spain). Science of the Total Environment, 408, 5729-5736.

Johnson, P.T.J., Thieltges, D.W., 2010. Diversity, decoys and the dilution effect: how ecological communities affect disease risk. 213, 961-970.

Juhasz, A.L., Weber, J., Smith, E., 2011. Impact of soil particle size and bioaccessibility on children and adult lead exposure in peri-urban contaminated soil. Journal of Hazardous Materials. 186, 1870-1879.

Karanasiou, A., Moreno, N., Moreno, T., Viana, M., de Leeuw, F., Querol, X., 2012. Health effects from Sahara dust episodes in Europe: Literature review and research gaps. Environmental International, 47, 107-114.

Keesing, F., Belden, L.K., Daszak, P., Dobson, A., Harvell, C.D., Holt, R.D., Hudson, P., Jolles, A., Jones, K.E., Mitchell, C.E., Myers, S.S., Bogich, T., Ostfeld, R.S., 2010. Impact of biodiversity on the emergence and transmission of infectious diseases. Nature, 468, 647-652.

Keesstra, S.D., Bouma, J., Wallinga, J., Tittonell, P., Smith, P., Cerdà, A., Montanarella, L., Quinton, J.N., Pachepsky, Y., van der Putten, W.H., Bardgett, R.D., Moolenaar, S., Mol, G., Jansen, B., Fresco, L.O., 2016. The significance of soils and soil science towards realization of the United Nations Sustainable Development Goals, SOIL, 2, 111-128, https://doi.org/10.5194/soil-2-111-2016.

Kennedy, I.T.R., Petley, D.N., Williams, R., Murray, V.A., 2015. A systematic review of the health impacts of mass earth movements (landslides). PLOS Currents Disasters. 2015 Apr 30. Edition 1. doi: 10.1371/currents.dis.1d49e84c8bbe678b0e70cf7fc35d0b77.

Keshishian, C., 2014. Using routine health data for surveillance of the health effects of floods. Public Health England, 249, London, pp. 62.

Khan, N.T., Jamee, I.N., Khan, M.J., 2018. A Brief Overview of Contaminated Soil Remediation Methods. BioTechnology: An Indian Journal, 14, 168.

Kilpatrick, A.M., Salkeld, D.J., Titcomb, G., Hahn, M.B., 2017. Conservation of biodiversity as a strategy for improving human health and well-being. Philosophical Translations Royal Society B, 372, 20160131, http://dx.doi.org/10.1098/rstb.2016.0131.

Kogevinas, M., 2011. Epidemiological approaches in the investigation of environmental causes of cancer: the case of dioxins and water disinfection by-products. Environmental Health, 10(Suppl 1):S3, http://www.ehjournal.net/content/10/S1/S3.

Kraemer, F.B., Chagas, C.I., Morras, H.J.M., Moretto, J., Paz, M., Garibaldi, L.A., 2013. Influence of physical and chemical soil properties on the adsorption of Escherichia coli in Mollisols and Alfisols of Argentina. Revista Internacional de Contaminacion Ambiental, 29:1, 7-20.

Lado, L.R., Hengl, T., Reuter, H.I., 2008. Heavy metals in European soils: A geostatistical analysis of the FOREGS Geochemical database. Geoderma,14, 8, 189–199. Doi:10.1016/j.geoderma.2008.09.020.

Lal, R., 2009. Soil degradation as a reason for inadequate human nutrition. Food Security, 1, 45-57.

Lal, R., 2015. Restoring soil quality to mitigate soil degradation. Sustainability, 7, 5875-5895.

Lamothe, D.-N., Neveu, G., Görlach, B., Interwies, E., 2005. Evaluation of the impact of floods and associated protection policies. DG Env. Report, pp. 108.

Landrigan, P.J., Fuller, R., Acosta, N.J.R., Adeyi, O., Arnold, R., Basu, N., Baldé, A. B., Bertollini, R., Bose-O'Reilly, S., Boufford, J.I., Breysse, P. N., Chiles, T., Mahidol, C., Coll-Seck, A.M., Cropper, M.L., Fobil, J., Fuster, V., Greenstone, M., Haines, A., Hanrahan, D., Hunter, D., Khare, M., Krupnick, A., Lanphear, B., Lohani, B., Martin, K., Mathiasen, K.V., McTeer, M.A., Murray, C.J.L., Ndahimananjara, J.D., Perera, F., Potočnik, J., Preker, A.S., Ramesh, J., Rockström, J., Salinas, C., Samson, L.D., Sandilya, K., Sly, P.D., Smith, K.R., Steiner, A., Stewart, R.B., Suk, W.A., van Schayck, O.C. P., Yadama, G.N., Yumkella, K., Zhong, M., 2018. The Lancet Commission on pollution and health. Lancet, 391, 462-512.

Langley, A., 2011. Hazard assessment and contaminated sites. In: Swartjes FA (ed) Dealing with contaminated sites. From theory towards practical application. Dortrecht: Springer publishers, 517-570.

Lei, M., Zhang, L., Lei, J., Zong, L., Li, J. et. al., 2015. Overview of emerging contaminants and associated human health effects. BioMed Research International, Volume 2015, Article ID 404796, 12 pages. 10.1155/2015/404796.

Li, G., Sun, G.-X., Ren, Y., Luo, X.-S., Zhu, Y.-G., 2018. Urban soil and human health: a review. European Journal of Soil Science, 69, 196-215.

Li, F., Li, Z., Mao, P., Li, Y., Li, Y., McBride, M.B., Wu, J., Zhuang, P., 2019. Heavy metal availability, bioaccessibility, and leachability in contaminated soil: effects of pig manure and earthworms. Environmental Science and Pollution Research, 26, 20030-20039.

Liddicoat, C., Waycott, M., Weinstein. P., 2016. Environmental change and human health: Can environmental proxies inform the biodiversity hypothesis for protective microbial-human contact? BioScience, 66:12, 1023-1034.

LiPuma, J.J., Spilker, T., Coenye, T., Gonzalez, C.F., 2002. An epidemic *Burkholderia cepacia* complex strain identified in soil. Lancet, 359, 2002-2003.

Liu, L., Li, W., Song, W., Guo, M., 2018. Remediation techniques for heavy metal-contaminated soils: Principles and applicability. Science of the Total Environment, 633, 206–219. doi:10.1016/j.scitotenv.2018.03.161.

Maifredi, G., Donato, F., Magoni, M., Orizio, G., Gelatti, U., Maiolino, P., Zani, C., Vassallo, F., Scarcella, C., 2011. Polychlorinated biphenyls and non-Hodgkin's lymphoma: A case–control study in Northern Italy. Environmental Research, 111, 254-259.

Maring, L., 2019. Policy brief "Soil quality in spatial planning" SOILS4EU deliverable 1.6.

Marra M.L., Zullo F., De Felice B., Nappi L., Guida M., Trifuoggi M., Nappi C.,Di Spiezio Sardo A.., Zizolfi B., Capece G., Visconti F.,Troisi J., Ciccone C., Guida M., 2012. Environmental Pollution Effects on Reproductive Health – Clinical-Epidemiological Study In Southern Italy. Translational Medicine, 4:5,39-56.

Martin-Olmendo, P., Hams, R., Santoro, M., Ranzi, A., Hoek, G., de Hoogh, K., Leonardi, G.S., 2018. Environmental and health data needed to developed national surveillance systems in industrially contaminated sites. Epidemiologia & Prevenzione, 42:5-6, Suppl 1, 11-20.

Menne, B., Murray, V., editors, 2013. Floods in the WHO European Region: health effects and their prevention. Copenhagen: WHO Regional Office for Europe (http://www.euro.who.int/ floods-in-the-who-european-region), pp. 146.

Middleton, D.R.S., McComack, V.A., Watts. M.J., Schűz, J., 2019. Environmental geochemistry and cancer: a pertinent global health problem requiring interdisciplinary collaboration. Environmental Geochemistry and Health, doi:10.1007/s10653-019-00303-9.

Middleton, N.J., 2017. Desert dust hazards: a global review. Aeolian Research, 24, 53-63.

Middleton, N., Kang, U., 2017. Sand and dust storms: impact mitigation. Sustainability, 9, 1053, doi:10.3390/su9061053.

Morman, S.A., Plumlee, G.S., 2013. The role of airborne mineral dust in human disease. Aeolian Research, 9, 203-212.

Morris, J., Tassone, V., de Groot, R., Camilleri, M., Moncada, S., 2011. A framework for participatory impact assessment: involving stakeholders in European policy making, a case study of land use change in Malta. Ecology and Society 16(1): 12. [Online] URL: http://www.ecologyandsociety.org/vol16/iss1/art12/.

Mullany, P., 2014. Functional metagenomics for the investigation of antibiotic resistance. Virulence, 5:3, 443-447.

Naidu, R., Semple, K.T., Megharaj, M., Juhasz, A.L., Bolan, N.S., Gupta, S.K., Cloher, B.E., Schulin, R., 2008. Bioavailability: definition, assessment and implications for risk assessment. In: Chemical bioavailability in terrestrial environments. Naidu R. (eds), Elsevier, Developments in soil science, vol. 32, 39-52.

Naidu, R., Espan, V.A.A., Liu, Y., Jit, J., 2016. Emerging contaminants in the environment: Risk-based analysis for better management. Chemosphere, 154, 350-357.

Naveed, M., Moldrup, P., Arthur, E., Holstrup, M., Nicolaisen, M., Tuller, M., Herath, L., Hamamoto, S., Kawamoto, K., Komatdu, T., Vogel, H.-J., de Jonge, L.W., 2014. Simultaneous loss of soil biodiversity and functions along a copper contamination gradient: when soil goes to sleep. Soil Science Society of America Journal, 78, 1239-1250.

Nesme, J., Simonet, P., 2015. The soil resistome: a critical review on antibiotic resistance origins, ecology and dissemination potential; of telluric bacteria. Environmental Microbiology, 17:14, 913-930.

Nieuwenhuijsen, M.J., Kruize, H., Gidlow, C., et al. 2014. Positive health effects of the natural outdoor environment in typical populations in different regions in Europe (PHENOTYPE): a study programme protocol. BMJ Open, 4:e004951, doi:10.1136/bmjopen-2014-004951.

Olexsey, R.A. and Parker, R.A., 2006. Current and future in situ treatment techniques for the remediation of hazardous substances in soil, sediments, and groundwater. In: Twardowska, I., Allen, H.E., Häggblom, M.M., Stefaniak, S., (Eds.), Soil and Water Pollution Monitoring, Protection and Remediation, NATO Science Series, vol. 69, Springer, Dordrecht, The Netherlands (2006), pp. 211-219

Oliver, M.A., Gregory, P.J., 2015. Soil, food security and human health: a review. European Journal of Soil Science, 66, 257-276.

Orgiazzi, A., Panagos, P., Yigini, Y., Dunbar, M.B., Gardi, C., Montanarella, L., Ballabio, C., 2016. A knowledge-based approach to estimating the magnitude and spatial patterns of potential threats to soil biodiversity. Science of the Total Environment, 545–546, 11–20.

Panagos, P., Bouraoui, F., Gardi, C., Jeffery, S. et.al., 2012. The State of Soil in Europe. A Contribution of the JRC to the European Environment Agency's Environment State and Outlook.

Panagos, P., Van Liedekerke, M., Yigini, Y., Montanarella, L., 2013. Contaminated Sites in Europe: Review of the Current Situation Based on Data Collected through a European Network. Journal of Environmental and Public Health, Volume 2013, Article ID 158764, 11 pages. Doi:10.1155/2013/158764.

Pasetto, R., Martin-Olmedo, P., Martuzzi, M., Iavarone, I., 2016. Exploring available options in characterising the health impact of industrially contaminated sites. Annali Dell Istituto Superiore Di Sanita, 52:4, DOI: 10.4415/ANN_16_04_, 1-7.

Pepper, I.L., 2013. The Soil Health-Human Health Nexus. Critical Reviews in Environmental Science and Technology, 43:24, 2617-2652.

Pepper, I.L., Gerba, C.P., Newby, D.T., Rice C.W., 2009. Soil: A Public Health Threat or Saviour? Critical Reviews in Environmental Science and Technology, 39:5, 416-432.

Perez, L., Tobias, A., Querol, X., Kűnzli, N., Pey, J., Alastuey, A., Viana, M., Valero, N., González-Cabré, M., Sunyer, J., 2008. Coarse particles from Saharan dust and daily mortality. Epidemiology, 19, 800-807.

Pérez, A.P., Peláez Sánchez, S., Van Liedekerke, M., 2015. Remediated sites and brownfields. Success stories in Europe. A report of the European Information and Observation Network’s National Reference Centres for Soil (Eionet NRC Soil), doi: 10.13140/RG.2.1.3466.0882.

Pérez, A., Rodriguez-Eugenio, N., 2018. JRC Technical Report. Status of local soil contamination in Europe.

Pérez, R., Suelves, T., Molina, Y., Corpas-Burgos, F., Yusà, V., 2019. Biomonitoring of mercury in hair of children living in the Valencian Region (Spain). Exposure and risk assessment. Chemospere, 217, 558-566.

Petraviciene, I., Grazuleviciene, R., Andrusaityte, S., Dedele, A., Nieuwenhuijsen, M.J., 2018. Impact of the social and natural environment on preschool-age children weight. International Journal of Environmental Research and Public Health, 15, 449; doi:10.3390/ijerph15030449

Picó, Y., Alvarez-Ruiz, R., Alfarhan, A.H., El-Sheikh, M., Alobaid, S.M. et. al., 2019. Uptake and accumulation of emerging contaminants in soil and plant treated with wastewater under real-world environmental conditions in the Al Hayer area (Saudi Arabia). Science of the Total Environment, 652, 562–572.

Pimentel, D., 2006. Soil erosion: a food and environmental threat. Environment, Development and Sustainability, 8, 119-137.

Pimentel, D., Burgess, M., 2013. Soil erosion threatens food production. Agriculture, 3, 443-463.

Pirastu, R., Pasetto, R., Zona, A., Ancona, C., Iavarone, I., Martuzzi, M., Comba, P., 2013. The health profile of populations living in contaminated sites: Sentieri approach. Journal of Environmental and Public Health, Hindawi Publishing Corporation, Article ID 939267, pp.13.

Pistocchi, A., Calzolari, C., Malucelli, F., Ungaro, F., 2015. Soil sealing and flood risks in the plains of Emilia-Romagna, Italy. Journal of Hydrology: Regional Studies, 4, 398-409.

Piver, W.T., Ando, M., Ye, F., Portier, C.J., 1999. Temperature and air pollution as risk factors for heat stroke in Tokyo, July–August 1980–1995. Environ Health Perspective, 107: 911–916

Prokop, G., Jobstmann, H., Schönbauer, A., 2011. Technical Report 2011. Overview of best practices for limiting soil sealing or mitigating its effects in EU-27. 50 European Communities, Brussels.

Qu, C., Albanese, S., Li, J., Cicchella, D., Zuzolo, D., Hope, D., Cerino, P., Pizzolante, A., Doherty, A.L., Lima, A., De Vivi, B., 2019. Organochlorine pesticides in the soils from Benevento provincial territory, southern Italy: Spatial distribution, air-soil exchange, and implications for environmental health. Science of the Total Environment, 674, 159-170.

Riding, M.J., Doick, K.J., Martin, F.L., Jones, K.C., Semple, K.T., 2013. Chemical measures of bioavailability/bioaccessibility of PAHs in soil: Fundamentals to application. Journal of Hazardous Materials, 261, 687-700.

Rodgers, K., Udesky, J.O., Rudel, R.A., Brody, J.G., 2018. Environmental chemicals and breast cancer: An updated review of epidemiological literature informed by biological mechanisms, Environmental Research, 160, 152-182.

Rodríguez-Eugenio, N., McLaughlin, M., Pennock, D., 2018. Soil Pollution: a hidden reality. Rome, FAO. pp. 142.

Rostami, I., Juhasz, A.L., 2011. Assessment of persistent organic pollutant (POP) bioavailability and bioaccessibility for human health exposure assessment: a critical review. Critical Reviews in Environmental Science and Technology, 41, 623-656.

Ruijsbroek, A., Mohnen, S.M., Droomers, M., Kruize, H., Gidlow, C., Regina Gražulevičiene, R., Andrusaityte, S., Maas, J., Nieuwenhuijsen, M.J., Triguero‐Mas, M., Masterson, D., Ellis, N., van Kempen, E., Hardyns, W., Stronks, K., Groenewegen, P.P., 2017. Neighborhood green space, social environment and mental health: an examination in four European cities. International Journal of Public Health, 62, 657-667.

Ruszkiewicz, J.A., Pinkas, A., Ferrer, B., Peres, T., Tsatsakis, A. et.al., 2017. Neurotoxic effect of active ingredients in sunscreen products, a contemporary review. Toxicology Reports, 4, 245–259. Doi:10.1016/j.toxrep.2017.05.006.

Sandifer, P.A., Sutton-Grier, A.E., Ward, B.P., 2015. Exploring connections among nature, biodiversity, ecosystem services, and human health and well-being: Opportunities to enhance health and biodiversity conservation. Ecosystem Services. 12, 1-15.

Santonen, T., et al., 2017. Human biomonitoring in risk assessment: analysis of the current practice and 1st examples on the use of HBM in risk assessments of HBM4EU priority chemicals. Deliverable Report D 5.1 HORIZON2020 Programme, Contract No. 733032 HBM4EU, pp123.

Sarwar, N., Imran, M., Shaheen, M.R., Ishaque, W., Kamran, M.A., Matloob, A., Rehim, A., Hussain, S., 2017. Phytoremediation strategies for soil contaminated with heavy metals: modifications and future perspective. Chemosphere, 171, 710-721.

Sasidharan, S., Torkzaban, S., Bradford S.A., Kookaa, R., Page, D., Cook, P.G., 2016. Transport and retention of bacteria and viruses in biochar amended sand. Science of the Total Environment, 548-549, 100-109.

Schenker, M., 2000. Exposure and health effects from inorganic agricultural dusts. Environmental Health Perspectives, 108, sup. 4, 661-664.

Schmieder, R., Edwards, R., 2012. Insights into antibiotic resistance through metagenomics approaches. Future Microbiology, 7:1, 73-89.

Schram-Bijkerk, D., Otte, P., Dirven, L., Breure, A.M., 2018. Indicators to support healthy urban gardening in urban management. Science of The Total Environment, 621, 863-871.

Science Communication Unit, University of the West of England, 2013. Science for Environment Policy In-depth Report: Soil Contamination: Impacts on Human Health. Report produced for the European Commission DG Environment, September 2013. Available at: http://ec.europa.eu/science-environment-policy.

Shaddick, G., Ranz, A., Thomas, M.L., Aguirre-Perez, R., Bekker-Nielsen Dunbar, M., Parmagnani, F., Martuzzi, M., 2018. Towards an assessment of the health impact of industrially contaminated sites: waste landfills in Europe. Epidemiologia a Prevenzione, 42:5-6, Suppl 1, 69-75. doi: 10.19191/EP18.5-6.S1.P069.089

Siddique, S., Kubwabo, C., Harris, S., 2016. A review of the role of emerging environmental contaminants in the development of breast cancer in women. Emerging contaminants, 2, 204-219.

Siebielec, G., Prokop, G., van Delden, H., Verzandvoort, S., Miturski, T., Łopatka, A., 2015. Soil sealing. In: Stolte J. et al. Soil threats in Europe. Status, methods, drivers and effects on ecosystem services. JRC Technical Reports. European Union, ISBN 978-92-79-54018-9, s. 79-91.

Siebielec, G., Stuczyński, T., Lopatka, A., Czaban, P., 2011. Stakeholder network for impact assessment of soil protection scenarios. Urban SMS project: Deliverable number: 6.3.1, pp.24.

Siebielec, G., Hagemann, N., Klimkowicz-Pawlas, A., 2019. Policy brief „Sustainable soil management" SOILS4EU deliverable 1.3.

Sing, D., Sing, C.F., 2010. Impact of direct soil exposure from airborne dust and geophagy on human health. International Journal of Environmental Research and Public Health, 7, 1205-1223.

Snow, D.D., Cassada, D.A., Larsen, M., Mware, N.A., Li, X. et. al., 2017. Detection, Occurrence and Fate of Emerging Contaminants in Agricultural Environments. Water Environ Res., 89, 897–920. Doi:10.2175/106143017X15023776270160.

Stafoggia, M., Zauli-Sajani, S., Pey, J., Samoli, E., Alessandrini, E., Basagaña, X., Cernigliaro, A., Chiusolo, M., Demaria, M., Díaz, J., Faustini, A., Katsouyanni, K., Kelessis, A.G., Linares, C., Marchesi, S., Medina, S., Pandolfi, P., Pérez, N., Querol, X., Randi, G., Ranzi, A., Tobias, A., Forastiere, F., and the MED-PARTICLES Study Group, 2016. Desert dust outbreaks in Southern Europe: Contribution to daily OM10 concentrations and short-term associations with mortality and hospital admissions. Environmental Health Perspective, 124:4, 413-419.

Staples, C.A., Dorn, P.B., Klecka, G.M., O'Block, S.T., Harris, L.R., 1998. A review of the environmental fate, effects, and exposures of bisphenol-A. Chemosphere, 36, 2149-2173.

Steffan, J.J., Brevik, E.C., Burgess, L.C., Cerdà, A., 2018. The effect of soil on human health: an overview. European Journal of Soil Science, 69, 159-171.

Steinnes, E., 2009. Soils and geomedicine. Environmental Geochemistry and Health. 31, 523-535.

Stephenson, J., Vaganay, M., Cameron, R., Joseph, P., 2014. The long-term health impacts of repeated flood events. Flood recovery, innovation and response IV. In: WIT Transactions on Ecology and The Environment, 184, 201-212.

Sterk, G., Goossens, D., 2007. Emissions of soil dust and related problems in Europe: an overview. https://www.researchgate.net/publication/228482022.

Stolte, J., Tesfai, M., Øygarden, L., Kværnø, S., Keizer, J., Verheijen, F., Panagos, P., Ballabio, C., Hessel, R., 2016. Soil threats in Europe: status, methods, drivers and effects on ecosystem services. A review report, deliverable 2.1 of the RECARE project EUR 27607 EN. JRC Technical Reports 206 pp.

Stone, D., Ritz, K., Griffiths, B.G., Orgiazzi, A., Creamer, R.E., 2016. Selection of biological indicators appropriate for European soil monitoring. Applied Soil Ecology, 97, 12-22.

Swartjes, F.A., 2011. Introduction to contaminated site management. In: Swartjes FA (ed) Dealing with contaminated sites. From theory towards practical application. Dortrecht: Springer publishers, 3-87.

Swartjes, F.A, Cornelis, C., 2011. Human Health Risk Assessment. In: Swartjes FA (ed) Dealing with contaminated sites. From theory towards practical application. Dortrecht: Springer publishers, 209-259.

Swartjes, F.A., 2015. Human health risk assessment related to contaminated land: state of the art. Environmental Geochemistry and Health, 37, 651-673.

Swartjes, F.A., Rutgers, M., Lijzen, J.P.A., Jassen, P.J.C.M., Otte, P.F., Wintersen, A., Brand, E., Posthuma, L., 2012. State of the art of contaminated site management in The Netherlands: Policy framework and risk assessment tools. Science of the Total Environment, 427-428, 1-10.

Tasnim, N., Abulizi, N., Pither, J., Hart, M.M., Gibson, D.L., 2017. Linking the Gut Microbial Ecosystem with the Environment: Does Gut Health Depend on Where We Live? Frontiers in Microbiology, 8, 1935, doi: 10.3389/fmicb.2017.01935.

Thiele-Bruhn, S., 2003. Pharmaceutical antibiotic compounds in spoil – a review. Journal of Plant Nutrition and Soil Science, 166, 145-167.

Tóth, G., Hermann, T., Szatmári, G., Pásztor L., 2016. Maps of heavy metals in the soils of the European Union and proposed priority areas for detailed assessment. Science of the Total Environment, 565, 1054–1062, doi: 10.1016/j.scitotenv.2016.05.115.

Tunstall, S., Tapsell, S., Green, C., Floyd, P., George, C., 2006. The health effects of flooding: social research results from England and Wales. Journal of Water and Health, 04.3, 365-380.

Umeh, A.C., Duan, L., Naidu, R., Semple, K.T. 2017. Residual hydrophobic organic contaminants in soil: Are they a barrier to risk-based approaches for managing contaminated land? Environment International, 98, 18-34.

Valentin, L., Nousiainen, A., Mikkonen, A., 2013. Introduction to organic contaminants in soil: concept and risks. In: Vicent, T. et al. (eds.), Emerging Organic Contaminants in Sludges: Analysis, Fate and Biological Treatment, Springer-Verlag Berlin Heidelberg, Hdb Env Chem, 24, 1–30, DOI 10.1007/698_2012_208.

van Wezel, A.P., Franken, R.O.G., Drissen, E., Versluijs, K.C.W., van den Berg, R., 2008. Societal cost-benefits analysis for soil remediation in the Netherlands. Integrated Environmental Assessment and Management, 4:1, 61-74.

Wagg, C, Bender, S.F., Widmer, F., van der Heijden, M.G.A., 2014. Soil biodiversity and soil community composition determine ecosystem multifunctionality. PNAS, 111:14, 5266-5270, doi:10.1073/pnas.1320054111.

Waite, T.D., Chaintarli, K., Beck, C.R., Bone, A., Amlot, R., Kovats, S., Reacher, M., Armstrong, B., Leonardi, G., Rubin, G.J., Oliver, I., 2017. The English national cohort study of flooding ad health: cross-sectional analysis of mental health outcomes et year one. BMC Public Health, 17:129, 1-9.

Wall, D.H., Nielsen, U.N., Six, J., 2015. Soil biodiversity and human health. Nature, 528, 69-76.

WHO IPCS, 2002. Global assessment of the state-of-the-science of endocrine disruptors. Geneva, Switzerland, World Health Organization, International Programme on Chemical Safety.

Winkens, K., Vestergren, R., Berger, U., Cousins, I.T., 2017. Early life exposure to per- and polyfluoroalkyl substances (PFASs): A critical review. Emerging Contaminants, 3, 55-68.

World Health Organization (WHO), 2013a. Contaminated sites and health. Report of two WHO workshops: Syracuse, Italy, 18 November 2011, Catania, Italy, 21-22 June 2012.

World Health Organization (WHO) 2013b. State of the science of endocrine disrupting chemicals – 2012. Edited by: Å. Bergman, J.J. Heindel, S. Jobling, K.A. Kidd and R. T. Zoeller.

World Health Organization (WHO), 2017. Flooding: Managing health risks in the WHO European Region. Copenhagen, Denmark, pp. 91.

World Health Organization and Secretariat of the Convention on Biological Diversity (WHO), 2015. Connecting global priorities: biodiversity and human health: a state of knowledge review, pp. 364.

Xie, W.-Y., Shen, Q., Zhang, F.J., 2018. Antibiotics and antibiotic resistance from animal manures to soil: a review. European Journal of Soil Science, 69, 181-195.

Zaghi, D., Calaciura, B., Basili, M., Romi, R., Spinelli, O., 2010. Literature study on the impact of biodiversity changes on human health. Cominita Ambiente Srl, report for the European Commission (Directorate General Environment), July 2010, pp. 156.

Zelinkova, Z., Wenzl, T., 2015.The occurrence of 16 EPA PAHs in food – A Review. Polycyclic Aromatic Compounds, 35, 248–284. Doi:10.1080/10406638.2014.918550.

Zhang, X., Zhao, L., Tong, D.Q., Wu, G., Dan, M., Teng, B., 2016. A systematic review of global desert dust and associated human health effects. Atmosphere, 7, 158, 1-30.

Zhang, X.Q., 2016. The trends, promises and challenges of urbanisation in the world. Habitat International 54: 241-252.

Zhao, W., Liu, X., Huang, Q., Cai, P., 2015. *Streptococcus suis* sorption on agricultural soils: Role of soil physico-chemical properties. Chemosphere, 119, 52-58.

Zhao, W.Q., Liu, X., Huang, Q.Y., Rong, X.M., Liang, W., Dai, K, Cai, P., 2012. Sorption of *Streptococcus suis* on various soil particles from Alfisol and effects on metabolic activity. European Journal of Soil Science, 63, 558-564.

Annex I Case studies on epidemiological effects

Case study 1 – cross-sectional study of a polychlorinated biphenyl (PCB)-contaminated area in Brescia, Italy (Donato et al., 2008)

Brescia is an industrial town in North Italy where a factory produced organochlorine compounds, including PCBs, from the 1930s to 1984. The Local Health Unit and Local Environmental Protection Agency monitored the level of PCBs in this area and found high level of these pollutants in soils, surface waters, vegetables and animal products (chicken, eggs and cow's milk); this presence can impact human health.

A cross-sectional population-based study was conducted to investigate links between the levels of total PCB and PCB 153 and thyroid hormone in Brescia residents. Five-hundred-thirty-seven subjects were enrolled in the study, from whom blood was collected for content of 24 PCB congeners and hormone levels. They were interviewed in person about their history of thyroid disease using a structured questionnaire. The interview included: demographic variables, residential and occupational history, weight and height, past and present diseases (mainly thyroid diseases) whey they are taking hormones and/or other drugs. All thyroid hormones were analysed: total triiodothyronine (FT3 and TT3), thyroxine (FT4 and TT4) and thyroid-stimulating hormone (TSH), anti-thyroperoxidase (anti-TPO) and anti-thyroglobulin (anti-TGA).

The median serum concentration of total PCB was 713.7 ng/g lipid, and the range was 55.4–34377.8 ng/g lipid; the median of PCB153 was 207.9 ng/g lipid (range 4.7–5618.1 ng/g lipid). However, there was no statistically significant difference in total PCB and PCB 153 concentrations in subjects with and without thyroid disease. Inverse, weak and statistically significant correlations were found between total PCBs and FT3, and between total PCB and PCB 153 and TSH. These links were affected by confounding factors, including age, gender, and body mass index. The obtained results did not support the hypothesis that relatively high PCB environmental exposure can determine substantial alterations in thyroid functions in adult people. The authors indicated some limitations of their study: a small number of subjects with thyroid diseases, sometimes a long interval (even 20 years) between onset of thyroid disease and the determination PCB serum concentrations, not including the analysis of fat tissues and interviews only with adult residents.

Case study 2 – a case-control study of a PCB-contaminated area in Brescia, Italy (Maifredi et al., 2011)

This population-based case-control study was conducted in the North Italy, in the four areas close to a PCB-producing factory in a city Brescia. The aim was to investigate the possible association between PCB pollution and the risk of non-Hodgkin lymphoma. The level of PCB identified in soil was very high (8.3 mg/kg) and exceeded the Italian limit for residential area by almost 140-fold. Both incident and deceased non-Hodgkin lymphoma cases were identified from the Cancer Registry of the Brescia Local Health Authority and included in the study. Only adult cases of disease were analysed. A total of 495 cases (287 incident cases) and 1,467 controls were enrolled. Exposure to PCBs was estimated on the basis of the lifetime residential history of cases and controls provided by the Brescia Municipal Authority.

There was a positive association between non-Hodgkin lymphoma and having resided for at least 10 years in the most polluted area (odds ratio [OR] = 1.8). However, there was no statistically significant increase in this disease in subjects who resided 20 years or more in the polluted areas. Due to the absence of individual biological measures of exposure, these results should be considered with caution.

Case study 3 – clinical-epidemiological study on reproductive health consequences of environmental pollution in Southern Italy (Marra et al., 2012)
This study was conducted in the Campania Region and in Salerno and aimed to address the clinical, statistical and epidemiological relationship between birth defects and environmental pollution. Four groups were examined: pregnant women living in Salerno, pregnant women living in highly polluted areas, a control sample of pregnant women and residents from outside the Campania Region in unpolluted areas and in the Salerno area. Epidemiological data were based on a questionnaire administered to patients, namely women aged between 25 and 35 years. The total malformation prevalence has evaluated. Toxicological and genetic analysis was conducted on patients. Exposure was assessed by analysis of 30 PCB congeners in maternal blood. The epidemiological link between environmental pollution and reproductive health was found in the Salerno area. In total, 284 malformations cases were reviewed; the highest prevalence (53.8%) was recorded among women living near landfill sites in Salerno Province.

Case study 4 – epidemiological study of residents' mortality in Italian contaminated sites – SENTIERI project (Pirastu et al., 2013)
The SENTIERI project concerned human mortality in the sites identified as National Priority Contaminated Sites (NPCS) of interest for environmental remediation. Human exposure was connected with the following sources of contamination: chemical industries, petrochemical and refineries, steel plants, power plants, mines and quarries, harbour areas, asbestos or other mineral fibres, landfills and incinerators. The study population comprised residents in 44 NPCS, with special focus on causes of death for which environmental exposure was suspected or ascertained to play an aetiologic role. Mortality from 1995 to 2002 was studies for 63 single or grouped at the municipal level (small-area) by computing: crude rate, standardised rate, standardised mortality ratios (SMR) and SMR adjusted for an *ad hoc* deprivation index. Regional populations were used as a reference for SMR calculations. Characteristic element of this study was *a priori* evaluation of the epidemiological evidence of the causal association between cause of death and exposure. The epidemiological evidence was classified into three categories: sufficient (to infer the presence of causal association), limited (to infer the presence of a causal association) and inadequate (to infer the presence or the absence of a causal association).
Due to the presence of asbestos, six NPCSs were included in a national environmental remediation programme. In these sites, there was increased malignant pleural neoplasm mortality; in four sites, the excess was in both genders. Asbestos and pleural neoplasm represented a unique case. Unlike mesothelioma, most causes of death analysed in the project had a multifunctional aetiology. Increased mortality from lung cancer and respiratory diseases was observed in the sites with refineries and petrochemical plants activity (two NPCS).
Emissions from metal industries elevated mortality from respiratory diseases (two NPCS). An aetiological role for air pollution with regards to increased congenital anomalies and perinatal disorders was suggested in six NPCS. Mortality from renal failure in six other NPCS was probably caused by the presence of HM, PAHs and halogenated compounds. In the 44 NPCS, among children 0–1 year old, mortality from all causes and from perinatal conditions was, respectively 4% (3,328 cases) and 5% higher (1,903 cases) than the Italian reference population. The overall mortality was significantly increased in one or more age groups (0–1, 0–14 and 0–19 years) among children living in 25% (11 NPCS) sites.

Case study 5 – cancer incidence in Italian contaminated sites (Comba et al., 2014)
This study was conducted within the SENTIERI project in 44 NPCS. Data from the Italian Association of Cancer Registries was analysed for cancer cases in residents from 23 NPCS. For each site, the incidence of all malignant cancers combined and 35 cancer sites, coded according to International Classification of Diseases, was analysed for 1996–2005. The observed cases were compared to the expected numbers based on age, gender, calendar period, geographical area and cancer sites specific rates. Standardised Incidence Ratios were computed. The investigation included more than 2 million people living in 23 NPCS.
An excess for overall cancer incidence was observed in both genders: 9% in men and 7% in women. There were increases for specific cancer sites (colon and rectum, liver, gallbladder, pancreas, lung, skin melanoma, bladder and non-Hodgkin lymphoma). Excess mesothelioma and malignant neoplasms of the prostate, testis, kidney and brain were present among men. Elevated incidence of breast cancer, lymph haematopoietic system and chronic myeloid leukaemia was observed among women.

The study had some limitations: lack of a quantitative indicator of population exposure and the use of municipality as the smallest level of data aggregation; risk estimates were not adjusted for potential confounding factors such as alcohol consumption, smoking and socioeconomic status; incomplete coverage of cancer registration in some NPCS.

Case study 6 – industrial pollution and cancer in Spain (Fernández-Navarro et al., 2017)
This study described the industrial emissions in the vicinity of Spanish towns ad their temporal changes. Data on industrial pollutant sources for 2007–2020 were obtained from the European Pollutant Release and Transfer Register (E-PRTR) and supplied by the Spanish Ministry of Agriculture, Food & Environment. Population exposure to industrial pollution was estimated by referencing the distance from town centre to industrial facilities. The total amount of emissions was calculated for each of the carcinogens classified as recognised and suspected by IARC (Group 1, 2A and 2B). The exposed population was estimated as the annual average resident population of any town situated less than 5 km from the emission source of each substance. Relative risks of dying from cancer between exposed and non-exposed municipalities were estimated based on a spatial epidemiology technique. All estimates were adjusted for potential confounding sociodemographic indicators: population size, percentage of illiteracy, percentage of farmers, percentage of unemployed, average persons per household and mean income. The study revealed an approximately 17% excess mortality associated with malignant tumours of the digestive system and respiratory tract, gallbladder cancer, leukaemia, prostate, breast and ovarian cancers.

Annex II Characteristics of the main pollutants

Soil contamination identification—and the related risk assessment—is a complex task due to heterogeneity of the chemical properties of contaminants that affect their persistence, toxicity and bioavailability (Pérez and Rodríguez-Eugenio, 2018). Important groups of contaminants found in the soil are: heavy metals (HM), persistent organic pollutants (POP), monocyclic aromatic hydrocarbons and emerging pollutants (EP).

According to the WHO (Science Communication Unit, University of the West of England, 2013), 10 chemicals (e.g., dioxin, hazardous pesticides, arsenic, cadmium and lead) are considered as pollutants of major public concern (Table A1).

Table A1: WHO-identified chemicals of major public health concern in relation to soils and human health impacts (according to Science Communication Unit, University of the West of England, 2013).

Chemical of concern	Sources/uses	In soil?/ Used by humans as a nutrient?	Exposure	Health effect
Air Pollution		No		
Arsenic	Pesticides; Ag, Pb, Cu, Ni, Fe and steel mining or processing. Pharmaceutical and glass industry, sheep dip, leather preservatives, pigments, poison bait, agrochemicals, antifouling paint electronics industry	Yes/No	- consumption of groundwater with high natural levels of inorganic As, -food prepared with this water, -food crops irrigated with water high in As	Long period intake of inorganic As lead to: - a chronic arsenic poisoning (arsenicosis) -gastrointestinal tract, skin, heart, liver and neurological damage -diabetes -none marrow and blood diseases -cardiovascular disease -carcinogenic Organic As compounds are less harmful to health Increased risk of miscarriage, stillbirth and pre-term birth
Asbestos	Mining and milling of raw asbestos for construction and product manufacturing. Historical: release into the air and soil around refineries, power plants, factories handling asbestos, shipyard, steel mills, vermiculite mines, building demolition Current: repair, renovation, removal, or maintenance of asbestos. Gardening	Yes/No	Inhalation, ingestion and lodge on the skin of asbestos fibres during crumbling of asbestos-containing materials	After inhalation, asbestos fibres remain for a long time in lung tissue and cause: -parenchymal asbestosis -asbestos-related pleural abnormalities -lung carcinoma -pleural mesothelioma Lung cancer and pleural mesothelioma have high mortality rates

Chemical of concern	Sources/uses	In soil?/ Used by humans as a nutrient?	Exposure	Health effect
Benzene		No/No		
Cadmium	Zn smelting, mine tailings, burning coal or garbage containing Cd, rechargeable nickel-cadmium batteries, pigments, TVs, solar cells, steel, phosphate fertiliser, metal plating, water pipes, sewage sludge	Yes/No Cd in soil may enter plant crop	Through human food chain consumption of plant and animal products that may accumulate Cd	-liver and kidney damage, low bone density -diets poor in Fe and Zn vastly increase the negative health effects of Cd -carcinogenic (by inhalation)
Dioxin polychlorinated dibenzodioxins (PCDD) and dibenzofurans (PCDF)	Waste incineration, reprocessing metal industry, paper and pulp industry, contaminated herbicides; stored PCB-based industrial waste oils	Yes/No	Through consumption of contaminated food; 90% of exposure is through food, mainly meat and dairy products, fish and shellfish	Highly toxic, can cause reproductive and developmental problems, damage the immune system, interfere with hormones and also cause cancer
Fluoride		Yes/Yes In soil immobile; Appropriate level strengthens teeth	High level of fluoride in drinking water	Skeletal fluorosis as a result of long accumulation in the bone; early symptoms include stiffness and pain in the joints
Lead	Batteries, solder, ammunition, pigments, paint, ceramic glaze, hair colour, fishing equipment, leaded gasoline, mining, plumbing, coal burning, water pipes	Yes/No	Leaded fuel and mining activities are common causes for elevated lead levels in topsoil	-neurological damage -lowers IQ and attention -hand-eye coordination impaired -encephalopathy -bone deterioration -hypertension -kidney disease
Mercury	Electrical switches, fluorescent light bulbs, lamps, batteries, thermometers, dental fillings, mining, pesticides, medical waste, burning coal and fuel oil, chlor-alkali industry	Yes/No	Eating contaminated seafood; for children, direct ingestion of soil	-central nervous system ad gastric system damage -affects brain development, resulting in a lower IQ -affects coordination, eyesight and sense of touch -liver, heart and kidney damage -teratogenic

Chemical of concern	Sources/uses	In soil?/ Used by humans as a nutrient?	Exposure	Health effect
Hazardous pesticides	Herbicides Insecticides (DDT)	Yes/No	Organic pesticides accumulate in the food chain	Organic chemicals, including pesticides, have been linked to a wide range of health problems, but we tend to be exposed to a cocktail of these chemicals at low levels.

The main **HM** sources in soil are deposition from the contaminated atmosphere (by production of energy and construction materials, transport and metallurgy), sewage sludge application, industrial and mining solid waste, agricultural application of fertilisers, pesticides and mulch (Gebreyesus, 2014). In Europe, HM contamination concerns mainly regions with the presence of the historical mining and heavy industry (Tóth et al., 2016). The HM soil pollution is the most problematic in areas which are simultaneously contaminated by other groups of contaminants, especially because acidifying chemicals increase HM mobilisation and bioavailability (Lado et al., 2008).

POP are organic contaminants resistant to chemical and biological degradation processes and to photolytic processes (Swartjes, 2011; Rodríguez-Eugenio et al., 2018). For this reason, they can persist in the environment and bioaccumulate in human and animal tissue. POP are often halogenated, usually with chlorine. The main contaminants that belong to this group are: pesticides (aldrin, chlordane, DDT, dieldrin, endrin, heptachlor, hexachlorobenzene and mirex), polychlorinated biphenyls (PCB), polychlorinated dibenzo-p-dioxins (PCDD), polychlorinated dibenzofurans (PCDF), brominated flame-retardants and PAH. Many POP are pesticides that are banned in European countries but are still are found in soils and will reside in the soil for many decades. POP compounds have a low water solubility, high lipid solubility and limited volatility; many of them accumulate in adipose tissue (Swartjes, 2011; Rodgers et al., 2018). Due to these properties and their biochemically stable molecular structures, they are not easily biodegraded in the environment.

Dioxins (PCDD and PCDF) and some organochlorine pesticides (e.g., DDT) are bioaccumulate and concentrated in the food chain. Currently, the largest source of dioxins is the incineration of chlorine-containing waste, which creates dioxin air pollution. Human exposure to dioxins is mainly through consumption of dairy products, meat, fish and eggs (Valentin et al., 2013).

PAH are released into air, but when they fall to the ground, they easily accumulate in the soil (Abdel-Shafy and Mansour, 2016). They are formed during anthropogenic (main source) and natural incomplete combustion of the organic substances and are widespread in the environment (Valentin et al., 2013; Abdel-Shafy and Mansour, 2016). Due to their low solubility, PAH tend to be deposited into soil, where they may be degraded or transformed. PAHs bound to organic matter have reduced mobility, but they also have a higher resistance to biodegradation (Valentin et al., 2013). Some PAH are toxic towards soil organisms and carcinogenic and mutagenic to humans (Table A2; Abdel-Shafy and Mansour, 2016).

According to the Stockholm Convention[29], the presence of POP in soil leads to serious health effects (Table A2).

Monocyclic aromatic hydrocarbons categorised as BTEX (e.g., benzene and toluene) are commonly found in crude oil. These volatile substances are released into air, but they can enter the soil and pose serious environmental and health problems (Table A1). Inhaling BTEX-polluted air is the greatest hazard to humans by these compounds. BTEX are water-soluble compounds, and thus improper handling can also cause groundwater contamination (Valentin et al., 2013; Godambe and Fulekar, 2017).

Table A2: Overview of possible health effects for the main groups of soil organic pollutants (compilation based on the literature).

Contaminants	Source	Health effects	Reference
Polycyclic aromatic hydrocarbons (PAH)	Power generation, transportation, industry	Genotoxic, oestrogenic and anti-oestrogenic activity, inflammation, mammary gland tumours, lung, bladder and skin cancer, cardiovascular diseases, diabetes, immune system disorders	Rodgers et al., 2018; Chen et al., 2019
Organochlorine pesticides (DDT, DDE, HCB, lindan, deldrin, aldrin, toxaphene, mirex and HCB)	Agriculture and pest control, lice control	-Endocrine disrupting chemical, disrupt hormonal homeostasis, oestrogenic and anti-androgenic effects in men -Non-Hodgkin lymphoma -Testicular cancer -Breast cancer	Bornman et al. 2018; Rodgers et al., 2018 Qu et al., 2019
Non-organochlorine pesticides (chlorpyrifos, parathion, malathion and atrazine)	Pesticide application	Neurotoxicity Mammary tumours Breast cancer	Rodgers et al., 2018
Dioxins	Combustion and industrial processes involving chlorine, including production of pesticides, bleached paper, and polyvinyl chloride	Breast cancer immunotoxicity, developmental effects, soft-tissues sarcoma	Kogevinas, 2011; Rodgers et al., 2018

[29] The Stockholm Convention on Persistent Organic Pollutants (http://chm.pops.int/TheConvention/Overview/TextoftheConvention/tabid/2232/Default.aspx).

Contaminants	Source	Health effects	Reference
Polychlorinated biphenyls (PCB)	Electrical equipment and other industrial applications	-Endocrine-disrupting chemical Pseudo-oestrogen activity, anti-oestrogenic activity, changes of thyroid hormone production, enlargement of the thyroid gland, immunotoxicity, hypothyroidism or hyperthyroidism, -Non-Hodgkin lymphoma -Malignant melanoma -Breast cancer	Donato et al., 2008; Maifredi et al., 2011; Rodgers et al., 2018; Alias et al., 2019

Annex III Characteristics of emerging contaminants

Emerging pollutants (EP) are defined as chemicals of a synthetic origin or derived from a natural source that have recently been discovered. However, they are not yet commonly monitored in the environment, although they have the potential to enter the environment and cause known or suspected adverse ecological and human health effects (Geissen et al., 2015; Naidu et al., 2016). EP release into the environment has likely occurred for a long time but may not have been recognised until new detection methods were developed. Synthesis of new chemicals or changes in use and disposal of existing chemicals can create new EP sources (Geissen et al., 2015). Many EP are present in every day products, e.g., medicines, textiles, cosmetics, detergent products, petrol additives, etc. (see Table A3).

EP are a wide and diverse group of chemicals; more than 900 emerging substances were found in the European environment (Geissen et al., 2015). EP are divided into more than 20 groups related to their origin. Among them, the most common are: perfluorinated compounds (PFC), disinfection by-products (DBPs), gasoline additives, man-made nanomaterials, human pharmaceuticals, veterinary pharmaceuticals (VA), endocrine–disrupting compounds (EDC), cyanotoxins and rare earth elements (Table A3).

EP are released to the environment from many anthropogenic sources, mainly from waste water treatment plants from urban and industrial areas, through atmospheric deposition or from crop and animal production (Gavrilescu et al., 2014; Geissen et al., 2015; Picó et al., 2019). They are spread to the different environmental compartment (water, air, soil and organisms); due to their persistence, bioaccumulative nature and toxicity they can affect human health (Table A4; Lei et al., 2015; Naidu et al., 2016; Snow et al., 2017). A group of pollutants of special concern are perfluoroalkyl and polyfluoroalkyl substances (PFAS), which are derived from different sources, e.g., industrial sites, landfills, waste water treatment plants and places where large fires have occurred and/or fire-fighting trainings have been performed (Concawe, 2016). Some PFAS compounds (mainly short-chain) are persistent, soluble in water and mobile in soil, and thus they are likely to contaminate ground and surface water when released into environment (Concawe, 2016).

Table A3: Characterisation of the common groups of emerging contaminants (compilation based on the listed references).

Group of the EPs	Group characterisation	Representatives of the group	Application	Reference
Perfluoroalkyl and polyfluoroalkyl substances (PFAS)	Persistent, bioaccumulative and potentially hazardous to animals and humans	Perfluorooctane sulfonate (PFOS), Perfluorooctanoic acid (PFOA), Salts of PFOS and PFOA Perfluoroalkyl carboxylic acids (PFCAs) Perfluoroalkane sulfonic acids (PFSAs)	In fire-fighting foams, lubricants, metal spray plating and detergent products, inks, varnishes, coating formulations (for walls, furniture, carpeting, and food packaging), waxes, and water and oil repellents for leather, paper, and textiles	Lei et al., 2015 Concawe, 2016; Siddique et al., 2016; Winkens et al., 2017

Group of the EPs	Group characterisation	Representatives of the group	Application	Reference
Disinfection by-products (DBP)	Created as a by-product from the disinfection chemicals	- Chlorinated DBPs (CDBPs) - Trihalomethanes (THMs) - Halogenated acetic acids (HAAs)	Disinfection chemicals are used in purification of the swimming pool and drinking water	Lei et al., 2015
Gasoline additives	More than 500 substances with varied chemical properties	- Methyl tert-butyl ether (MTBE)	MTBE is used as unleaded petrol additive	Lei et al., 2015
Man-made nanomaterials	Substances with a particle size of 1–100 nm, characterised by a strong adsorption capacity in the environment (air, water and soil) and catalytic character, easily transformed into new pollutants	- Amorphous silicon dioxide (SiO_2), - Carbon nanotubes (CNT), - Titanium dioxide (TiO_2)	Sunscreen products, agriculture, transport, healthcare, materials, energy and information technologies	Lei et al., 2015
Human pharmaceuticals	Chemicals used in human medicine	- approximately 3000 substances, e.g.: lipid regulators, anti-inflammatory drugs, analgesics, contraceptives, neuroactive medicine, antibiotics, and beta-blockers	Increasing application in human medicine	Lei et al., 2015
Veterinary pharmaceuticals (VP)	Chemicals used in animal medicine	- antimicrobial, anthelmintic, steroidal and nonsteroidal, anti-inflammatory, antiparasitic, astringent, oestrus synchronisation, nutritional supplement, and growth promoter	Increasing applications in animals medicine	Lei et al., 2015
Sunscreens/Ultra violet Filters	Substances protected against harmful UV radiation	- Inorganic: scatter UV radiation with wavelengths of 290 to 400 nm - Organic: absorb novel photons of UV	Personal care products (e.g., skin care products, makeup) and non-cosmetic products (e.g., furniture, plastics, carpets and washing powder)	Lei et al., 2015

Group of the EPs	Group characterisation	Representatives of the group	Application	Reference
Endocrine–disrupting compounds (EDC)	- Substances or mixtures that change the endocrine system function, leading to the disadvantageous health effects - Mainly substances that working as agonists or antagonists on oestrogen or androgen receptors in vertebrates	- Biocides, - Industrial compounds, - Surfactants - Plasticisers, e.g., bisphenol A (BPA), which is both an agonist for the oestrogen receptor and an antagonist for the androgen receptor, and is widespread in the environment (air, soil, water and wildlife)	- BPA is used for the plastic resins production (food and beverage packaging, flame retardants, adhesives, - building materials, electronic components, and paper coatings	Staples et al., 1998; WHO IPCS, 2002; Flint et al., 2012; Valentin et al., 2013; Corrales et al., 2015
Cyanotoxins	Microcystins (MC) are formed by diverse cyanobacterial strains (e.g., *Microcystis* spp.) and can bioaccumulate in some edible plants	MC is the most common group; microcystin-LR (MC-LR) is the most toxic in this group	Introduced to the soil: - intentionally as fertiliser - non-intentionally with irrigation water	Cao et al., 2018; Drobac et al., 2013; Snow et al., 2017
Rare earth elements		Lanthanides	-Solid waste and wastewaters from mining and mineral processing plants; -Acid mine drainage -Waste electronic and electrical equipment -Fertilisers and animals feeds	Gwenzi et al., 2018

A detailed overwiew of health effects caused by emerging compounds is provided in Table A4.

Table A4. Characterisation of suspected health effects caused by common groups of the EP (compilation based on the available literature).

Group of the EP	Suspected health effects (unproven so far)	Reference
Per- and polyfluoroalkyl substances (PFAS)	1) carcinogenic 2) Perfluorooctane sulfonate (PFOS) and Perfluorooctanoic acid (PFOA), may reduce fecundity in women 3) PFOS might decrease a woman's capacity to lactate 4) some reproductive dysfunction in men 5) might change thyroid hormone levels	Valentin et al., 2013; Lei et al., 2015; Concawe, 2016; Siddique et al., 2016; Winkens et al., 2017
Disinfection by-products (DBP)	1) carcinogenic 2) factor of risk for: infertility, foetal loss, long gestational duration, poor foetal growth, foetal anomalies	Lei et al., 2015
Gasoline additives	1) some gasoline additives are known or suspected carcinogenic substances 2) MTBE - suspected carcinogen 3) MTBE (in the air, quickly absorbed by inhalation) is potentially toxic in the human body	Lei et al., 2015
Man-made nanomaterials	1) carcinogenic 2) at high doses reproductive and developmental toxicity	Lei et al., 2015; Naidu et al., 2016;
Human and veterinary pharmaceuticals	1) direct effects (acting of the residues of drugs in the body) 2) indirect effects of antibiotics present in the environment (increase of the antibiotic resistance of microorganisms) are unknown in the context of the human antibiotic resistance 3) long-term exposure effects on the low level present in the environment (up to know has not been studied)	Lei et al., 2015; Hashmi et al., 2017; Grenni e al., 2018
Sunscreens/Ultraviolet Filters	Neurotoxicity	Lei et al., 2015; Ruszkiewicz et al., 2017
Endocrine–disrupting compounds (EDC)	EDC: 1) Impact on reproduction in males and females 2) Breast cancer 3) Abnormal growth patterns and neurodevelopmental delays in children 4) Immune function disrupting - BPA might be the cause of the cancer, infertility, diabetes and obesity	Valentin et al., 2013; WHO, 2013b; Naidu et al., 2016; Almeida et al., 2018
Cyanotoxins	1) hepatotoxic (microcystins, nodularin, cylindrospermopsin) 2) neurotoxic (saxitoxins, anatoxin-a, anatoxin-a(s), homoanatoxin-a) 3) cytotoxic (aplysiatoxin, debromoaplysiatoxin, lyngbyatoxin, lipopolysaccharide endotoxin) 4) skin and gastrointestinal irritation	Drobac et al., 2013

Group of the EP	Suspected health effects (unproven so far)	Reference
Rare earth elements (lanthanides)	-nephrogenic systemic fibrosis (gadolinium-based contrast agents) -dysfunctional neurological disorders (reduced IQ in children associated with La) -Cerium oxide (Ce)-pneumoconiosis -bone alterations, genotoxicity and fibrotic tissue injury -anti-testicular effects and male sterility	Gwenzi et al., 2018

Currently, little is known about the fate, occurrence and hazardous effects of EP to the natural environment and human health (Gavrilescu et al., 2014). Therefore, improving knowledge about EP is an important and necessary challenge. Future research on EP should consider the entire lifecycle of the pollutants, from the source of emission to their removal through treatment and remediation techniques, including also the impacts and risks they may pose to the environment and human health (Figure A1).

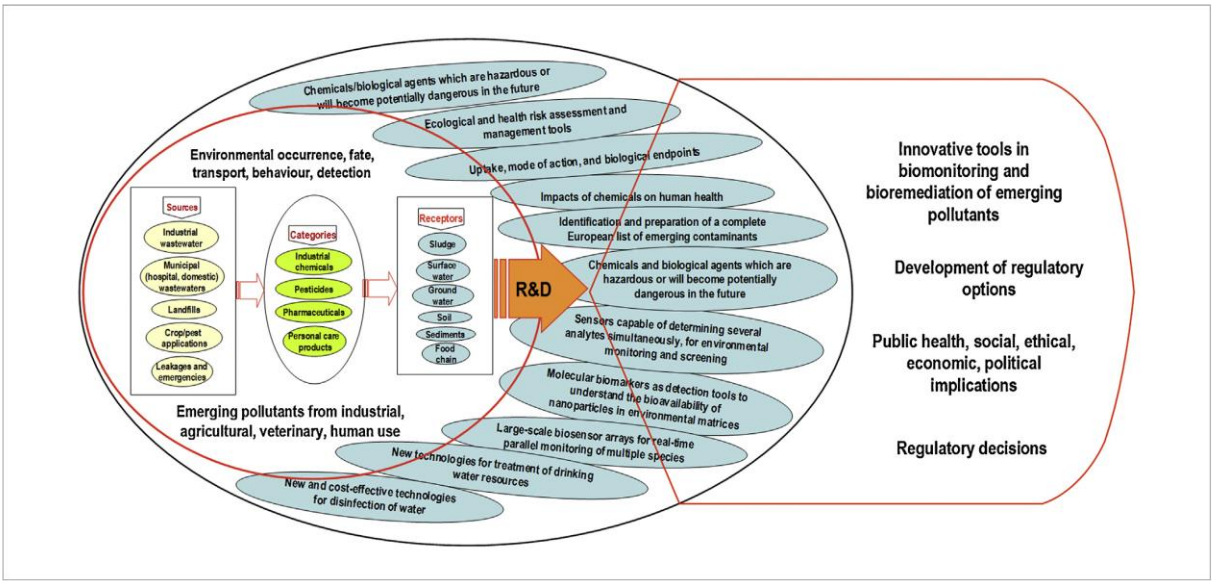

Figure A1: Research and development needs for emerging pollutants in the environment considering sources, biomonitoring, ecological risks and bioremediation (according to Gavrilescu et al., 2014).