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Using new contaminants information to re-assess environmental risks from sewage sludge

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List of Abbreviations

AF – Assessment Factor

AMPA - Aminomethylphosphonic Acid

BBP - Benzyl Butyl Phthalate

BDE – Brominated Diphenyl Ether

BFR – Brominated Flame Retardant

BHT – Butylated Hydroxytoluene

BPA – Bisphenol A

DEHP – Di(2-ethyhexyl)phthalate

E1 – Estrone (oestrone)

E2 – Estradiol (oestradiol)

E3 – Estriol (oestriol)

EC – Effect Concentration

EC50 – Half of the (maximum) Effect Concentration

EE2 – Ethinylestradiol (ethinyloestradiol)

EPM – Equilibrium Partitioning Method

FTIR – Fourier Transform Infrared

GC-MS – Gas Chromatography/Mass Spectrometry

HBCD - Hexabromocyclododecane

HC – Hazardous Concentration

HC5 – Hazardous Concentration for 5% of species studied

HONEC – Highest Observed No Effect Concentration

LAS – Linear Alkylbenzene Sulphonates

LC50 – Lethal Concentration with 50% mortality

LOEC – Lowest Observed Effect Concentration

MP - Microplastic

NOEC – No Observed Effect Concentration

NP – Nonylphenol

NP1EO – Nonylphenol Monoethoxylate

NP2EO – Nonylphenol Diethoxylate

NPE – Nonylphenol Esters

OCC – Organic Chemical Contaminants

OP – Organophosphate

OPE – Organophosphate Esters

OT - Organotin

PAE – Phthalate Acid Esters

PAH – Polyaromatic Hydrocarbon

PBDE - Polybrominated Diphenyl Ether

PCA – Polychlorinated Alkane

PCB – Polychlorinated Biphenyl

PCN – Polychlorinated Naphthalene

PCDD/F - Polychlorinated Dibenzodioxins and Furans

PDMS - Polydimethylsiloxanes

PEC – Predicted Effect Concentration

PFAS – Per- and Polyfluoroalkyl Substances

PFC - Perfluorochemicals

PFHxS – Perfluorohexanesulfonic Acid

PFNA – Perfluorononanoic Acid

PFOA – Perfluorooctanoic Acid

PFOS – Perfluorooctanesulfonic Acid

PNEC – Predicted No-Effect Concentration

POP – Persistent Organic Pollutant

PPCP – Pharmaceutical and Personal Care Products

QAC – Quaternary Ammonium Compounds

QRA – Quantitative Risk Assessment

RQ – Risk Quotient

RR – Relative Risk

SSD – Species Sensitivity Distribution

SSRI - Selective Serotonin Reuptake inhibitor

TBBPA – Tetrabromobisphenol A

TCC - Triclocarban

TCEP – Tris(2-chloroethyl)phosphate

TCPP – Tris(1-chloro-2propyl)phosphate

TCS - Triclosan

TDCP or TDCPP – Tris[2-chloro-1-(chloromethyl)ethyl]phosphate

TED-GC-MS – Thermal Extraction Desorption Gas Chromatography/Mass Spectrometry

TFA – Trifluoroacetic Acid

TPP – Triphenyl Phosphate

VOC – Volatile Organic Compounds

WWTP – Waste Water Treatment Plant

Executive summary

This is a comprehensive report on assessing environmental risks associated with the agricultural use of sewage sludge, focusing on the impacts of unregulated microplastic, organic chemical and pharmaceutical contaminants on soil health. This report, prepared by the James Hutton Institute at the request of Fidra, revisits the risk assessment conducted in 2018 for the Scottish Government that assessed risks to human and environmental health. This assessment of the environmental impacts of microplastics, organic chemical and pharmaceutical contaminants found in sewage sludge is an important resource for policy makers, regulators, industry, academic and environmental NGO stakeholders.

Key Highlights:

Background

Treated sewage sludge, often referred to as biosolids, is commonly used as a soil amendment in agriculture due to its low cost and nutrient content, especially nitrogen and phosphorus. Around 87% of the 3.6 million tonnes of sewage sludge produced in the UK is recycled for agricultural purposes. Despite the potential nutrient benefits, sewage sludge also introduces potentially hazardous contaminants, with regulation focusing on heavy metals. More recent attention has shifted to the unregulated microplastics, organic chemicals and pharmaceuticals and personal care products (PPCPs). The evolving nature of our scientific understanding of contaminants necessitated the reassessment of risks posed by biosolids to soil health and the wider environment.

Project Scope and Objectives

The reassessment aimed to update and extend the 2018 risk assessment by focusing on unregulated contaminants, investigating their impacts on the environment, particularly soil health. The key contaminants of concern included:

- Microplastics and fibres
- Organic and other emerging chemicals
- Pharmaceuticals and personal care products (PPCPs)

Although focussing on the environment rather than human health, this reassessment used the same approach as the 2018 risk assessment, consisting of four stages: hazard assessment, exposure assessment, dose-response analysis, and risk characterization. The focus was on microplastic and chemical contaminants in soils, but risks to water quality and human health were also considered where relevant.

Microplastics and Fibres

Microplastics, defined as synthetic polymers smaller than 5 mm, have been widely detected in all ecosystems, with significant quantities entering agricultural soils through sewage sludge application. Research indicates that microplastics alter soil properties, microbial activity, and can harm soil organisms through ingestion and other means. In this report, microplastics were found to accumulate in soil, and repeated applications of sewage sludge could result in appreciable risks to soil biota over time.

Organic and other emerging chemicals

Organic chemical contaminants include persistent organic pollutants (POPs) such as PCBs, PAHs, and PFAS. These contaminants often bioaccumulate in soil and biota, presenting significant environmental risks. The report highlights the presence of these pollutants in sewage sludge and their potential long-term accumulation in soils due to repeated applications of biosolids. Particular focus was placed on PFAS due to their widespread use and persistence in the environment.

Pharmaceuticals and Personal Care Products (PPCPs)

The report also evaluates the risks posed by pharmaceuticals and personal care products (PPCPs), which enter wastewater through consumer use. These substances have been detected in sewage sludge and pose risks to soil health and biota, though the exact mechanisms and impacts are still being studied. This reassessment sought to provide updated toxicity metrics and exposure estimates for these contaminants.

Methodology

The risk assessment followed a classical quantitative risk assessment (QRA) approach, as used in the 2018 study. The process involved:

1. **Hazard Identification:** Identifying and prioritizing hazardous agents in sewage sludge, especially those that had emerged post-2018. The assessment included three key categories of contaminants: microplastics, organic chemicals, and PPCPs.
2. **Exposure Assessment:** Estimating the concentration of hazardous agents in soil, following the application of sewage sludge, that could encounter soil biota. This involved modelling the distribution of contaminants between different soil compartments (air, water, and sludge-amended soil) using fugacity modelling.
3. **Dose-Response Assessment:** Evaluating the toxicity of identified hazards by deriving metrics such as predicted no-effect concentrations (PNEC) for soil biota. The assessment used published toxicity data and applied assessment factors to account for uncertainty.
4. **Risk Characterization:** Comparing predicted exposure levels to toxicity thresholds (PNECs) to estimate the risk to soil organisms. Relative risk (RR) was calculated, with RR values greater than 1.0 indicating a potential for adverse effects.

Key Findings

Microplastics

Microplastics in sewage sludge present a potential risk to soil health, particularly after repeated applications of sludge. While a single application of sewage sludge was found to pose minimal risk, multiple applications could result in significant accumulation of microplastics in the soil. The report recommends minimizing the input of microplastics into soils by improving wastewater treatment processes and through source control measures to reduce plastic use in consumer products.

Organic and other emerging chemicals

The report underscores the risks associated with persistent organic pollutants (POPs), particularly PFAS and PCBs, which can accumulate in soils and have long-term environmental impacts. The risk characterization for these substances resulted in certain organic contaminants demonstrating elevated risk ratios, that could impact soil health. The study recommends further research into the long-term fate of these contaminants in agricultural soils and stricter regulations to limit their release into wastewater systems.

Pharmaceuticals and Personal Care Products (PPCPs)

PPCPs present risks to soil health, with the study showing potential for accumulation in soil biota over time. A number of PPCPs demonstrated elevated risk ratios, suggesting significant risk to soil health. The report highlights the need for more research on the environmental fate of PPCPs and recommends precautionary measures to reduce their presence in sewage sludge. It also advocates for the development of new treatment technologies that can effectively remove PPCPs from wastewater, and improved source control.

Recommendations

1. **Adopt the Precautionary Principle:** Given the persistence, bioaccumulation potential and emerging nature of many contaminants, the findings of this report suggest precautionary measures are needed to protect soil health and the wider environment. This may involve limiting the application of sewage sludge on agricultural land until more comprehensive risk data are available and includes adopting recommendations 2 – 7.
2. **Improve Wastewater Treatment:** Given the risks from contaminants and uncertainties identified in this report enhancement of wastewater treatment processes to reduce the levels of contaminants is needed, especially microplastics, organic chemicals, and PPCPs, in sewage sludge. This includes upgrading treatment facilities and introducing stricter regulations on pollutant discharge.
3. **Increase Research on Emerging Contaminants:** Further research into the environmental risks posed by unregulated microplastic and emerging chemical contaminants is needed. More data of improved accuracy and reproducibility will enhance our understanding of long-term impacts of these substances on soil biota and the wider environment.
4. **Regular Monitoring of Contaminant Levels:** ongoing monitoring of chemical and microplastic contaminant levels in sewage sludge and agricultural soils is needed to help identify trends in contamination and enable timely interventions to minimise environmental damage.
5. **Develop Alternative Treatment Technologies:** To address the limitations of current wastewater treatment systems, new technologies for removing persistent and other contaminants from sewage sludge should be explored.
6. **Develop Enhanced Quality Standards**: Encourage upstream source control solutions for reducing and/or eliminating contaminants in sewage sludge before recycling to land.
7. **Improved Sludge Use in Agriculture Policies**: Widen scope of regulations to enforce appropriate management strategies and best practice to ensure soil health and the wider environment are protected from a more comprehensive range of contaminants. Build in flexibility and review processes, so that regulations adapt to changes in the contamination profile in a timely manner.

Conclusions

The reassessment of environmental risks from sewage sludge highlights significant concerns regarding the accumulation of contaminants, particularly microplastics, organic chemicals, and pharmaceuticals, in agricultural soils. While the application of sewage sludge may provide nutrients and organic matter to the soil, the risks posed by unregulated microplastic and chemical contaminants necessitate caution. The findings suggest enhanced regulation, improved wastewater treatment, alternative treatment technologies, enhanced quality standards, robust monitoring and further research are needed to ensure the safe and sustainable use of sewage sludge in agriculture.

Using new contaminants information to re-assess environmental risks from sewage sludge

1. Introduction and methodology

1.1. Background

Treated sewage sludge, also known as biosolids, has been recycled to agricultural land for many decades in the UK, Europe, the USA and beyond. The most frequently cited data suggest that about 87% of the 3.6 million tonnes (fresh weight) of sewage sludge produced in the UK is currently recycled to agricultural land (Black et al., 2016; BAS, 2024). This 87% roughly equates to £25M equivalent in inorganic fertiliser costs (September 2022 prices). Sewage sludge is a source of nitrogen and phosphorus, as well as other major and minor plant nutrients. Sewage sludge can also have value as a liming agent, but this is dependent on production methodology (SRUC, 2013). Sewage sludge also contains organic matter and is therefore a practicable approach to replenishing soil organic matter levels providing long-term benefits to soil structure and fertility. Therefore, the recycling of sewage sludge to land is widely recognised as being the best practicable environmental option by the UK Government in most circumstances.

Despite these benefits and proponents, the application of sewage sludge to land has always been associated with the introduction of potentially hazardous agents to the soil environment, with the potential to enter food chains and aquatic environments. Contaminants get into wastewater streams via legitimate industrial activities and use of consumer products, as well as clandestine non-compliant activities. The nature and knowledge of these hazards has changed over the decades and this evolution can be summarised as focussing on heavy metals four decades ago to now researching a huge spectrum of organic chemicals and emerging contaminants. While efforts have been made to regulate some contaminants (such as heavy metals) other specific contaminants (such as PFAS) have risen to prominence. Any increased concerns around chemical contaminants are due to a variety of factors. These include changes in regulations surrounding manufacture of specific chemicals, the use of a wider variety of synthetically manufactured chemicals in our consumer products, food, and medicines, developments in water treatment, and technological developments in analytical (bio)chemistry that have enabled a far greater suite of contaminants to be studied at increasingly lower concentrations. In addition, advances in our knowledge and understanding of specific issues such as microplastics and anti-microbial resistance has occurred. The practice is also associated with malodour and can be perceived negatively by the general population. Sewage sludge is an organic fertiliser and soil conditioner, but it also has negative impacts that cannot be ignored.

In this context, partly due to complaints and concerns from communities reporting potential issues associated with the recycling of sewage sludge to land, a review of the existing legislation and procedures relating to the spreading of sewage sludge to land was

commissioned in 2014 (Scottish Government, 2016). This work was overseen by Scottish Government, SEPA and Scottish Water and brought forward various recommendations. One key recommendation focused on the potential human health impacts associated with the spreading of sewage sludge and the lack of up-to-date, robust evidence. Several gaps in current knowledge, such as antimicrobial resistance, microplastics, pharmaceuticals and personal care products (PPCPs), as well as odour were highlighted for specific attention.

In response, in 2016, the James Hutton Institute along with RSK-ADAS and AquaEnviro were commissioned to significantly update and extend the (at the time) existing 2008 sewage sludge risk assessment (SNIFFER, 2008). This work was delivered in 2018 (Scottish Government, 2018) but not published online until October 2021.

This report provides an update to these previous studies with a focus on risks to soil health for chemical contaminants and microplastics.

1.1.1. Precautionary approach and circular economy concerns

Many environmental NGOs, governments and academic experts recognise that lack of data does not equate to lack of risk and therefore advocate for the adoption of the precautionary principle where policy action should be taken if there is a credible danger of significant harm to the environment but no scientific certainty regarding the risks (Fidra & Environmental Investigation Agency, 2024) . This is in line with the UK's Environment Act and UK's legally binding Environmental Principles Policy Statement. There are also concerns that contaminated sewage sludge use in agriculture represents a barrier to Scotland's (and the rest of the UK's) future vision of healthy productive soils and a safe, clean circular economy, especially where emerging scientific evidence supports the need for updated risk analysis.

1.1.2. Report scope and objectives

This updated risk research report utilised data published since 2018 to re-assess, re-analyse and update the risk analysis components of the sewage sludge research completed six years ago (Scottish Government, 2018). While the 2018 assessment had a focus on human health impacts (primarily) via the terrestrial food chain, this project has particular focus on impacts to the soil and wider environment.

This project follows the same classical risk approach as adopted for the 2018 assessment (Scottish Government, 2018), namely four stages; hazard assessment, exposure assessment, dose-response analysis, and risk characterisation. The focus was chemical contaminants, although other hazards such as microplastics were included. This project had the following objectives:

1. Update the original hazard screening literature review undertaken for the 2018 risk assessment, this is to identity new and emerging hazards present in sewage sludge, as well as to identify new sources of data for hazards that were poorly understood during the 2016 – 2018 project. The hazard screening will be mindful of the change in scope towards an emphasis on impacts to soil health and the wider environment. All identified hazards will be qualitatively screened for relevance to the scope (presence in sewage at levels of concern, ability to present an exposure of environmental concern, etc.) prior to being passed to the latter stages of the assessment. Specifically, the project included:

a. Microplastics and fibres – to identify if there are new data available, specifically information on the mobility of microplastics via exposure routes which were absent during the 2016-2018 assessment.

b. Organic and other emerging chemicals – this is a fast-moving area of environmental chemistry, both in terms of number and range of chemicals now being measured, and in terms of toxicological understanding of those chemicals. Significant changes/updates are expected to this category of hazard compared to 2018.

c. Pharmaceutical and personal care products – this is also a fast-moving area of environmental chemistry, and we would also expect significant updates compared to 2018.

2. Estimate potential exposure for new and updated hazards identified above (1a – c). This will be undertaken for those hazards that pass through the screening process and are therefore likely to present an exposure of concern to receptors via e.g., the food chain. Key receptors will be identified as part of the project, but it is envisaged that soil biota will be a key receptor associated with risks to soil health/function.

3. Evaluate the toxicity of the identified hazards. For quantitative risk assessment, we are particularly interested in metrics of toxicity such as reference doses or published no effects levels from experimental data. We anticipate very different types of information, and quality of information, for each of the identified hazards. We will account for this by providing an indication of uncertainty for each toxicity metric.

The final objective is the risk characterisation of the identified hazards. This will only be performed where the underlying data are considered suitable for making a reasonable risk estimate. As with objective 3, while leaning towards pragmatism using the precautionary principle, an indicative level of certainty will be provided for each risk estimate.

1.2. General methodology

This section provides an overview of the methodology used within this generalised Quantitative Risk Assessment (QRA). Specific details and assumptions for individual potentially hazardous agents are described further in Sections 2 – 4.

The aim of this work was to undertake a quantitative risk assessment that establishes the potential for harm to soil biota resulting from the application of sewage sludge products in agriculture. The assessment considered a range of treatment and processing parameters, as well as different agricultural end uses (Table 1.1).

Table 1.1 - Summary of sewage sludge products and end use parameters used to constrain input data to the quantitative risk assessment.

Input materials	Treatment Method	End Use Parameters
• Thickened sludge from primary and secondary water treatment	• Conventionally (anaerobically) digested and dewatered cake • Dewatered raw cake with limed pasteurisation (lime caked) • Thermal hydrolysis pasteurisation (THP) digested and dewatered cake	• Agriculture (grazing land) • Agriculture (land used to grow grain/root/leafy crops for animal consumption) • Agriculture (land used to grow grain/root/leafy crops for human consumption)

This assessment only considered sewage sludge that has been produced under regulatory control. Activities outside of this specification, including unauthorised contamination of feedstocks and illegal use of sewage sludges, have not been considered. This assessment examines potential risks to soil biota from a specified and regulated product, and therefore does not make predictions about system failure, bypass of processing systems, or illegal activities. The assessment did not consider contamination already present in the receiving environment. As such, this assessment estimates *additional* risks associated with sewage sludge as a product and makes no attempt to compound these risks with existing environmental conditions.

The assessment was undertaken using the *classical model* for QRA. This approach has been widely adopted, including by relevant UK agencies such as the Department of the Environment, Food & Rural Affairs (DEFRA), the Institute of Environment and Health (Defra, 2011), and was the approach adopted by the 2018 assessment (Scottish Government, 2018). Throughout the assessment process, *realistic worst-case* assumptions were adopted. The phrase *realistic worst case*, and its definition, have been developed by the author and applied in many generalised risk assessments (e.g., see Qvarforth et al., 2022; Stubberfield et al., 2022; Mitchell et al., 2020; Longhurst et al., 2019; Beesley et al., 2018; Troldborg et al., 2017; Mollon et al., 2016; Zhang et al., 2015; Avery et al., 2012; Minh et al., 2012; Hough et al. 2004, 2006, 2010 and 2012). The purpose of a realistic worst case, as opposed to a worst case, is to support precautionary decision making. This means that the risks are examined under unusual but legitimate scenarios. If an agent is deemed to pose little risk under these circumstances, then there is confidence that it poses little risk under most circumstances. If, however, an agent poses a risk under the realistic worst-case scenario, it is a plausible outcome and indicates that further investigation is required before real-world implementation of such a scenario.

The standard QRA model involves four key stages, namely *hazard identification*, *dose-response assessment*, *exposure assessment*, and *risk characterisation*. Briefly, the hazard identification comprises a literature-based review to identify which hazards, if any, are of most concern/most likely to pose a risk; the dose-response assessment is used to characterise the magnitude of effect likely to result from a specific exposure to specific hazards; the exposure assessment determines the extent to which receptors (i.e., soil biota) are exposed to the hazards of concern; finally the risk characterisation quantifies the level of risk, i.e., the probability that exposure to a specific hazard will result in a specific adverse outcome. The risk characterisation may then be

used to inform *risk management*, i.e., management of risk factors to reduce impacts of causative agents.

While the original 2018 assessment (Scottish Government, 2018) considered six categories of potentially hazardous agents, the current assessment will attempt to update three of these, namely microplastics and fibres, organic and other emerging chemicals, and pharmaceuticals and personal care products (PPCPs); Table 1.2. It was considered that these three categories were of most pertinence to the aims and objectives of Fidra and their existing projects.

Table 1.2 – Categories of potentially hazardous agents associated with sewage sludge

Categories of Potentially Hazardous Agent included in this Assessment
Microplastics and Fibres defined as synthetic polymers measuring less than 5 mm in diameter (i.e., largest dimension) – Section 2
Organic and other emerging chemicals including perfluoroalkyl and polyfluoroalkyl substances (PFAS), polychlorinated biphenyls (PCBs), dioxins and furans, flame retardants, plasticisers, synthetic phenolic compounds, etc. – Section 3
Pharmaceutical and Personal Care Products (PPCPs) including anti-inflammatories, anti-epileptics, antihistamines, selective serotonin reuptake inhibitors (SSRIs), antacids, antibiotics, caffeine, etc. – Section 4

1.2.1. Hazard identification

The approach adopted for the hazard identification was the same as that used for the 2018 assessment (Scottish Government, 2018) and used successfully in previous projects assessing risks from soil amendments (Longhurst et al., 2019; Hough et al., 2012). The 2018 assessment undertook a comprehensive process of identifying all possible hazards that might be present in sewage sludge, and then reducing this list by applying a series of filters (as per the methodology of Pollard et al., 2008) to reduce the long lists down to a shorter *priority* list. It was not the intention of the current assessment to fully repeat this hazard screening exercise, and the priority lists identified by Scottish Government (2018) were adopted by the current assessment. Concentrations of the priority hazards measured in sewage sludge were updated relative to the 2018 assessment. It was decided that potentially hazardous agents should be added to the 2018 priority lists where they had *emerged* post-2018 (n = 0) and where emerging chemicals of high priority were not included in the existing priority lists (n = 8, all classified as organic contaminants). Full details of the hazard screening approach can be found in Scottish Government 2018.

1.2.2. Exposure assessment

The purpose of the exposure assessment was to estimate the concentration of each hazardous agent that had the potential to come into contact with receptors, namely soil biota. Given the range and complexity of soil biota, no specific exposure routes (ingestion, contact, etc.) were defined or modelled. To this end, the exposure modelling estimated the predicted effect concentration (PEC, mg kg^{-1}) of each hazard in the soil. Estimating the PEC was achieved in one of two ways. For the microplastics and fibres, levels of microplastics in soil were determined as

the mass balance between the concentration introduced via the application of sewage sludge and the concentration of microplastics leaving the soil matrix (primarily via leaching). For the organic and emerging contaminants, and PPCPs, multi-media fugacity modelling was implemented. This approach aimed to estimate the partitioning of these pollutants within the soil, indicating what proportion is likely to remain in the soil matrix relative to what is leached or volatilised from the system.

To make the fugacity calculations, a three-compartment soil matrix consisting of: air (*A*, pore space), water (*W*, soil pore water), and sewage sludge amended soil (*SS*) was modelled as per Hough et al. (2012). If these compartments are assumed to be in equilibrium, the total mass of a specific potentially hazardous agent in the system (*T*, mol) can be described by:

$$T_x = V_A C_A + V_W C_W + V_{SS} C_{SS} \qquad \text{(Equation 1.1)}$$

Where T_x is the total mass of the agent of interest in the system, V represents the volume of each compartment (m^3), and C represents the concentration of each agent of interest in each compartment (mol m^{-3}). If the total mass of the agent of interest T_x is known, Equation 1.2 – 1.4 can be used to estimate the partitioning of the agent of interest between the three phases (A, W, SS) within the soil matrix.

To quantify the equilibrium between the compartments, the relationships between C_A, C_W, and C_{SS} were estimated by deriving partition coefficients (Equations 1.2 and 1.3):

$$C_A = HC_W \qquad \text{(Equation 1.2)}$$

$$C_{SS} = k_d \rho_b C_W \qquad \text{(Equation 1.3)}$$

Where H is the Henry's Law constant, K_d is the distribution coefficient in soil, L kg^{-1}, and ρ_b is the bulk density, kg L^{-1}.

The partition coefficients can be used to characterise the distribution of the agent of interest within the system (Equation 1.4):

$$T_x = V_A H C_W + V_W C_W + V_{SS} \rho_b k_d C_W \qquad \text{(Equation 1.4)}$$

Subsequently, the fractions of the agent of interest in water (W_x), air (A_x), and sludge-amended soil (SS_x) can be derived from Equation 1.5 – 1.7:

$$W_x = \frac{V_W}{(V_W + HV_A + \rho_b k_d V_{SS})} \qquad \text{(Equation 1.5)}$$

$$A_x = \frac{HV_A}{(V_W + HV_A + \rho_b k_d V_{SS})} \qquad \text{(Equation 1.6)}$$

$$SS_x = \frac{\rho_b k_d V_{SS}}{(V_W + HV_A \rho_b k_d V_{SS})} \qquad \text{(Equation 1.7)}$$

Finally, following *realistic worst-case* assumptions, the predicted effect concentration (PEC) was considered to be:

$$PEC = SS_x + (W_x - L) \qquad \text{(Equation 1.8)}$$

Where L represents loss from the system due to leaching.

To parameterise Equation 1.1, the volumetric composition of the sewage sludge-amended agricultural soil was derived using a simple *ploughing model* by assuming a unit area of 1 m^2

and a plough depth of 0.25 m in order to calculate a total working volume of 0.25 m^3. These, and other parameters are detailed in Table 1.3.

Table 1.3 – Input data set for the standard model for the calculation of the Phase I fugacity equilibrium (Equations 1.1 – 1.7), normalised to 0.25 m^3 soil. From Trapp and Legind (2011).

Symbol	Input [unit]	Value
	Fugacity equilibrium	
R_A	Sewage sludge application rate (fresh weight) [t ha^{-1}]	50
A	Unit area [m^2]	1
ρ_{SS}	Sewage sludge bulk density [kg l^{-1}]	0.6
V_{SS}	Applied sewage sludge volume [m^3]	0.0083
Pd	Plough depth [m]	0.25
V	Soil-air-water volume [m^3]	0.25
θ_A	Air content of soil	0.1
θ_W	Water content of soil	0.3
N	Total porosity	0.4
V_S	Soil volume [m^3]	0.15
$f_{oc,\ soil}$	Fraction of organic carbon in soil	0.05
$F_{oc,\ ss}$	Fraction of organic carbon in sewage sludge	0.5
P_a	Partial/vapour pressure [Pa]	9900
T	Temperature [K]	298
R	Gas constant [$J(molK)^{-1}$]	8.313
S	Solubility in water [mg l^{-1}]	300
ρ_b	Soil bulk density [kg l^{-1}]	1.6
K_{OW}	Octanol:Water partition coefficient [l kg^{-1}]	3.32
K_{OC}	Organic carbon distribution coefficient [l kg^{-1}] =10^(1.04 log(K_{OW}) – 0.84)	410.02
f_{OC}	Fraction of organic carbon in mixed system = (V_S $f_{OC,\ soil}$ + V_{SS} $f_{OC,\ SS}$)/(V_S + V_{SS})	0.068
k_d	Distribution coefficient in soil [l kg^{-1}] =K_{OC} f_{OC}	27.88
H	Henry's Law constant [dimensionless] =(P_aMW/(RT)/S)	1.75

Sewage sludge application rate based on expert judgement and previous investigations (WRAP, 2016; Hough et al., 2012).

1.2.3. Dose-response assessment

Dose-response data describe the magnitude of an outcome (response) in relation to the magnitude of a dose (exposure) of a specific agent. With the focus of this assessment being impacts on soil biota, the main interest was to be able to derive a predicted no-effect concentration for soil biota ($PNEC_{soil}$) with reasonable confidence. Primarily, for the agents studied, this meant identifying published $PNEC_{soil}$ values and critically appraising their derivation.

Values of PNEC are generally derived using two methods, the assessment factor (AF) method, and the species sensitivity distribution (SSD) method.

The AF method uses data from ecotoxicological studies where organisms are exposed to different levels of an agent, and the outcome (mortality, morbidity) is recorded. While there are some instances where in-situ data are available, most ecotoxicological studies are laboratory based. Several groups of organisms are used so that several exposures of different magnitude can be administered, and a dose-response curve obtained. The lowest toxicity value from this dose-response curve (e.g., the lethal concentration where 50% of organisms die (LC50), half the maximal effective concentration (EC50), or the no observed effect concentration (NOEC)) is then divided by an AF that typically ranges between 10 to 1000 depending on the quantity and quality of the dose-response data.

The values of AF are designed to account for uncertainties in the dose-response data. Many ecotoxicological experiments use levels of the agent of interest far greater than environmental levels in order to achieve an observable response in a limited number of study organisms. Hence care must be taken in extrapolating such data to environmentally relevant concentrations. Other considerations include the statistical power of the data used to derive the dose-response function (data quantity), and the accuracy and repeatability of the experimental set-up and analytical measurements (data quality).

The species sensitivity distribution (SSD) method is used when toxicity data are available for multiple species or multiple dose-response functions for a single species. In SSD, a statistical distribution of species sensitivities is created, and the 5th percentile of this distribution (termed the HC5) is often used as the PNEC. As with the AF method, additional assessment factors may also be applied to the HC5 to account for uncertainties in the derivation of the data incorporated into the SSD.

For soil biota specifically, i.e., $PNEC_{soil}$, values are derived ether directly from toxicity studies on soil-dwelling organisms (although the toxicity study might not be performed in soil but in some other media) and appropriate AFs applied to the lowest toxicity value or HC5 if multiple studies of multiple organisms are available. Where no direct measurements of toxicity are available for soil organisms, the equilibrium partitioning method (EPM) can be used to derive a $PNEC_{soil}$ from an aquatic $PNEC_{water}$. There are far more ecotoxicity tests conducted on aquatic organisms, partly due to historical concerns of chemical impact on aquatic ecosystems, and partly because such tests are relatively easy compared to terrestrial species. The $PNEC_{soil}$ can be estimated from $PNEC_{water}$ using Equation 1.9:

$$PNEC_{soil} = \left(\frac{K_{SW}}{\rho_b}\right) PNEC_{water} \times 1000 \qquad \text{(Equation 1.9)}$$

Where K_{SW} is the soil-water partition coefficient and ρ_b is the bulk density of wet soil.

It is important to note that for substances with $\log(K_{OW}) > 5$, additional factors may need to be considered due to potential direct ingestion of soil particles by organisms. Also, the EPM is generally not recommended for substances that may pose a high hazard to soil organisms, i.e., $\log(K_{OW}$ or $K_{OC}) > 5$ and LC50 or EC50 < 1 mg l^{-1} for aquatic species.

In all cases, data and PNEC values were sought using the concept of *principle source documents* as adopted by the Environment Agency (Defra, 2011). These are set out below in order of priority:

- Authoritative bodies in the UK (DEFRA), Scottish Government, Scottish Environment Protection Agency (SEPA), Environment Agency (EA))
- European Commission Committees

- Other national organisations (e.g. United States Environment Protection Agency (USEPA))
- Reports produced by authoritative organisations, but for different purposes

Where authoritative bodies, such as ECHA or USEPA had reviewed multiple ecotoxicological studies and multiple PNEC values, prior to deriving a consensus PNEC, these values were used in preference to single studies.

1.2.4. Risk Characterisation

For the majority of agents, *risk* was defined as the modelled probability that after spreading sewage sludge on agricultural land, soil biota would experience deleterious effects arising from direct contact or other forms of exposure (e.g. ingestion) with chemicals present in the sewage sludge.

Relative risk was calculated as a simple risk ratio (RR) of the exposure (predicted effect concentration in soil, PEC, mg kg^{-1}) to the relevant predicted no effect concentration ($PNEC_{soil}$); Equation 1.10. If the PEC exceeds the PNEC, we might expect to observe deleterious effects on the soil biota.

$$RR = {PEC_{soil}}/{PNEC_{soil}} \qquad \text{(Equation 1.10)}$$

1.2.5. Sensitivity analysis

A simple point sensitivity analysis was conducted to identify which input parameters the risk assessment is most sensitive to and therefore are most important to characterise accurately to reduce the output uncertainty. A point sensitivity analysis investigates how the model output changes relative to the change in each input parameter while keeping all the other inputs at a fixed level. The sensitivity can be expressed in different ways. Here, the sensitivity of the model output, O, to a parameter i taking the value xi is expressed through a normalised sensitivity index, SI, calculated as (Spitz and Moreno, 1996):

$$SI_i = \frac{|dO|}{\left({|dx_i|}/{x_i}\right)} \qquad \text{(Equation 1.11)}$$

1.3. References

Avery, L.M., Booth, P., Campbell, C.D., Tompkins, D. & Hough, R.L. 2012. Prevalence and survival of potential pathogens in source-segregated green waste compost. *Science of the Total Environment* **431:**128-138. https://doi.org/10.1016/j.scitotenv.2012.05.020

BAS (Biosolids Assurance Scheme). 2024. UK Assured Biosolids, UK BAS, https://assuredbiosolids.co.uk/

Beesley, L., Hough, R.L., Deacon, C.M. & Norton, G.J. 2018. The impacts of applying metal(loid) enriched wood ash to soils on the growth and elemental accumulation of rice. *Exposure and Health* **11:**311-324. https://doi.org/10.1007/s12403-018-0273-2

Black, S., Grieve, D. & Llewellyn A. (2016). Biosolids Assurance Scheme. 21st European Biosolids and Organic Resources Conference and Exhibition. Edinburgh, November 2016.

Defra. 2011. Guidelines for environmental risk assessment and management. Green Leaves III. https://assets.publishing.service.gov.uk/media/5a79d20540f0b66d161ae5f9/pb13670-green-leaves-iii-1111071.pdf

Fidra & Environmental Investigation Agency UK. 2024. Agricultural use of sewage sludge in the UK. Joint agency position paper, July 2024. https://www.fidra.org.uk/download/joint-position-paper-sludge/

Hough, R.L., Booth, P., Avery, L.M., Rhind, S., Crews, C., Bacon, J., Campbell, C.D. & Tompkins, D. 2012. Risk assessment of the use of PAS100 green composts in sheep and cattle production in Scotland. *Waste Management* **32:**117-130. https://doi.org/10.1016/j.wasman.2011.08.008

Hough, R.L., Fletcher, T., Leonardi, G.S., Goessler, W., Gnagnarella, P., Clemens, F., Gurzau, E., Koppova, K., Rudnai, P., Kumar, R. & Vahter, M. 2010. Lifetime exposure to arsenic in residential drinking water in Central Europe. *International Archives of Occupational and Environmental Health* **83:**471-481. https://doi.org/10.1007/s00420-010-0519-1

Hough, R.L., Stephens, C., Busby, A., Cracknell, J. & Males, B. 2006. Assessing and communicating risk with communities living on contaminated land. *International Journal of Occupational and Environmental Health* **12:**1-8. https://doi.org/10.1179/oeh.2006.12.1.1

Hough, R.L., Breward, N., Young, S.D., Crout, N.M.J., Tye, A.M., Moir, A.M. & Thornton, I. 2004. Assessing potential risk of heavy metal exposure from consumption of home-produced vegetables by urban populations. *Environmental Health Perspectives* **112:**215-221. https://doi.org/10.1289/ehp.5589

Longhurst, P.J., Tompkins, D., Pollard, S.J.T., Hough, R.L., Chambers, B., Gale, P., Tyrrel, S., Villa, R., Taylor, M., Wu, S., Sakrabani, R., Litterick, R., Snary, Leinster, P. & Sweet, N. 2019. Risk assessments for quality-assured, source-segregated composts and anaerobic digestates for a circular bioeconomy in the UK. *Environment International* **127:**253-266. https://doi.org/10.1016/j.envint.2019.03.044

Minh, N.D., Hough, R.L., Thuy, L.T., Nyberg, Y., Mai, L.B., Vinh, N.C., Khai, N.M. & Oborn, I. 2012. Assessing dietary exposure to cadmium in a metal recycling community in Vietnam: Age and gender aspects. *Science of the Total Environment* **416:**164-171. https://doi.org/10.1016/j.scitotenv.2011.11.068

Mitchell, K., Moreno-Jimenez, E., Jones, R., Zheng, L., Trakal, L., Hough, R.L. & Beesley, L. 2020. Mobility of arsenic, chromium and copper arising from soil application of stabilised aggregates made from contaminated wood ash. *Journal of Hazardous Materials* **393:**112479. https://doi.org/10.1016/j.jhazmat.2020.122479

Mollon, L.C., Norton, G.J., Trakal, L., Moreno-Jimenez, E., Elouali, F.Z., Hough, R.L. & Beesley, L. 2016. Mobility and toxicity of heavy metal(loid)s arising from contaminated wood ash application to a pasture grass soil. *Environmental Pollution* **218:**419-427. https://doi.org/10.1016/j.envpol.2016.07.021

Pollard, S.J.T., Hickman, G.A.W., Irving, P., Hough, R.L., Gauntlett, D.M., Howson, S.F., Hart, A., Gayford, P. & Gent, N. 2008. Exposure assessment of carcass disposal options in the event of a notifiable exotic animal disease: application to avian influenza virus. *Environmental Science & Technology* **42:**3145-3154. https://doi.org/10.1021/es702918d

Qvarforth, A., Lundgren, M., Rodushkin, I., Engstrom, E., Paulukat, C., Hough, R.L., Moreno-Jimenez, E., Beesley, L., Trakal, L. & Augustsson, A. 2022. Future food contaminants: as assessment of the plant uptake of technology-critical elements verses traditional metal contaminants. *Environment International* **169:**107504.

Scottish Government. 2018. The impacts on human health and environment arising from the spreading of sewage sludge to land CR/2016/23: research reports. https://www.gov.scot/collections/sewage-sludge-research/

Scottish Government. 2016. Review of the storage and spreading of sewage sludge on land in Scotland (The Sludge Review). Final Recommendations. https://www.gov.scot/publications/review-storage-spreading-sewage-sludge-land-scotland-sludge-review-final/

SNIFFER (2008). Human health and the environmental impacts of using sewage sludge on forestry and for restoration of derelict land. UKLQ09 August 2008.

Spitz, K. & Moreno, J. 1996. A practical guide to groundwater and solute transport modelling. John Wiley & Sons, Inc., New York. https://www.wiley.com/en-us/A+Practical+Guide+to+Groundwater+and+Solute+Transport+Modeling-p-9780471136873

SRUC (2013). Optimising the application of bulky organic fertilisers. Technical Note TN650. April 2013. https://www.monitorfarms.co.uk/wp-content/uploads/2020/10/ferttechnotebill.pdf

Stubberfield, J., Troldborg, M., Ander, L., Crout, N., Young, S.D. & Hough, R.L. 2022. Exercise, urban food production, preparation and consumption: Implications, benefits and risks to grow-your-own (GYO) gardeners. *Agronomy* **12:**181. https://doi.org/10.3390/agronomy12010181

Trapp, S. & Legind, C.N. 2011. Uptake of organic contaminants from soils into vegetables and fruits. In: Swartjes, F.A. (Ed.), Dealing with contaminated sites – from theory towards practical application. Springer, pp. 369-408. https://link.springer.com/book/10.1007/978-90-481-9757-6

Troldborg, M., Duckett, D., Allan, R., Hastings, E. & Hough, R.L. 2017. A risk-based approach for developing standards for irrigation with reclaimed water. *Water Research* **126:**372-384. https://doi.org/10.1016/j.watres.2017.09.041

WRAP. 2016. Risk assessment for the use of PAS100 green composts in Scottish livestock production. Report OAV021-004. Waste and Resources Action Programme, Banbury, UK; available on request from risk.assessments@wrap.org.uk

Zhang, Z., Le Velly, M., Rhind, S.M., Kyle, C.E., Hough, R.L., Duff, E.I. & McKenzie, C. 2015. A study on temporal trends and estimates of fate of Bisphenol A in agricultural soils after sewage sludge amendment. *Science of the Total Environment* **515:**1-11. https://doi.org/10.1016/j.scitotenv.2015.01.053

2. Microplastics and Fibres

2.1. Background

Microplastics (MPs) are widely defined as plastic particles smaller than 5 mm (GESAMP, 2015), and arise from various sources such as plastic products, cosmetics, lubricants, additives, etc., as well as in the form of synthetic fibres shed during the washing of textiles (Li et al., 2018). Microplastics can be primary MPs, i.e., manufactured to desired specifications (e.g., microbeads in facial scrubs), or secondary MPs, i.e., those derived from the degradation of larger plastics. Of these, secondary MPs are by far the most abundant in the environment (Liu et al., 2019).

Microplastics have been ubiquitously detected in all ecosystems (Plastics Europe, 2023). Globally, 400 million tonnes of plastics were produced in 2022, while only 38 million tonnes (9.6%) were recycled or treated (Plastics Europe, 2023). If these patterns of manufacture and recycling continue, we can expect ~12 billion tonnes of plastic waste to end up in landfill and natural environments by 2050 (UNEP-WCMC, IUCN and NGS, 2020). Additionally, MPs released into the environment, can adsorb certain inorganic and organic pollutants which can be re-released following degradation by natural agents (water, sunlight, microorganisms, etc.).

Despite much of the microplastics research focussing on the marine environment, the main release pathway for plastics is to the terrestrial environment. It is estimated that globally, plastic pollutants are 4 – 23 times more abundant in soils than in oceans (Horton et al., 2017). In some land-locked European countries, emissions of plastics to soils have been estimated to be 40 times higher than to surface waters (Kawecki & Nowack, 2019). In agriculture, MPs find their way into soils primarily via sewage sludge application (Corradini et al., 2019; Nizzetto et al., 2016) as well as other agricultural amendments (e.g., composts and digestates), plastic waste arising from agriculture, and general littering (Braun et al. 2023).

There have been various studies looking at the presence of MPs in wastewater treatment (Habib et al., 2020; Raju et al., 2020; Zhao et al., 2018). It has been reported that ~80 – 90% of MPs passing through wastewater treatment plants are retained in the generated biosolids that are widely applied as fertilisers, thus representing a substantive potential source of contamination of the soil environment (Alvim et al., 2020a, b; Campo et al., 2019; He et al., 2018; Okoffo et al., 2019; Murphy et al., 2016; Talvitie et al., 2017). The presence of MPs in sewage sludge derived from wastewater treatment has only been studied to a limited extent with most studies focusing on specific water treatment processes, or more general identification and detection methods (Azizi et al., 2021; Rolsky et al., 2020; Zhang & Chen, 2020).

Microplastics can alter soil properties, microbial activity (de Souza Machado et al., 2017), enzyme activity (Zhao et al., 2018), and the composition of microbial communities (Seeley et al., 2020), and their leachates can negatively impact soil biota (Kim et al., 2020). Direct ingestion of MPs by soil organisms (e.g., earthworms, Lwanga et al., 2017; snails, Panebianco et al., 2019; collembola, Maaß et al., 2017) can cause deleterious effects on the digestive system, including damage to the oesophagus as well as intestinal obstruction (Ju et al., 2019; Song et al., 2019; Zhu et al., 2018). Overall, MPs have been reported to affect reproduction, growth, and survival of soil organisms (Cao et al., 2017; Lahive et al., 2019).

Despite the potential negative impacts described above, risk assessments of MPs in soil systems (as opposed to aquatic systems) have been very limited. Indeed, the 2018 assessment (Scottish Government, 2018), reported it was not possible to undertake a formal risk analysis of microplastics in soils due to lack of key information on fate and transport of microplastics, but also due to lack of dose-response data for humans (the focus of the 2018 assessment). The current analysis focusses on ecological risks, which gets around some of these limitations as ecotoxicological data are available for MPs for a wide range of species.

In addition, despite calls for standardisation (e.g., Sa'adu & Farsang, 2023), there are no agreed reporting standards for MPs. This means that data on levels of MPs in soils can be reported on either a mass-, i.e., mg kg^{-1}, or number-, i.e., n kg^{-1} basis. There are no standard methodologies for determining microplastics either with a variety of methodologies used in isolation or in tandem including density separation, spectroscopy, pyrolysis-GC-MS, TED-GC-MS, fluorescence microscopy, visual sorting, enzymatic digestion, chemical digestion, FTIR, Raman spectroscopy (Ziajahromi & Leusch, 2022). The units of characterisation, i.e., mass vs. number, are determined by the analytical approach. Any risk analysis methodology needs to be able to handle the considerable uncertainty in any data reporting concentration of MPs in soil.

Similarly, ecotoxicological data are derived from both experimental and in-situ evidence, uses a variety of species, and derives a variety of different toxicological end points. Toxicological studies characterise toxic dose using different points of departure from and dose-response relationship, including no-observed effect concentrations (NOEC), lowest observed effect concentrations (LOEC), and highest observed no effect concentrations (HONEC). All these points of departure have different interpretations and implications for the toxicological nature of the exposure. Any risk analysis methodology needs to be able to handle considerable uncertainty to take advantage of all relevant data.

Since the publication of the 2018 assessment (Scottish Government, 2018), there have been two substantive risk assessments of MPs in soils published that provide a complete environmental risk assessment. Jacques & Prosser (2021) derived environmental exposure and species sensitivity distributions from either no-observed effect concentrations (NOECs) or lowest observed effect concentrations (LOECs), with risk characterised as the overlap between these two distributions. Their results indicated that current levels of MPs in soils might affect 7 – 28% of species 5% of the time (Jaques & Prosser, 2021). Tunali et al. (2023), extend the methodology of Jaques & Prosser (2021) by incorporating probabilistic species sensitivity distributions to account for the additional variabilities introduced by using different dose-response end points, i.e., NOEC vs. LOEC or other end points such as highest observed no effect concentration, HONEC. This methodology had previously been proven advantageous in a risk assessment of MPs in aquatic systems (Adam et al., 2021). Thus, Tunali et al. (2023) were able to use a far wider range of data sources than Jaques & Prosser (2021).

Importantly for the current study, Tunali et al. (2023) derived predicted no effect concentrations (PNECs) for MPs toxicity to soil organisms both for mass and number of MPs. The PNECs were derived from 63 toxicity values across 16 different terrestrial species and as such are probably the most comprehensive PNECs available for MPs impact on soil biota. We adopt these PNECs in the current study to assess the additional risk associated with MPs present in sewage sludge applied to land.

2.2. Risk Assessment

Due to the limited available information on the potential risk posed by MPs in the agricultural environment on human health (i.e., exposure to PMs in food products), here the risk assessment focusses entirely on potential risk to soil ecology.

Data on levels of microplastics in treated sewage sludge were collated from literature. Data included were those incorporated into the 2018 assessment (Scottish Government, 2018) plus additional data published since 2018 up until June 2024. Studies providing convincing measurements of MPs in sewage sludge are still relatively sparse, with 6 studies included here representing 61 separate measurements (Table 2.1).

Table 2.1 - Levels of microplastics and fibres measured in sewage sludge (*cf.* 2018 assessment that characterised *high* levels as 120 n kg^{-1} (Scottish Government, 2018)).

Reference	Country	Microplastic	Range	Units
Mahon et al., 2017	Ireland	Fibres	4007 – 13675	n kg^{-1}
		Films	11 – 366	n kg^{-1}
		Primary	0 - 89	n kg^{-1}
		Secondary	511 – 5228	n kg^{-1}
		Other	0 - 178	n kg^{-1}
Carr et al., 2016	USA	Generic	1000	n kg^{-1}
Leslie et al., 2017	Netherlands	Fibres	9.7 - 26	n kg^{-1}
Corradinni et al., 2019	Chile	Generic	1.78 – 50.2	mg kg^{-1}
Ziajahromi & Leusch, 2022	*Various*	Generic	$3.16x10^{3}$ - $1.58x10^{5}$	n kg^{-1}
Karthika et al., 2024	India	Generic	800 – 1650	n kg^{-1}

A simple *ploughing model* was established using the same assumptions as those used to derive most quantitative risk estimates in Scottish Government (2018); see Section 1.2.2.

In addition to the assumptions adopted from Scottish Government (2018), MPs were considered in their totality. I.e., number or mass of MPs was not split by MP type. This assumption was maintained throughout all processes considered by this assessment, including leaching and exposure of biota to microplastics. This is similar to the approach used by Tunali et al. (2023).

Input hazard variables (where *hazard* refers to levels of potentially harmful agents in sewage sludge applied to land), as the number or mass of microplastic particles per kg applied sewage sludge, were defined as the 95th %ile measured values from the available data, i.e., 15,325 n kg^{-1} and 39 mg kg^{-1}(Table 2.1). This follows the same logic as Scottish Government (2018) that assumed 95th %ile hazard data to be a *realistic worst-case* whilst avoiding the most uncertain data at the extremes of the distribution.

Leaching rate of microplastics from soils was considered to be 0.2% d^{-1} based on the experimental work by Zhang et al. (2022). While the leaching rate presented by Zhang et al. (2022) was calculated on a number basis, it was assumed that the same rate of leaching would apply on a per mass basis. It was also assumed that leached MPs were removed from the system of interest, and hence no longer posed an exposure to soil biota but it should be noted that this mobilisation potentially poses risks to water quality.

When simulating the impacts of a single application of sewage sludge, it was necessary to identify the most appropriate time to take a *snapshot* and use that point in time to derive a risk

estimate from. Exposure to soil biota was assessed at 21 days post-application of sewage sludge, as representative of compliance with the safe sludge matrix no-grazing period, after which the introduction of livestock is likely to influence soil ecology. This assumption was removed when simulating multiple applications of sewage sludge over time, where one application was assumed per year with a constant leaching rate unadjusted for influence of livestock.

2.2.1. Risk Characterisation

Relative risk (RR) was defined as the predicted concentration (number or mass-based) of MPs added to the soil following application of sewage sludge relative to published *safe* values. In this instance, the predicted no-effect concentrations (PNEC) reported by Tunali et al. (2023) were adopted as the comparators. Specifically, the 25th%ile PNEC values derived from data that included highest observed no-effect concentrations (HONECs) as the point of departure. These were selected in part due to the conservative nature of 25th%ile estimates (*realistic worst case*), and in part because toxicity of fibres was only described using HONECS. Our input data reported fibres as the most prevalent form of MP in sewage sludge, thus PNECS derived from HONEC data enabled us to evaluate risks associated with the entirety of the input data. In summary, the PNECs used were (Table 2.2):

Table 2.2 - Predicted no effect concentrations (PNEC) adopted as *safe* levels for comparison with estimated exposures.

Unit of measurement	PNEC
number/kg	2100 (Tunali et al., 2023)
mg/kg	8 (Tunali et al, 2023)

Figure 2.1 displays the estimated *additional* relative risk to soil biota following a single application of sewage sludge to land for a range of application rates (25 – 50 t ha^{-1}). Relative risk is estimated using both number-based (n kg^{-1}) and mass-based (mg kg^{-1}) MPs data and PNECs. Relative risk (RR) is a relative index, as such it is unitless. Where RR > 1.0 (unity) exposure (PEC) exceeds the PNEC and it is therefore probable that an appreciable negative impact will be manifest. Intuitively, risk estimates increase with increasing sewage sludge application rate. Mass-based estimates are lower than number-based ones by ~0.1 (although this difference does increase slightly with increasing application rate). Even at a very high rate of application, number-based estimates of RR do not exceed 0.4 (i.e., less than unity). A single application of sewage sludge therefore does not present an appreciable risk to soil biota at typical application rates.

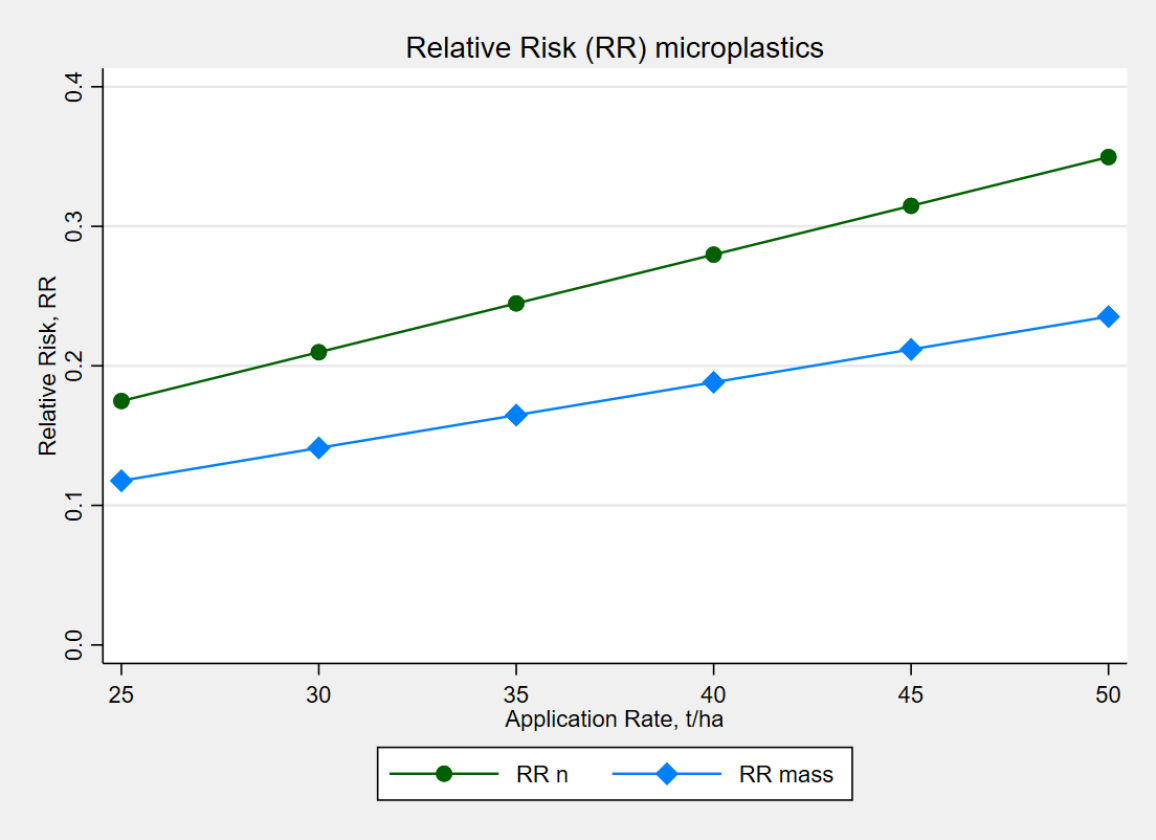

Figure 2.1 - Relative risk (RR) to soil biota from microplastics following a single application of sewage sludge at a range of application rates (25 – 50 t ha^{-1}). Relative risk was defined as the ratio of exposure to MPs relative to PNEC values published by Tunali et al. (2023). Relative risk was calculated 21 days after sewage sludge application. Leaching of MPs from the soil system was assumed to be 0.2 % d^{-1} (Zhang et al., 2022).

Figure 2.2 shows the number of applications of sewage sludge necessary for the additional relative risk to exceed unity, i.e., RR > 1.0. Where RR exceeds 1.0, exposure is greater than the PNEC values, and an appreciable deleterious effect on soil biota is probable. Again, number-based estimates tend to characterize the relative risk more strongly indicating that between 2 – 6 applications would be required for RR to exceed 1.0, depending on application rate. Mass-based estimates are more optimistic, suggesting somewhere between 4 and 10 applications would be required before RR exceeded unity. In this simulation, it was assumed that a single application of sewage sludge is applied each year with 0.2 % leaching of microplastics from the soil system per day (Zhang et al., 2022). Other routes of MP removal, such as uptake by crops and transportation via wind erosion, were not included in this assessment as it was assumed that bare soil under stable meteorological conditions was the *realistic worst-case*. This is an over-simplification of reality as sewage sludge would not be applied to land that was not being used for production, but this assumption maintained the requirement to support precautionary decision making whilst avoiding difficult to quantify sources of variation such as crop type, crop and land management, irrigation, etc.

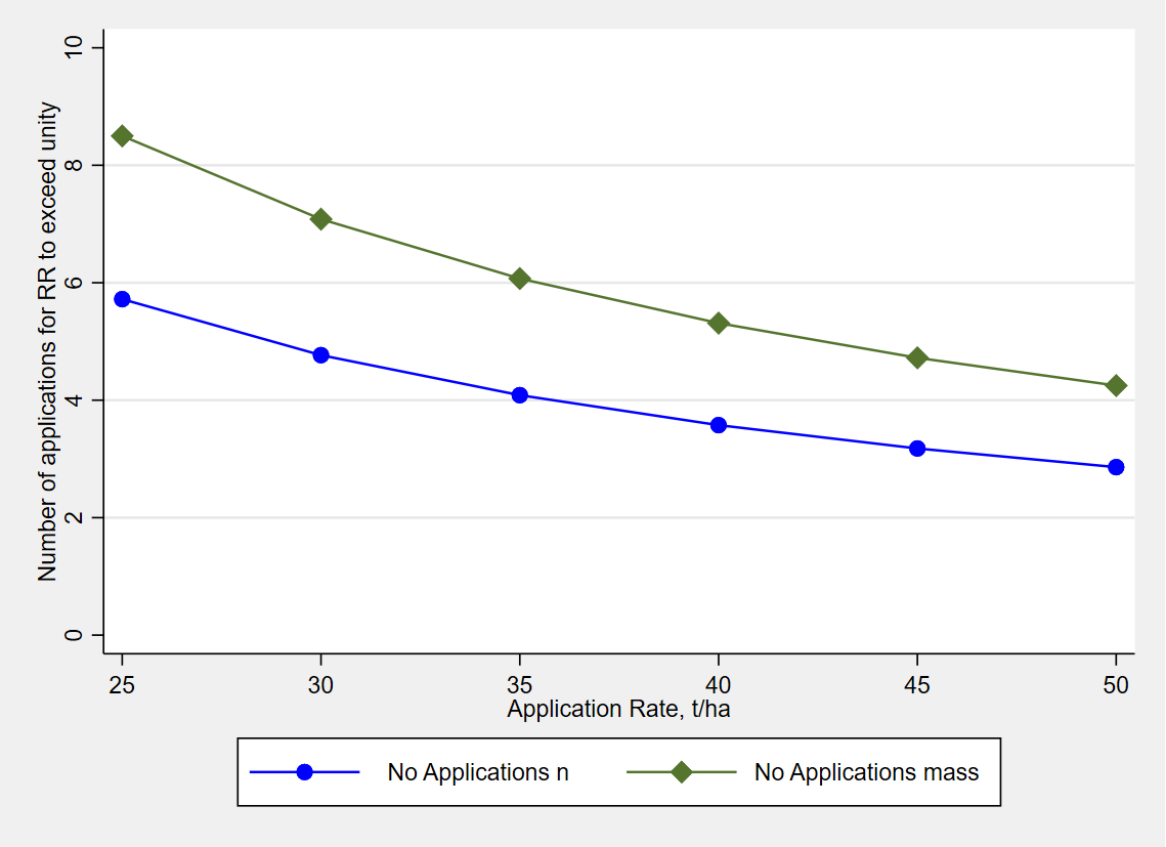

Figure 2.2 - Number of applications of sewage sludge at different application rates (25 – 50 t ha^{-1}) required before relative risk (RR) from microplastics exceeds unity, i.e., exposure to microplastics exceeds published *safe* PNEC values. Relative risk was defined as the ratio of exposure to MPs relative to PNEC values published by Tunali et al. (2023). One application of sewage sludge every 365.25 days was assumed and leaching of MPs from the soil system was assumed to be 0.2 % d^{-1} (Zhang et al., 2022).

2.3. Conclusions

Based on the simulations presented, a single application of sewage sludge poses minimal risk to soil biota from microplastics. Having said this, the simulations also indicate that it does not take many repeat applications (2 to 10) for risks to soil biota associated with MPs to become appreciable. This is in part because accumulation of MPs exceeds their removal from the system due to leaching. While specific application frequencies are crop- and soil-dependent, typically farmers in Scotland (and the rest of the UK) would apply sewage sludge annually or less frequently to cereal crops (depending on soil tests and crop rotation), and on a longer cycle (~every 2 – 3 years) to grassland and other crops with restrictions. Thus, accumulation of MPs over 5 – 10 years' timeframe becomes the consideration for environmental protection and is analogous to accumulation of potentially toxic metals which many sewage sludge regulations and best practice are built around. These results highlight that sewage sludge provides a significant route for MPs to reach soils and the wider environment. Removal of MPs from wastewater during treatment is imperative to prevent accumulation of MPs in our soil resource but source control and restricting use of plastics in products is an important part of the solution. In addition, it should be noted that MPs leaving the soil system via leaching will be

entering the aquatic environment where their negative impacts on water quality and sensitive aquatic organisms are well characterised (Ali et al., 2024).

2.4. References

Adam, V., von Wyl, A. & Nowack, B. 2021. Probabilistic environmental risk assessment of microplastics in marine habitats. *Aquatic Toxicology* **230:**105689. https://doi.org/10.1016/j.aquatox.2020.105689

Ali, N., Khan, M.H., Ali, M., Sidra, Ahmad, S., Khan, A., Nabi, G., Ali, F., Bououdina, M. & Kyzas, G.Z. 2024. Insight into microplastics in the aquatic ecosystem: Properties, sources, threats and mitigation strategies. *Science of the Total Environment* **913:**169489. https://doi.org/10.1016/j.scitotenv.2023.169489

Alvim, C.B., Mendoza-Roca, J.A. & Bes-Piá, M.A. 2020a. Wastewater treatment plant as microplastics release source - quantification and identification techniques. *Journal of Environmental Management* **255:**109739. https://doi.org/10.1016/j.jenvman.2019.109739

Alvim, C.B., Bes-Piá, M.A. & Mendoza-Roca, J.A. 2020b. Separation and identification of microplastics from primary and secondary effluents and activated sludge from wastewater treatment plants. *Chemical Engineering Journal* **402:**126293. https://doi.org/10.1016/j.cej.2020.126293

Azizi, S.M.M., Hai, F.I., Lu, W., Al-Mamun, A. & Dhar, B.R. 2021. A review of mechanisms underlying the impacts of (nano) microplastics on anaerobic digestion. *Bioresource Technology* **329:**124894. https://doi.org/10.1016/j.biortech.2021.124894

Braun, M., Mail, M., Krupp, A.E. & Amelung, W. 2023. Microplastic contamination of soil: Are input pathways by compost overridden by littering? *Science of the Total Environment* **855:**158889. https://doi.org/10.1016/j.scitotenv.2022.158889

Campo, P., Holmes, A. & Coulon, F. 2019. A method for the characterization of microplastics in sludge. *MethodsX* **6:**2776-2781. https://doi.org/10.1016/j.mex.2019.11.020

Cao, D., Wang, X., Luo, X., Liu, G. & Zheng, H. 2017. Effects of polystyrene microplastics on the fitness of earthworms in an agricultural soil. *IOP Conference Series: Earth and Environmental Science* **61:**012148. https//doi.org/10.1088/1755-1315/61/1/012148

Carr, S.A., Liu, J. & Tesoro, A.G. 2016. Transport and fate of microplastic particles in wastewater treatment plants. *Water Research* **91:**174-182. https://doi.org/10.1016/j.watres.2016.01.002

Corradini, F., Meza, P., Eguiluz, R., Casado, F., Huerta-Lwanga, E. & Geissen, V. 2019. Evidence of microplastic accumulation in agricultural soils from sewage sludge disposal. *Science of the Total Environment* **671:**411-420. https://doi.org/10.1016/j.scitotenv.2019.03.368

de Souza Machado, A.A., Kloas, W., Zarfl, C., Hempel, S. & Rillig, M.C. 2017. Microplastics as an emerging threat to terrestrial ecosystems. *Global Change Biology* **24:**1405-1416. https://doi.org/10.1111/gcb.14020

GESAMP. 2015. Sources, fate and effects of microplastics in the marine environment: a global assessment. P.J. Kershaw (Ed.), IMO/FAO/UNESCO-IOC/UNIDO/WMO/IAEA/UN/UNEP/UNDP Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection. Rep. Stud. GESAMP No. 90, 96p. http://www.gesamp.org/publications/reports-and-studies-no-90

Habib, R.Z., Thiemann, T., Al Kendi, R. 2020. Microplastics and wastewater treatment plants – a review. *Journal of Water Resource and Protection* **12:**1-35. https://doi.org/10.4236/jwarp.2020.121001

He, D., Luo, Y., Lu, S., Liu, M., Song, Y. & Lei, L. 2018. Microplastics in soils: Analytical methods, pollution characteristics and ecological risks. *TrAC Trends in Analytical Chemistry* **109:**163-172. https://doi.org/10.1016/j.trac.2018.10.006

Horton, A.A., Walton, A., Spurgeon, D.J., Lahive, E. & Svendsen, C. 2017. Microplastics in freshwater and terrestrial environments: Evaluating the current understanding to identify the knowledge gaps and future research priorities. *Science of the Total Environment* **586:**127-141. https://doi.org/10.1016/j.scitotenv.2017.01.190

Jacques, O. & Prosser, R.S. 2021. A probabilistic risk assessment of microplastics in soil ecosystems. *Science of the Total Environment* **757:**143987. https://doi.org/10.1016/j.scitotenv.2020.143987

Ju, H., Zhu, D. & Qiao, M. 2019. Effects of polyethylene microplastics on the gut microbial community, reproduction and avoidance behaviours of the soil springtail, *Folsomia candida*. *Environmental Pollution* **247:**890-897. https://doi.org/10.1016/j.envpol.2019.01.097

Karthika, S., Sivasubramanian, K., Dhevagi, P., Thavamani, P., Rajkishore, S.K., Amrutha, V.M., Sudhakaran, R., Chitra, N. & Lakshmanan, A. 2024. Distribution pattern and risk assessment of microplastics contamination in different agricultural systems. *International Journal of Environmental Analytical Chemistry* 1-20. https://doi.org/10.1080/03067319.2024.2309553

Kawecki, D. & Nowack, B. 2019. Polymer-specific modelling of the environmental emissions of seven commodity plastics as macro- and microplastics. *Environmental Science & Technology* **53:**9664-9676. https://doi.org/10.1021/acs.est.9b02900

Kim, S.W., Waldman, W.R., Kim, T.Y. & Rillig, M.C. 2020. Effects of different microplastics on nematodes in the soil environment: tracking the extractable additives using an ecotoxicological approach. *Environmental Science & Technology* **54:**13868-13878. https://doi.org/10.1021/acs.est.0c04641

Lahive, E., Walton, A., Horton, A.A., Spurgeon, D.J. & Svendsen, C. 2019. Microplastic particles reduce reproduction in the terrestrial worm *Enchytraeus crypticus* in a soil exposure. *Environmental Pollution* **255:**113174. https://doi.org/10.1016/j.envpol.2019.113174

Leslie, H.A., Brandsma, S.H., van Velzen, M.J.M. & Vethaak, A.D. 2017. Microplastics en route: Field measurements in the Dutch river delta and Amsterdam canals, wastewater treatment plants, North Sea sediments and biota. *Environment International* **101:**133-142. https://doi.org/10.1016/j.envint.2017.01.018

Li, X., Chen, L., Mei, Q., Dong, B., Dai, X., Ding, G. & Zeng, E.Y. 2018. Microplastics in sewage sludge from the wastewater treatment plants in China. *Water Research* **142:**75-85. https://doi.org/10.1016/j.watres. 2018.05.034

Liu, H., Yang, X., Liang, C., Li, Y., Qiao, L., Ai, Z & Liu, G. 2019. Interactive effects of microplastics and glyphosate on the dynamics of soil dissolved organic matter in a Chinese loess soil. *Catena* **182:**104177. https://doi.org/10.1016/j.catena.2019.104177

Lwanga, E.H., Vega, J.M., Quej, V.K., Chi, J de los A., del Cid, L.S., Chi, C., Segura, G.E., Gertsen, H., Salánki, T., van der Ploeg, M., Koelmans, A.A. & Geissen, V. 2017. Field evidence for transfer of plastic debris along a terrestrial food chain. *Scientific Reports* **7:**14071. https://doi.org/10.1038/s41598-017-14588-2

Maaß, S., Daphi, D., Lehmann, A. & Rillig, M.C. 2017. Transport of microplastics by two collembolan species. *Environmental Pollution* **225:**456-459. https://doi.org/10.1016/j.envpol.2017.03.009

Mahon, A.M., O'Connell, B., Healy, M.G., O'Connor, I., Officer, R., Nash, R. & Morrison, L. 2016. Microplastics in sewage sludge: Effects of treatment. *Environmental Science & Technology* **51:**810-818. https://doi.org/10.1021/acs.est.6b04048

Murphy, F., Ewins, C., Carbonnier, F. & Quinn, B. 2016. Wastewater treatment works (WwTW) as a source of microplastics in the aquatic environment. *Environmental Science & Technology* **50:**5800-5808. https://doi.org/10.1021/acs.est.5b05416

Nizzetto, L., Futter, M. & Langaas, S. 2016. Are agricultural soils dumps for microplastics of urban origin? *Environmental Science & Technology* **50:**10777-10779. https://doi.org/10.1021/acs.est.6b04140

Okoffo, E.D., O'Brien, S., O'Brien, J.W., Tscharke, B.J. & Thomas, K.V. 2019. Wastewater treatment plants as a source of plastics in the environment: a review of occurrence, methods for identification, quantification and fate. *Environmental Science: Water Research & Technology* **5:**1908-1931. https://doi.org/10.1039/C9EW00428A

Panebianco, A., Nalbone, L., Giarratana, F. & Ziino, G. 2019. First discoveries of microplastics in terrestrial snails. *Food Control* **106:**106722. https://doi.org/10.1016/j.foodcont.2019.106722

Plastics Europe. 2023. Plastics – the fast facts 2023. Plastics Europe, Brussels. Available at Plastics – the fast Facts 2023 • Plastics Europe

Raju, S., Carbery, M., Kuttykattil, A., Senthirajah, K., Lundmark, A., Rogers, Z. & Palanisami, T. 2020. Improved methodology to determine the fate and transport of microplastics in a secondary wastewater treatment plant. *Water Research* **173:**115549. https://doi.org/10.1016/j.watres.2020.115549

Rolsky, C., Kelkar, V.P. & Halden, R.U. 2020. Nationwide mass inventory and degradation assessment of plastic contact lenses in US wastewater. *Environmental Science & Technology* **54:**12102-12108. https://doi.org/10.1021/acs.est.0c03121

Sa'adu, I. & Farsang, A. 2023. Plastic contamination in agricultural soils: a review. *Environmental Sciences Europe* **35:**13. https://doi.org/10.1186/s12302-023-00720-9

Scottish Government. 2018. The impacts on human health and environment arising from the spreading of sewage sludge to land CR/2016/23: research reports. https://www.gov.scot/collections/sewage-sludge-research/

Seeley, M.E., Song, B., Passie, R. & Hale, R.C. 2020. Microplastics affect sedimentary microbial communities and nitrogen cycling. *Nature Communications* **11:**2372. https://doi.org/10.1038/s41467-020-16235-3

Song, Y., Cao, C., Qiu, R., Hu, J., Liu, M., Lu, S., Shi, H., Raley-Susman, K.M. & He, D. 2019. Uptake and adverse effects of polyethylene terephthalate microplastics fibres on terrestrial snails (*Achatina fulica*) after soil exposure. *Environmental Pollution* **250:**447-455. https://doi.org/10.1016/j.envpol.2019.04.066

Talvitie, J., Mikola, A., Setälä, O., Heinonen, M & Koistinen, A. 2017. How well is microliter purified from wastewater? – A detailed study on the stepwise removal of microliter in a tertiary level waste-water treatment plant. *Water Research* **109:**164-172. https://doi.org/10.1016/j.watres.2016.11.046

Tunali, M., Adam, V. & Nowack, B. 2023. Probabilistic environmental risk assessment of microplastics in soils. *Geoderma* **430:**116315. https://doi.org/10.1016/j.geoderma.2022.116315

UNEP-WCMC, IUCN and NGS. 2020. Protected Planet Report 2020. UNEP-WCMC, IUCN and NGS: Cambridge UK; Gland, Switzerland; and Washington, D.C., USA. https://livereport.protectedplanet.net/

Zhang, Z. & Chen, Y. 2020. Effects of microplastics on wastewater and sewage sludge treatment and their removal: a review. *Chemical Engineering Journal* **382:**122955. https://doi.org/10.1016/j.cej.2019.122955

Zhang, X., Chen, Y., Li, X., Zhang, Y., Gao, W., Jiang, J., Mo, A. & He, D. 2022. Size/shape-dependent migration of microplastics in agricultural soil under simulative and natural rainfall. *Science of the Total Environment* **815:**152507. http://dx.doi.org/10.1016/j.scitotenv.2021.152507

Zhao, J., Liu, L., Zhang, Y., Wang, X. & Wu, F. 2018. A novel way to rapidly monitor microplastics in soil by hyperspectral imaging technology and chemometrics. *Environmental Pollution* **238:**121-129. https://doi.org/10.1016/j.envpol.2018.03.026

Zhu, D., Chen, Q-L., An, X-L., Yang, X-R., Christie, P., Ke, X., Wu, L-H. & Zhu, Y-G. 2018. Exposure of soil collembolans to microplastics perturbs their gut microbiota and alters their isotopic composition. *Soil Biology and Biochemistry* **116:**302-310. https://doi.org/10.1016/j.soilbio.2017.10.027

Ziajahromi, S. & Leusch, F.D.L. 2022. Systematic assessment of data quality and quality assurance/quality control (QA/QC) of current research on microplastics in biosolids and agricultural soils. *Environmental Pollution* **294:**118629. https://doi.org/10.1016/j.envpol.2021.118629

3. Organic and other emerging chemicals

3.1. Background

Organic contaminants are carbon-based chemical substances that persist in the environment, bioaccumulate through the food web, and pose a risk of causing adverse effects to biota. Thus, when present in soils, organic contaminants present a threat to soil function and quality. A sub-set of organic contaminants are known as *emerging* chemicals, for the purposes of this report *emerging* chemicals are those that are currently not regulated and does not necessarily indicate that the chemical is recently discovered. Within this study, organic and emerging chemicals include industrial chemicals (such as polychlorinated biphenyls – PCBs), unintentional by-products of industrial processes (such as dioxins and furans), products of incomplete combustion (e.g., polycyclic aromatic hydrocarbons – PAHs), flame retardants (such as polybrominated diphenyl ethers – PBDEs), plasticisers (such as benzyl butyl phthalate – BBP), and chemicals used widely in consumer products (e.g., perfluoroalkyl and polyfluoroalkyl substances – PFAS).

Given the diversity of chemicals, organic contaminants can enter wastewater treatment plants (WWTPs) from a wide variety of sources. These include urban and agricultural run-off, domestic wastewater, industrial point source discharges, and via atmospheric deposition. Organic contaminants are typically lipophilic (fat soluble) and hydrophobic, thus sorption to sewage sludge solids is the primary pathway for their removal from wastewater. Thus, the application of sewage sludge to land, whether on to the soil surface or via incorporation into the topsoil, will directly expose the soil and soil biota to organic contaminants. Some non-persistent organic pollutants, such as lower molecular mass PAHs, can be degraded during composting and through natural attenuation in the soil (Brandli et al., 2007). In contrast, persistent organic pollutants (POPs), such as PCBs or polychlorinated dibenzodioxins and furans (PCDD/Fs) can accumulate in soil when contaminated sewage sludge is applied repeatedly (Umlauf et al., 2011).

More recently there has been considerable focus on the persistent synthetic organofluorine chemicals PFAS. The PFAS are used in many applications (surfactants, protective treatments, polymer manufacturing) and due to their widespread presence in consumer products, WWTPs have become a sink for PFAS. They resist degradation by conventional wastewater treatment processes and are usually adsorbed to sewage sludge and biosolids, with most reports revealing significantly elevated levels of PFAS in sewage sludges globally (Saliu & Sauvé, 2024).

Growing evidence suggests that organic contaminants have significant negative impacts on the sustainable development of the ecological environment (Vodyanitskii & Yakovlev, 2016). Taking earthworms as an example of highly visible macro-invertebrates that play vital roles in soil function and quality. Many organic contaminants are ingested by earthworms (Zhao et al., 2022) and accumulate in the digestive tract (Sidhu et al., 2019; Navarro et al., 2016) causing detectable damage (Gu et al., 2017; Šmídová et al., 2015). Earthworms form part of a wider food web being important prey for birds and fish thus the accumulation of organic contaminants in earthworms presents a wider threat beyond the soil environment (Kesic et al., 2021).

Like the 2018 assessment (Scottish Government, 2018) of sewage sludge risks to human health and the environment, the focus of the 2024 assessment is on *emerging* and *priority*

contaminants. The majority of *traditional* organic chemicals having reduced significantly in wastewater due to tighter source controls and governance through regulatory instruments and multilateral treaties such as the Stockholm Convention on Persistent Organic Pollutants (POPs) (Stockholm Convention - Home page (pops.int)). In the 2018 assessment (Scottish Government, 2018) reference was made to the EU Future Brief and other position statements on emerging organic contaminants as published in 2017 (Science for Environment Policy, 2017). At that time there was significant focus on perfluorochemicals (PFCs, including PFAS) and polybrominated diphenyl ethers (PBDEs). The concerns raised then were around the smaller evidence base for these chemicals and hence subsequent behaviour in the environment (Science for Environment Policy, 2017). Since then, new policy papers have been published that highlight the need for a *watching brief* and increased restrictions on emerging contaminants. On 7th February 2023, the European Chemicals Agency ECHA published a comprehensive dossier concerning a ban on ~10,000 PFAS, with an aim to restrict manufacture, placing on the market, and use of potentially harmful PFAS (ECHA, 2023). The ban is to be implemented under regulation (EU) No. 1907/2006 REACH, timetabled for 2027. In the UK, the British Chamber of Commerce EU & Belgium Chemicals Working Group has pushed back on an outright ban of PFAS stating the negative impact this would have on several industrial sectors and supply chains (BRITCHAM EU, 2024). Sewage sludge legislation and best practice are trying to keep up with these developments and pre-empt the final UK position on PFAS. For example, in England, the Environment Agency is in the process of moving away from the outdated Sludge (Use in Agriculture) Regulations as they no longer provide a fit for purpose regulatory framework for handling emerging contaminants including PFAS (Environment Agency, 2023). It is in this context, that the current assessment has been made.

The 2018 assessment (Scottish Government, 2018) identified several reviews that prioritised emerging contaminants for research (Clarke et al., 2016; Stewart et al., 2016; Thomaidi et al., 2016; McCarthy et al., 2015; Jensen et al., 2012; Clarke & Smith, 2011; Table 3.1). It was considered that while some of the contaminants listed might have changed position within these lists in terms of their relative priority, the basic lists still provided a sound basis for inclusion within the assessment. In addition, priority contaminants, based on known current chemicals of interest and those being considered for restrictions, were also added into the assessment. It should be noted that where a chemical is *sensu stricto* an organic chemical but is used primarily as a pharmaceutical or in personal care products, this chemical has not been in the risk assessment presented here but in the assessment for pharmaceuticals and personal care products (Section 4).

Table 3.1 – Identification of priority chemicals in the literature.

Clarke & Smith 2011[1]	Jensen et al., 2012[2]	Thomaidi et al., 2016[3]	Stewart et al., 2016[4]	Clarke et al., 2016[5]
10/11: PFCs 9/11: PCAs; PCNs 7/11: PBDEs; OTs; TCS; TCC 6/11: Benzothiazoles 5/11: Antibiotics, pharmaceuticals 3/11: Synthetic musks 2/11: BPA; QACs; Steroids 1/11: PAEs; PDMSs	Low risk[b] • BFRs •Musks •Pharmaceuticals •PCBs •PFCs[c] For future review[d] • PCAs • PCNs • TCC • TCS • Parabens	Synthetic phenolic compounds • NP • NP1EO • NP2EO • TCS Siloxanes • Decamethylcyclopenta- • Dodecamethylcyclohexa- • Dodecamethylpenta- • Tetradecamethylhexa- Benzothiazoles • 2-hydroxybenzothiazole Pharmaceuticals • Caffeine • Benzothiazoles	Flame retardants • BDE 77, 99 and 209 • TDCP, • TPP • TCPP Plasticisers • DEHP • BBP • Bisphenol A Perfluorinated compounds • PFOS/PFOA	•NP, NP1EO, NP2EO (1-3) •TCC (4) •TCS (5) •BPA (6) •Carbamazepine (7) •PBDE (8) •PCBs (9) •PFOA (10) •PFOS (11) •PCDD/Fs (14)

[1] Ranking of research priorities for emerging organic contaminants in biosolids (10 highest priority to 1 least priority) perfluorochemicals (PFCs); polychlorinated alkanes (PCAs); polychlorinated naphthalenes (PCNs); polybrominated diphenyl ethers (PBDEs); organotins (OTs); triclosan (TCS); triclocarban (TCC); bisphenol A (BPA); quaternary ammonium compounds (QACs); phthalate acid esters (PAEs) and polydimethylsiloxanes (PDMSs) [2]Prioritisation and selection of chemicals to assess the risk to soil organisms from the application of sludge (brominated flame retardants (BFR), musks, pharmaceuticals, PCBs and PFCs). Identification of chemicals for future evaluation. [3]Greece was used as case study and the environmental risk associated with the existence of 99 emerging OCCs in sludge-amended soil was estimated using risk quotient (RQ) approach. Chemical with a risk quotient >1 are listed. [4]Identification of indicator compounds for use in the assessment of organic chemical removal during wastewater treatment and their fate in receiving environments (Tris[2-chloro-1-(chloromethyl)ethyl]phosphate (TDCP); Triphenyl phosphate (TPP); Tris (1-chloro-2-propyl) phosphate (TCPP); Di (2-ethyhexyl)phthalate (DEHP); benzyl butyl phthalate (BBP); = perfluorooctanesulfonic acid (PFOS); perfluorooctanoic acid (PFOA)). 5A quantitative risk ranking model was developed for human exposure to emerging contaminants following biosolids application to Irish agricultural land. Chemicals are ranked by predicted environmental concentration in soil.

3.2. Risk Assessment

Chemicals listed in Table 3.1 were included in the risk assessment if; (i) their primary application was not for pharmaceutical or personal care products, (ii) if reliable distribution coefficients were available that satisfied the requirements of the multi-media fugacity models (Section 1.2.2). The full list of organic chemicals, their physico-chemical properties and predicted no effect concentration (PNEC) values are listed in Table 3.2.

Table 3.2 Physico-chemical data and predicted no effect concentrations (PNEC) for organic chemicals included in the risk assessment. Chemicals highlighted in bold are additional to the previous 2018 assessment (Scottish Government, 2018).

Chemical	Water solubility (mg l^{-1} @25ºC)	Vapour pressure (mm Hg @25ºC)	Henry's constant (atm-m^3 mol^{-1})	Log K_{ow}	$PNEC_{soil}$ (mg kg^{-1})
Benzo(a)anthracene	$9.40x10^{-3}$ (May et al., 1983)	$2.10x10^{-7}$ (Sonnefeld et al., 1983)	$1.20x10^{-5}$ (NCCT, 2024)	5.76 (Wang et al., 1986)	$7.90x10^{-2}$ (Verbruggen, 2012)
Benzo(a)pyrene	$1.62x10^{-3}$ (May et al., 1983)	$5.49x10^{-9}$ (Murray et al., 1974)	$4.57x10^{-7}$ (NCCT, 2024)	6.13 (de Maagd et al., 1998)	$5.30x10^{-2}$ (Verbruggen, 2012)
Benzo(b)fluoranthene	$1.50x10^{-3(a)}$ (Yalkowski et al., 2010)	$5.00x10^{-7(b)}$ (Coover & Sims 1987)	$6.57x10^{-7}$ (NCCT, 2024)	5.78 (Wang et al., 1986)	$2.80x10^{-1}$ (Verbruggen, 2012)
Benzo(k)fluoranthene	$7.60x10^{-4}$ (USEPA, 1987)	$9.65x10^{-10}$ (Murray et al., 1974)	$5.84x10^{-7}$ (NCCT, 2024)	6.11 (de Maagd et al., 1998)	$2.70x10^{-1}$ (Verbruggen, 2012)
Chrysene	$2.00x10^{-3}$ (Miller at al., 1985)	$6.23x10^{-9}$ (Hoyer & Peperle 1958)	$9.40x10^{-7}$ (NCCT, 2024)	5.73 (Hansch et al., 1995)	$5.50x10^{-1}$ (Verbruggen, 2012)
Indeno(1,2,3-cd)pyrene	$6.20x10^{-2(b)}$ (Sims & Overcash 1983)	$1.25x10^{-10}$ (USEPA, 2012)	$3.48x10^{-7}$ (NCCT, 2024)	6.70 (USEPA, 2012)	$1.30x10^{-1}$ (Verbruggen, 2012)
Naphthalene	$3.10x10^{1}$ (Pearlman et al., 1984)	$8.50x10^{-2}$ (Ambrose et al., 1975)	$4.40x10^{-4}$ (NCCT, 2024)	3.30 (Hansch et al., 1995)	$1.00x10^{0}$ (Verbruggen, 2012)
PCB 28	$1.17x10^{-1}$ (USEPA OPERA, 2024)	$3.46x10^{-4}$ (USEPA OPERA, 2024)	$2.05x10^{-4}$ (USEPA OPERA, 2024)	5.64 (USEPA OPERA, 2024)	$7.00x10^{-5}$ (Beduk et al., 2023)
PCB 52	$3.00x10^{-2}$ (USEPA OPERA, 2024)	$9.90x10^{-5}$ (USEPA OPERA, 2024)	$3.50x10^{-5}$ (USEPA OPERA, 2024)	6.10 (USEPA OPERA 2024)	$1.50x10^{-5}$ (Beduk et al., 2023)
PCB 95	$1.00x10^{-2}$ (USEPA OPERA, 2024)	$1.45x10^{-5}$ (USEPA OPERA, 2024)	$9.34x10^{-5}$ (USEPA OPERA, 2024)	6.56 (USEPA OPERA, 2024)	$1.50x10^{-5(*)}$ (Beduk et al., 2023)
PCB 101	$1.00x10^{-2}$ (USEPA OPERA, 2024)	$1.62x10^{-5}$ (USEPA OPERA, 2024)	$8.39x10^{-5}$ (USEPA OPERA, 2024)	6.43 (USEPA OPERA, 2024)	$5.00x10^{-6}$ (Beduk et al., 2023)
PCB 118	$1.00x10^{-3}$ (USEPA OPERA, 2024)	$1.36x10^{-5}$ (USEPA OPERA, 2024)	$7.86x10^{-5}$ (USEPA OPERA, 2024)	6.77 (USEPA OPERA, 2024)	$1.57x10^{-4}$ (Beduk et al., 2023)

PCB 132	$1.00x10^{-3}$ (USEPA OPERA, 2024)	$3.19x10^{-6}$ (USEPA OPERA, 2024)	$3.91x10^{-5}$ (USEPA OPERA, 2024)	7.24 (USEPA OPERA, 2024)	$1.10x10^{-5(*)}$ (Beduk et al., 2023)
PCB 138	$1.00x10^{-3}$ (USEPA OPERA, 2024)	$4.39x10^{-6}$ (USEPA OPERA, 2024)	$2.46x10^{-5}$ (USEPA OPERA, 2024)	7.33 (USEPA OPERA, 2024)	$1.10x10^{-5}$ (Beduk et al., 2023)
PCB 149	$1.00x10^{-3}$ (USEPA OPERA, 2024)	$5.53x10^{-6}$ (USEPA OPERA, 2024)	$3.34x10^{-5}$ (USEPA OPERA, 2024)	7.16 (USEPA OPERA, 2024)	$1.10x10^{-5(*)}$ (Beduk et al., 2023)
PCB 153	$1.00x10^{-3}$ (USEPA OPERA, 2024)	$4.01x10^{-6}$ (USEPA OPERA, 2024)	$3.08x10^{-5}$ (USEPA OPERA, 2024)	6.97 (USEPA OPERA, 2024)	$6.00x10^{-6}$ (Beduk et al., 2023)
PCB 174	$1.00x10^{-3}$ (USEPA OPERA, 2024)	$1.05x10^{-6}$ (USEPA OPERA, 2024)	$9.45x10^{-6}$ (USEPA OPERA, 2024)	7.71 (USEPA OPERA, 2024)	$1.00x10^{-5}$ (Beduk et al., 2023)
PCB 180	$1.00x10^{-3}$ (USEPA OPERA, 2024)	$1.02x10^{-6}$ (USEPA OPERA, 2024)	$7.63x10^{-6}$ (USEPA OPERA, 2024)	7.56 (USEPA OPERA, 2024)	$1.00x10^{-5}$ (Beduk et al., 2023)
2,3,7,8-TeCDD	$2.00x10^{-4}$ (Shiu et al., 1988)	$1.50x10^{-9}$ (Rordorf, 1987)	$8.95x10^{-6}$ (USEPA OPERA, 2024)	6.80 (Shiu et al., 1988)	$2.2x10^{-7}$ (Huygens et al., 2022)
1,2,3,7,8-PeCDD	$1.93x10^{-5}$ (USEPA OPERA, 2024)	$6.03x10^{-10}$ (USEPA OPERA, 2024)	$1.09x10^{-5}$ (USEPA OPERA, 2024)	6.58 (USEPA OPERA, 2024)	$2.2x10^{-7(*)}$ (Huygens et al. 2022)
1,2,3,4,6,7,8-HpCDD	$1.90x10^{-3}$ (Miyata et al., 1989)	$7.40x10^{-8}$ (Nestrick et al., 1980)	$2.18x10^{-5}$ (Hine & Mookerjee 1975)	8.11 (USEPA OPERA, 2024)	$2.20x10^{-5(*)}$ (Huygens et al., 2022)
2,3,4,7,8-PeCDF	$4.19x10^{-4}$ (USEPA OPERA, 2024)	$2.01x10^{-9}$ (USEPA OPERA, 2024)	$3.97x10^{-5}$ (USEPA OPERA, 2024)	6.62 (USEPA OPERA, 2024)	$4.40x10^{-7(*)}$ (Huygens et al., 2022)
1,2,3,4,7,8-HxCDF	$2.40x10^{-6}$ (USEPA OPERA, 2024)	$2.13x10^{-9}$ (USEPA OPERA, 2024)	$9.31x10^{-6}$ (USEPA OPERA, 2024)	7.45 (USEPA OPERA, 2024)	$2.20x10^{-6(*)}$ (Huygens et al., 2022)
1,2,3,6,7,8-HxCDF	$8.53x10^{-4}$ (USEPA OPERA, 2024)	$2.12x10^{-9}$ (USEPA OPERA, 2024)	$9.31x10^{-6}$ (USEPA OPERA, 2024)	7.45 (USEPA OPERA, 2024)	$2.20x10^{-6(*)}$ (Huygens et al., 2022)
2,3,4,6,7,8-HxCDF	$8.55x10^{-4}$ (USEPA OPERA, 2024)	$2.12x10^{-9}$ (USEPA OPERA, 2024)	$9.31x10^{-6}$ (USEPA OPERA, 2024)	7.45 (USEPA OPERA, 2024)	$2.20x10^{-6(*)}$ (Huygens et al., 2022)

Nonylphenol	$7.00x10^{0}$ (Yalkowsky & Dannenfelser 1992)	$8.18x10^{-4}$ (Bidleman & Renberg 1985)	$3.50x10^{-5}$ (NCCT, 2024)	5.76 (Itokawa et al., 1989)	$3.40x10^{-1}$ (Janssen et al., 2004)
Nonylphenol diethoxylate	$1.05x10^{0}$ (ECHA, 2013)	$9.14x10^{-9}$ (ECHA, 2013)	$2.56x10^{-9}$ (ECHA, 2013)	5.30 (ECHA, 2013)	$1.10x10^{-1}$ (Lamastra et al., 2018)
PBDE 99	$1.33x10^{-2}$ (EU, 2001)	$3.50x10^{-7}$ (Hardy & Smith 1999)	$2.50x10^{-5}$ (NCCT, 2024)	6.84 (Geyer et al. 2004)	$8.47x10^{-1}$ (Huygens et al., 2022)
PBDE 209	$1.00x10^{-4}$ (ECHA, 2015)	$6.96x10^{-11}$ (Lorber & Cleverly 2010)	$1.20x10^{-8}$ (NCCT, 2024)	9.97 (Environment Canada 2010)	$9.80x10^{1}$ (Huygens et al., 2022)
TBBPA	$1.71x10^{-1}$ (Kuramochi et al., 2008)	$4.68x10^{-8}$ (PubChem, 2023)	$2.00x10^{-8}$ (PubChem, 2023)	4.75 (Kuramochi et al.. 2008)	$3.10x10^{-2}$ (Öko-Institute, 2019)
HBCD	$3.40x10^{-3}$ (Stenzel & Markley 1997)	$4.70x10^{-7}$ (Stenzel & Nixon 1997)	$1.17x10^{-4}$ (Australian Government, 2012)	5.60 (MacGregor & Nixon 1997)	$4.30x10^{-1}$ (Australian Government, 2012)
PFOA	$3.30x10^{3}$ (Inoue et al., 2012)	$3.16x10^{-2}$ (Bhhararai & Gramatica 2011)	$2.02x10^{-10}$ (USEPA OPERA, 2024)	4.15 (USEPA, 2012)	$1.13x10^{-2}$ (Huygens et al., 2022)
PFOS	$3.20x10^{-3}$ (USEPA, 2012)	$2.00x10^{-3}$ (USEPA, 2012)	$1.85x10^{-11}$ (USEPA OPERA, 2024)	4.49 (USEPA, 2012)	$2.61x10^{-3}$ (Huygens et al., 2022)
PFNA	$6.25x10^{-2}$ (ECHA, 2014)	$8.30x10^{-2}$ (ECHA, 2014)	No Data	5.48 (ECHA, 2014)	$1.13x10^{-2}$ (Huygens et al., 2022)
PFHxS	$2.30x10^{3}$ (ECHA, 2017)	$4.60x10^{-3}$ (PubChem, 2023)	$5.03x10^{-6}$ (Plassmann et al., 2011)	Inappropriate (Stockholm Convention, 2018)	$1.13x10^{-2}$ (Huygens et al., 2022)
TFA	$1.50x10^{6}$ (ECHA, 2023)	$9.30x10^{1}$ (ECHA, 2023)	$8.88x10^{-8}$ (ECHA, 2023)	0.79 (ECHA, 2023)	$4.70x10^{-3}$ (ECHA 2023)
TCDP	$7.00x10^{0}$ (Yalkowsky et al., 2010)	$2.86x10^{-7}$ (USEPA, 2012)	$2.60x10^{-9}$ (NCCT, 2024)	3.65 (Chem Insp Test Inst., 1992)	$3.20x10^{-1}$ (ARCADIS, 2011)
TCPP	$1.08x10^{3}$ (EU, 2008a)	$1.05x10^{-5}$ (EU, 2008a)	$3.91x10^{-9}$ (EU, 2008a)	2.68 (EU, 2008a)	$1.70x10^{0}$ (EU, 2008a)
TCEP	$7.94x10^{3}$ (ECHA, 2023)	$8.55x10^{-6}$ (EU, 2008b)	$4.10x10^{-5}$ (EU, 2008b)	1.78 (EU, 2008b)	$3.41x10^{-1}$ (EU, 2008b)
TDCPP	$7.00x10^{0}$ (PubChem, 2023)	$1.10x10^{0}$ (ChemScr, 2023)	$2.60x10^{-9}$ (PubChem, 2023)	1.79 (ChemScr, 2023)	$2.90x10^{-1}$ (EU, 2008c)

DEHP	$2.70x10^{-1}$ (DeFoe et al., 1990)	$1.42x10^{-7}$ (Hinckley et al., 1990)	$2.70x10^{-7}$ (NCCT, 2024)	7.60 (de Bruijin et al., 1989)	$1.30x10^{1}$ (Huygens et al., 2022)
BBP	$2.69x10^{0}$ (Howard et al., 1985)	$8.25x10^{-6}$ (Howard et al., 1985)	$1.30x10^{-6}$ (NCCT, 2024)	4.73 (Ellington & Floyd 1996)	$1.39x10^{0}$ (EU 2008d)
Glyphosate	$1.2x10^{4}$ (ATSDR, 2020)	$9.80x10^{-8}$ (ATSDR, 2020)	$2.10x10^{-12}$ (ATSDR, 2020)	-3.40 (ATSDR, 2020)	$1.25x10^{0}$ (Xiao et al., 2023)

(*) based on TEFs

Initial concentrations in sewage sludge were set based on the '*realistic worst-case*' paradigm (Longhurst et al., 2019; Scottish Government 2018; Hough et al. 2012). The data used in the 2018 assessment (Scottish Government 2018) were used as an initial starting point, and these data were added to with recent data published since 2018. Concentrations used in the current assessment were derived from 90%ile values from the combined data sets (Table 3.3).

Table 3.3 – Initial concentrations of organic contaminants in sewage sludge prior to spreading. Chemicals highlighted in bold are additional to the previous 2018 assessment (Scottish Government 2018).

Chemical (IUPAC)	Concentration in sewage sludge (90%ile, mg kg^{-1})	References
Benzo(a)anthracene (benzo[a]anthracene)	$3.50x10^{-1}$	Kominko et al., 2024 Beduk et al., 2023 Wluka et al., 2021 Chen et al., 2019 Stanczyk-Mazanek et al., 2019 Sun et al., 2019 WRAP 2016
Benzo(a)pyrene (benzo[a]pyrene)	$3.80x10^{-1}$	
Benzo(b)fluoranthene (benzo[b]fluoranthene)	$4.70x10^{-1}$	
Benzo(k)fluoranthene (benzo[k]fluoranthene)	$2.60x10^{-1}$	
Chrysene (chrysene)	$5.70x10^{-1}$	
Indeno(1,2,3-cd)pyrene (indeno[1,2,3-cd]pyrene)	$4.90x10^{-1}$	
Naphthalene (naphthalene)	$3.90x10^{-1}$	
PCB 28 (2,4,4'-trichlorobiphenyl)	$1.00x10^{-1}$	Kominko et al., 2024 Beduk et al., 2023 Kosnar et al., 2023 WRAP 2016
PCB 52 (2,2',5,5'-tetrachlorobiphenyl)	$3.00x10^{-2}$	
PCB 95 (2,2',3,5',6-pentachlorobiphenyl)	$3.00x10^{-2}$	
PCB 101 (2,2',4,5,5'-pentachlorobiphenyl)	$4.00x10^{-2}$	
PCB 118 (2,3',4,4',5-pentachlorobiphenyl)	$3.20x10^{-1}$	
PCB 132 (2,2',3,3',4,6'-hexachlorobiphenyl)	$7.00x10^{-2}$	
PCB 138 (2,2',3,4,4',5'-hexachlorobiphenyl)	$8.00x10^{-2}$	
PCB 149 (2,2',3,4',5',6-hexachlorobiphenyl)	$8.00x10^{-2}$	
PCB 153 (2,2',4,4',5,5'-hexachlorobiphenyl)	$8.00x10^{-2}$	
PCB 174 (2,2',3,3',4,5,6'-heptachlorobiphenyl)	$2.00x10^{-2}$	
PCB 180 (2,2',3,4,4',5,5'-heptachlorobiphenyl)	$4.00x10^{-2}$	
2,3,7,8-TeCDD (2,3,7,8-tetrachlorodibenzo-p-dioxin)	$1.20x10^{-6}$	Kominko et al., 2024 WRAP 2016
1,2,3,7,8-PeCDD (1,2,3,7,8-pentachlorodibenzo-p-dioxin)	$8.80x10^{-6}$	
1,2,3,4,6,7,8-HpCDD (1,2,3,4,6,7,8-heptachlorodibenzo-p-dioxin)	$1.20x10^{-4}$	
2,3,4,7,8-PeCDF	$1.1x10^{-5}$	

(4,5,6,11,12-pentachloro-8-oxatricyclo[7.4.0.0^{2,7}]trideca-1(13),2,4,6,9,11-hexaene)		
1,2,3,4,7,8-HxCDF (3,4,5,6,11,12-hexachloro-8-oxatricyclo[7,4,0,0^{2,7}]trideca-1(13),2,4,6,9,11-hexaene)	$1.50x10^{-5}$	
1,2,3,6,7,8-HxCDF (3,4,5,10,11,12-hexachloro-8-oxatricyclo[7.4.0.0^{2,7}]trideca-1(13),2,4,6,9,11-hexaene)	$1.60x10^{-5}$	
2,3,4,6,7,8-HxCDF (4,5,6,10,11,12-hexachloro-8-oxatricyclo[7.4.0.0^{2,7}]trideca-1(13),2,4,6,9,11-hexaene)	$1.40x10^{-5}$	
Nonylphenol (4-nonylphenol)	$2.38x10^{2}$	Madrid et al., 2020 Rivier et al., 2019 Ghanem et al., 2007 Gibson et al., 2005
NP2EO (2-[2-(4-nonylpohenoxy)ethoxy]ethanol)	$1.35x10^{2}$	Rivier et al., 2019 González et al., 2010
PBDE 99 (2,2',4,4',5-pentabromodiphenyl ether)	$2.45x10^{0}$	Leslie et al., 2021 Wang et al., 2020 Demitepe & Imamoglu 2019 Harrison et al., 2006
PBDE 209 (decabromodiphenyl ether)	$2.45x10^{0}$	Leslie et al., 2021 Wang et al., 2020 Demitepe & Imamoglu 2019 Harrison et al., 2006
TBBPA (2,6-dibromo-4-[2-(3,5-dibromo-4-hydroxyphenyl)propan-2-yl]phenol)	$2.40x10^{-1}$	Öko-Onstitute, 2019 Environment Canada, 2016 USEPA, 2014 Morris et al., 2004
HBCD (1,2,5,6,9,10-hexabromocyclododecane)	$9.10x10^{0}$	Morris et al., 2004
PFOA (pentadecafluorooctanoic acid)	$1.46x10^{-2}$	Holly et al., 2024 Saliu & Sauve 2024 Zhou et al., 2024 Johnson 2022 Zareitalabad et al., 2013
PFOS (1,1,2,2,3,3,4,4,5,5,6,6,7,7,8,8,8-heptadecafluorooctane-1-sulfonic acid)	$4.29x10^{-3}$	Holly et al., 2024 Saliu & Sauve 2024 Zhou et al., 2024 Johnson, 2022 Zareitalabad et al., 2013
PFNA	$3.34x10^{-1}$	Holly et a.,l 2024

(pentadecafluorononanionic acid)		Saliu & Sauve 2024 Zhou et al., 2024 Danish EPA, 2023 Johnson, 2022
PFHxS (1,1,2,2,3,3,4,4,5,5,6,6-tridecafluorohexane-1-sulfonic acid)	$1.33x10^{0}$	Holly et al., 2024 Saliu & Sauve 2024 Zhou et al., 2024 Danish EPA, 2023 Johnson, 2022
TFA (2,2,2-trifluoroethanoic acid)	ND	
TDCP (tris(1,3-dichloropropan-2-yl) phosphate)	$2.60x10^{-1}$	Österås et al., 2015 Marklund et al., 2005
TCPP (tris(2-chloro-1-methlyethyl) phosphate)	$6.44x10^{0}$	Rede et al., 2024 Bester, 2005
TCEP (tris(2-chloroethyl) phosphate)	$6.44x10^{0}$	Assumed same as TCPP from Bester, 2005
TDCPP (tris(1,3-dichloropropan-2-yl) phosphate)	$6.44x10^{0}$	Assumed same as TCPP from Bester, 2005
DEHP (bis(2-ethylhexyl) benzene-1,2-dicarboxylate)	$2.70x10^{0}$	Österås et al., 2024 Lamastra et al., 2018 EU 2008e Bright & Healy 2003
BBP (benzyl butyl benzene-1,2-dicarboxylate)	$3.80x10^{-1}$	Bright & Healy 2003
Glyphosate (N-(phosphonomethyl)glycine)	$5.71x10^{-1}$	Rede et al., 2023 Wydro et al., 2021 Ghanem et al., 2007

Combining partition coefficients (Table 3.2) with initial concentrations in sewage sludge (Table 3.3) within equations 1.1 – 1.7 (section 1.2.2), it was possible to estimate the concentrations of each organic contaminant that would remain in the sewage-amended soil compared to migration to soil pore water or soil pore air (Figure 3.1). Those contaminants with the greatest propensity to partition to the soil pore water are subsequently more likely to be taken up by plants (such as food crops) or leached to surface or ground waters, thus those contaminants with relatively high partitioning to water are more likely to pose risks in the aquatic environment and food chain. It should be noted that a full set of partition coefficients were not available for PFNA, PFHxS and the degradation product TFA so these contaminants were excluded from the partitioning analysis. In the case of PFHxS, the derivation of octanol:water partitioning is considered inappropriate (Stockholm Convention 2018) so cannot be included in a partitioning model of this type. However, it was possible to estimate concentrations of PFNA and PFHxS in sewage-amended soil. Thus, risk estimates are presented for PFNA and PFHxS in Section 3.2.1.

Figure 3.1 – Partitioning (ppm, %) of organic contaminants between soil amended with sewage sludge, soil pore water, and soil sir spaces given initial starting concentrations in sewage sludge (Table 3.3). This is based on a scenario that supports precautionary decision making using a single application of sewage sludge at 50 t ha^{-1}.

3.2.1. Risk Characterisation

The current assessment focuses on ecological impacts of the contaminants of interest on soil biota. To this end, the predicted concentrations of each chemical in the sewage-amended soil fraction were assumed to be equivalent to predicted effect concentrations (PEC) to which soil biota are exposed. Risk ratios or relative risk (RR), terms used inter-changeably in this report, were then derived as the comparison of the PEC to published predicted no-effect levels (PNEC) for soil (Table 3.2, see Equation 1.10 from methods). The un-logged values of RR are also listed in Table 3.4.

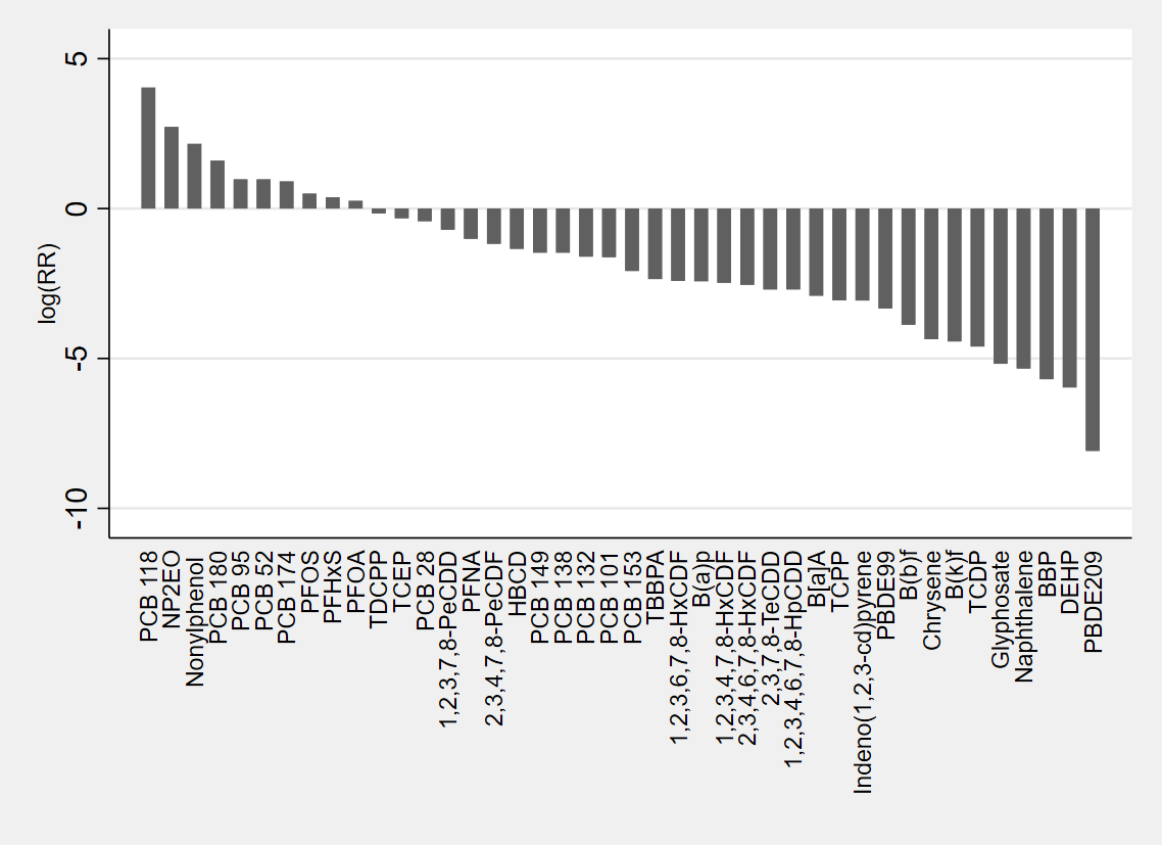

Figure 3.2 – Estimates of relative risk or risk ratio, presented using a logarithmic scale log(RR), for each organic contaminant based on the ratio of predicted effect concentration (PEC; see Figure 3.1) to the predicted no-effect concentration (PNEC; see Table x.2). These estimates are based on a single application of sewage sludge at 50 t $^{ha\text{-}1}$. A log(RR) > 0 indicates that further investigation may be required.

Table 3.4 – Estimates of relative risk (or risk ratio) (RR) for each organic contaminant based on the ratio of predicted effect concentration (PEC; see Figure 3.1) to the predicted no-effect concentration (PNEC; see Table x.2). These estimates are based on a single application of sewage sludge at 50 t^{ha-1}. An RR > 1 (shaded in blue) indicates that further investigation may be required.

Contaminant	Risk Ratio, RR
PCB 118	5.64×10^{1}
NP2EO	1.52×10^{1}
Nonylphenol	8.65×10^{0}
PCB 180	4.94×10^{0}
PCB 52	2.65×10^{0}
PCB 95	2.65×10^{0}
PCB 174	2.47×10^{0}
PFOS	1.64×10^{0}
PFHxS	1.46×10^{0}
PFOA	1.29×10^{0}
TDCPP	8.51×10^{-1}
TCEP	7.24×10^{-1}
PCB 28	6.57×10^{-1}
1,2,3,7,8-PeCDD	4.94×10^{-1}
PFNA	3.65×10^{-1}
2,3,4,7,8-PeCDF	3.09×10^{-1}
HBCD	2.61×10^{-1}
PCB 138	2.30×10^{-1}
PCB 149	2.30×10^{-1}
PCB 132	2.01×10^{-1}
PCB 101	1.98×10^{-1}
PCB 153	1.25×10^{-1}
TBBPA	9.52×10^{-2}
1,2,3,6,7,8-HxCDF	8.98×10^{-2}
B(a)p	8.85×10^{-2}
1,2,3,4,7,8-HxCDF	8.42×10^{-2}
2,3,4,6,7,8-HxCDF	7.86×10^{-2}
2,3,7,8-TeCDD	6.73×10^{-2}
1,2,3,4,6,7,8-HpCDD	6.73×10^{-2}
B[a]A	5.47×10^{-2}
TCPP	4.68×10^{-2}
Indeno(1,2,3-cd)pyrene	4.65×10^{-2}
PBDE99	3.57×10^{-2}
B(b)f	2.07×10^{-2}
Chrysene	1.28×10^{-2}
B(k)f	1.19×10^{-2}
TCDP	1.00×10^{-2}
Glyphosate	5.64×10^{-3}
Naphthalene	4.81×10^{-3}
BBP	3.38×10^{-3}
DEHP	2.56×10^{-3}
PBDE209	3.09×10^{-4}

3.3. Conclusions

Using a scenario that supports precautionary decision making with a single application of sewage sludge (50 t ha^{-1}), ten contaminants all returned a relative risk (or risk ratio) greater than unity, with a number of these indicating elevated risks to soil health (e.g. PCB 118).

These included several polychlorinated biphenyls (PCBs 118, 180, 95, 52, 174) which were not previously highlighted by the 2018 assessment (Scottish Government 2018) which had a focus on human health as the end point of the risk analysis. In updating the PCB database, it should be noted that measured concentrations of PCBs in sewage sludge have decreased since the 2018 assessment. This decrease has been significant enough to lower the 90th percentile concentrations entered into the assessment (Table 3.2). The improvement in sewage sludge quality is likely a result of PCBs having now been a focus of environmental concern since the 1960s, with production banned internationally since 2001. It is also worth noting that the receiving environment for the sewage sludge, i.e., the soil, has also decreased in PCB content (Dendievel et al., 2020). As the focus of this assessment is on sewage sludge as a product, the improvements in the receiving environment are not considered explicitly but are indicative of the success of legislative intervention. It was decided in this analysis to combine post-2018 data with the data already collated for the 2018 assessment. Given the continued decrease in PCB concentrations in sewage sludge, it is worth noting that lower risk estimates would have been achieved if only post-2018 data had been entered into the analysis simply because the input data have lower concentrations. Given the worldwide ban on the manufacture of PCBs, it is expected that levels of PCBs in sewage sludge would continue to improve. However, the overall quality of sewage sludge is still compromised by the presence of multiple types of harmful contaminants many of which cannot be assessed due to lack of data (see Section 5.3).

The detergents nonylphenol and nonylphenol diethoxylate (NP2EO) also returned values of RR greater than unity, similar to the human health risk estimates reported in the 2018 assessment. Given the prominence of these contaminants in both assessments the fate of nonylphenols in sewage sludge and sludge amended soils warrants concern and should prompt further investigation.

The per- and polyfluoroalkyl substances (PFAS) also returned appreciable risk estimates for those that could be assessed (PFOS, PFHxS and PFOA). It is worth noting that these are only three PFAS out of a group of over 10,000 PFAS chemicals. This contrasts with the 2018 assessment (Scottish Government 2018) where PFOS and PFOA both returned risk estimates significantly lower than unity (PFHxS was not assessed in 2018). This result is partly because data for PFAS in sewage sludge have become more plentiful and measurement techniques have become more refined over the past five years. The post-2018 data have increased 90th percentile concentrations for both PFOS and PFOA, but only to a small extent and these increases do not account for all the increase in risk seen when comparing the two assessments. The current assessment looks at the direct exposure of soil organisms to PFAS added to the soil via sewage sludge. As such, this is a far more direct exposure than the 2018 assessment that looked at impacts on human health. Risk estimates in the 2018 assessment (Scottish Government 2018) therefore reflect the potential for PFAS to enter the terrestrial food chain via uptake of these chemicals from the soil pore waters by crops. Overall, given the potential impact on soil biota (and therefore soil health), the fate of PFAS in sewage sludge and sludge-amended soils warrants further investigation.

The relative risk estimates presented here are based on a *realistic worst-case* scenario for informing decision making using a 'heavy' application of sewage sludge of 50 t ha^{-1}. Typical application rates of sewage sludge vary from 5 to 20 t ha^{-1} depending on soil type, crop type, and crop nutrient requirements (BAS, 2019). As such, this exposure scenario will likely only apply to a small proportion of the agricultural land bank. Assuming sludge providers and farmers are following guidance and testing sludges and receiving soils, the vast majority of agricultural soils are likely to experience lower levels of sewage sludge applications, if at all depending on the availability of alternative organic fertilisers. The use of *realistic worst-case* assumptions for informing decision making aimed at protecting the most vulnerable populations of soil biota, hence this approach will highlight chemicals posing risk under unusual but legitimate scenarios. This is helpful as it gives a level of confidence where we identify compounds that despite the worst-case scenario pose little risk to soil biota. The cocktail effect caused by multiple exposures to multiple agents is also worth consideration. Little is known about how the compounds assessed here combine, degrade, and change over time. The soil biota are experiencing exposure to this multitude of chemicals and knowledge of how this affects toxicity and impacts is extremely limited. It is most likely that the cocktail effect increases magnitude of risk, rather than reduces it. This further strengthens the argument for using scenarios that support precautionary decision making as these emphasise risks where they might occur.

It should also be noted that the study has only considered a single application of sewage sludge for the assessment of organic contaminants. The 42 contaminants assessed here all behave differently in the environment, some will persist and accumulate in soils, some will degrade, some will move away from the exposure site. For some of the contaminants, these behaviours are known enough to simulate; but for many they are not. For this reason, a single application has been assessed. There are currently no long-term sewage sludge experimental farm plots in the UK, and previous ones have focussed on *traditional* contaminants such as the heavy metals regulated under The Sludge (Use in Agriculture) regulations (Cd, Pb, Zn). Long-term monitoring data of organic contaminants on sludge amended soils is lacking but using the precautionary approach, based on evidence gathered from previous long-term experiments conducted in the UK, Sweden, Germany and the USA, where metals were found to have adverse effects on soil microbial parameters (e.g., McGrath et al., 1995) would support soil protection legislation and inform the safe and sustainable management of sewage sludge. Findings from the ongoing 2-year UK Water Industry Research Chemical Investigations Programme Phase 4 (CIP4 - Introducing the 4th Phase of the Chemical Investigations Programme - Jenni Hughes, UKWIR Strategic Programme Manager) investigating the fate and behaviour of metals, nutrients, and a selection of persistent organic pollutants in biosolids including perfluorinated compounds, pesticides, phthalates, plasticisers, tyre compounds, pharmaceuticals (including antibiotics) as well as microplastics will be valuable.

3.4. References

Ambrose, D., Ellender, J.H., Sprake, C.H.S. & Townsend, R. 1975. Thermodynamic properties of fluorine compounds. Part 15. – Vapour pressures of the three tetrafluorobenzenes and 1,3,5-trichloro-2,4,6-trifluorobenzene. *Journal of the Chemistry Society, Faraday Transactions 1: Physical Chemistry of Condensed Phases* **71:**35-41. https://doi.org/10.1039/F19757100035

ARCADIS. 2011. Identification and evaluation of data on flame retardants in consumer products. Report commissioned by EU DG Health and Consumers, Brussels, 17.020200/09/549040. https://www.europeanfiresafetyalliance.org/wp-content/uploads/2017/04/27.pdf

ATSDR. 2020. Toxicological profile for glyphosate. U.S. Department of Health and Human Services, Agency for Toxic Substances and Disease Registry, August 2020. https://www.atsdr.cdc.gov/toxprofiles/tp214.pdf

Australian Government. 2012. Hexabromocyclododecane. Priority Existing Chemical Assessment Report No. 34, Australian Government, Department of Health and Ageing NICNAS, June 2012. https://www.industrialchemicals.gov.au/sites/default/files/PEC34-Hexabromocyclododecane-HBCD.pdf

BAS (Biosolids Assurance Scheme). 2019. UK Assured Biosolids, Biosolids Agricultural Good Practice Guidance Leaflet. BAS 2019. https://assuredbiosolids.co.uk/wp-content/uploads/2019/01/Biosolids-Agric-Good-Practice-Guidance-January-2019.pdf

Beduk, F., Aydin, S., Ulvi, A. & Aydin, M.E. 2023. Persistent organic pollutants in sewage sludge: occurrence, temporal concentration variation and risk assessment for sewage sludge amended soils. *KSCE Journal of Civil Engineering* **27:**3694-3704. https://doi.org/10.1007/s12205-023-2385-x

Bester, K. 2005. Comparison of TCPP concentrations in sludge and wastewater in a typical German sewage treatment plant – comparison of sewage sludge from 20 plants. *Journal of Environmental Monitoring* **7:**509-513. https://doi.org/10.1039/B502318A

Bhhatarai, B. & Gramatica, P. 2011. Prediction of aqueous solubility, vapor pressure and critical micelle concentration for aquatic partitioning of perfluorinated chemicals. *Environmental Science & Technology* **45:**8120-8128. https://doi.org/10.1021/es101181g

Bidleman, T.F. & Renberg, L. 1985. Determination of vapor pressures for chloroguaiacols, chloroveratroles, and nonylphenol by gas chromatography. *Chemosphere* **14:**1475-1481. https://doi.org/10.1016/0045-6535(85)90005-0

Brandli, R.C., Kupper, T., Bucheli, T.D., Zennegg, M., Huber, S., Ortelli, D., Muller, J., Schaffner, C., Lozza, S., Schmid, P., Berger, U., Edder, P., Oehme, M., Stadelmann, F.X. & Tarradellas, J. 2007. Organic pollutants in compost and digestate. Part 2. Polychlorinated dibenzo-*p*-dioxins, and -furans, dioxin-like polychlorinated biphenyls, brominated flame retardants, perfluorinated alkyl substances, pesticides, and other compounds. *Journal of Environmental Monitoring* **9:**465-472. https://doi.org/10.1039/B617103F

Bright, D.A. & Healey, N. 2003. Contaminant risks from biosolids land application: Contemporary organic contaminant levels in digested sewage sludge from five treatment plants in Greater Vancouver, British Columbia. *Environmental Pollution* **126:**39-49. https://doi.org/10.1016/S0269-7491(03)00148-9

BRITCHAM EU. 2024. Position paper on the EU Universal PFAS restriction proposal. British Chamber of Commerce EU & Belgium, 1st July 2024. https://www.britcham.eu/blog/our-positions-1/position-paper-on-the-eu-universal-pfas-restriction-proposal-27

Chemicals Inspection and Testing Institute. 1992. Biodegradation and bioaccumulation data of existing chemicals based on the CSCL Japan. Japan Chemical Industry Ecology – Toxicology and Information Center. ISBN 4-89074-101-1.

Chen, C-F., Ju, Y-R., Lim, Y.C., Hsieh, S-L., Tsai, M-L., Sun, P-P., Katiyar, R., Chen, C-W. & Dong, C-D. 2019. Determination of polycyclic aromatic hydrocarbons in sludge from water and wastewater treatment plants by GC-MS. *International Journal of Environmental Research and Public Health* **16:**2604. https://doi.org/10.3390/ijerph16142604

Clarke, B.O. & Smith, S.R. 2011. Review of 'emerging' organic contaminants in biosolids and assessment of international research priorities for the agricultural use of biosolids. *Environment International* **37:**226-247. https://doi.org/10.1016/j.envint.2010.06.004

Clarke, R., Healy, M.G., Fenton, O. & Cummins, E. 2016. A quantitative risk ranking model to evaluate emerging organic contaminants in biosolids amended land and potential transport to drinking water. *Human and Ecological Risk Assessment* **22:**958-990. https://doi.org/10.1080/10807039.2015.1121376

Coover, M.P. & Sims, R.C. 1987. The effect of temperature on polycyclic aromatic hydrocarbon persistence in an unacclimated agricultural soil. *Hazardous Waste and Hazardous Materials* **4:**69-82. https://doi.org/10.1089/hwm.1987.4.69

Danish EPA. 2023. Derivation of cut-off values for PFAS in sewage sludge, revised edition. Ministry of Environment of Denmark, Environment Protection Agency, Environmental Project no. 2232, March 2023. https://www2.mst.dk/Udgiv/publications/2023/03/978-87-7038-497-1.pdf

De Bruijn, J., Busser, F., Seinen, W. & Hermens, J. 1989. Determination of octanol/water partition coefficients for hydrophobic organic chemicals with the "slow-stirring" method. *Environmental Toxicology and Chemistry* **8:**499-512. https://doi.org/10.1002/etc.5620080607

DeFoe, D.L., Holcombe, G.W., Hammermeister, D.E. & Biesinger, K.E. 1990. Solubility and toxicity of eight phthalate esters to four aquatic organisms. *Environmental Toxicology and Chemistry* **9:**623-636. https://doi.org/10.1002/etc.5620090509

de Maagd, P.G-J., ten Hulscher, D.Th.E.M., van den Heuvel, H., Opperhuizen, A. & Sijm, D.T.H.M. 1998. Physicochemical properties of polycyclic aromatic hydrocarbons: Aqueous solubilities, *n*-octanol/water partition coefficients, and Henry's law constants. *Environmental Toxicology and Chemistry* **17:**251-257. https://doi.org/10.1002/etc.5620170216

Demirtepe, H. & Imamoglu, I. 2019. Levels of polybrominated diphenyl ethers and hexabromocyclododecane in treatment plant sludge: Implications on sludge management. *Chemosphere* **221:**606-615. https://doi.org/10.1016/j.chemosphere.2019.01.060

Dendieval, A-M., Mourier, B., Coynel, A., Everard, O., Labadie, P., Ayrault, S., Debret, M., Koltalo, F., Copard, Y., Faivre, Q., Gardes, T., Vauclin, S., Budszinski, H., Grosbois, C., Winiarski, T. & Desmet, M. 2020. Spatio-temporal assessment of the polychlorinated biphenyl (PCB) sediment contamination in four major French river corridors (1945 – 2018). *Earth Systems Science Data* **12:**1153-1170. https://doi.org/10.5194/essd-12-1153-2020

ECHA. 2023. Restriction on the manufacture, placing on the market and use of PFASs. Annex XV Restriction Report. Proposal for a restriction per- and polyfluoroalkyl substances (PFAS). 22nd March 2023. https://echa.europa.eu/documents/10162/f605d4b5-7c17-7414-8823-b49b9fd43aea

ECHA. 2017. Agreement of the Member State Committee on the identification of perfluorohexane-1-sulphonic acid and its salts as substances of very high concern. According to Articles 57 and 59 of Regulation (EC) 1907/2006, Adopted 15th June 2017. https://echa.europa.eu/documents/10162/fdc986a0-7479-245a-b64a-7724d1ee760c

ECHA. 2015. Search for chemicals. Bis(pentabromophenyl)ether (CAS 1163-19-5) Registered Substances Dossier. European Chemicals Agency 2015. https://echa.europa.eu/substance-information/-/substanceinfo/100.013.277

ECHA. 2014. Committee for Risk Assessment RAC: Opinion proposing harmonised classification and labelling at EU level of perfluorononan-1-oic acid [1]; (2,2,3,3,4,4,5,5,6,6,7,7,8,8,9,9,9,-heptadecafluorononanoic acid (PFNA) and its sodium (PFN-S) [2] and ammonium (PFN-A) [3] salts. CAS 375-95-1. European Chemicals Agency 2014. CLH-O-0000004708-66-03/F. https://echa.europa.eu/documents/10162/e7038b5d-9d39-d39d-1a54-6c9eb0a23baf

ECHA. 2013. Member state committee support document for the identification of 4-nonylphenol, branched and linear, ethoxylated. European Chemicals Agency 2013. https://echa.europa.eu/documents/10162/9af34d5f-cd2f-4e63-859c-529bb39da7ae

Ellington, J.J. & Floyd, T.L. 1996. Octanol/water partition coefficients for eight phthalate esters. EPA/600/S-96/006, September 1996; Athens, GA: USEPA, National Exposure Research Laboratory 1996. https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=30003VNC.PDF

Environment Agency. 2023. Environment Agency strategy for safe and sustainable sludge use. Policy paper, 1st August 2023. https://www.gov.uk/government/publications/environment-agency-strategy-for-safe-and-sustainable-sludge-use/environment-agency-strategy-for-safe-and-sustainable-sludge-use#reasons-for-change

Environment Canada. 2010. Ecological state of the science report on decabromodiphenyl ether (decaBDE), bioaccumulation and transformation (August 2010). https://www.canada.ca/content/dam/eccc/migration/main/lcpe-cepa/documents/substances/decabde/ess_report_decabde-eng.pdf

Environment Canada. 2016. Canadian Environmental Protection Act, 1999. Federal Environmental Quality Guidelines. Tetrabromobisphenol A (TBBPA). Environment and Climate Change Canada, May 2016. https://www.canada.ca/content/dam/eccc/migration/ese-ees/2fe86525-c439-4bc8-b20a-315af1cb6f6a/feqg_tbbpa_en.pdf

EU. 2001. Diphenyl ether, pentabromo derivative (pentabromodiphenyl ether). European Union Risk Assessment Report. Luxembourg, Office for Official Publications of the European Committees https://echa.europa.eu/documents/10162/05a9c558-9472-4163-9496-1a8ab1ee6918

EU. 2008a. Tris(2-chloro-1-methylethyl)phosphate (TCPP). Summary Risk Assessment Report. European Commission EU Risk Assessment TCPP CAS 13674-https://echa.europa.eu/documents/10162/b8497638-00ad-442e-9a8f-02e212570b54

EU. 2008b. Tris(2-chloroethyl)phosphate, TCEP. Summary Risk Assessment Report. European Commission 2008 https://echa.europa.eu/documents/10162/2663989d-1795-44a1-8f50-153a81133258

EU. 2008c. Tris[2-chloro-1-(chloromethyl)ethyl]phosphate (TDCP). European Union Risk Assessment Report. European Commission, EU Risk Assessment – TDCP CAS 13674-87-8 https://echa.europa.eu/documents/10162/13630/trd_rar_ireland_tdcp_en.pdf/d5c05135-ed67-4c65-a090-a45ec72d9469

EU. 2008d. Benzyl butyl phthalate (BBP). Summary Risk Assessment Report. Institute for Health and Consumer Protection, Toxicology and Chemical Substances (TCS), European Chemicals Bureau I-21027 Ispra, Italy. https://echa.europa.eu/documents/10162/b8497638-00ad-442e-9a8f-02e212570b54

Geyer, H.J., Schramm, K.W., Darnedud, P.O., Aune, M., Feicht, A., Fried, K.W., Henkelmann, B., Lenoir, D., Schmid, P. & McDonald, T.A. 2004. Terminal elimination half-lives of the brominated flame retardants TBBPA, HBCDD and lower brominated PBDEs in humans. *Organohalogen Compounds* **66:**3820-3825 https://www.osti.gov/etdeweb/servlets/purl/20828431

Ghanem, A., Bados, P., Estuan, A.R., de Alencastro, L.F., Taibi, S., Einhorn, J. & Mougin, C. 2007. Concentrations and specific loads of glyphosate, diuron, atrazine, nonylphenol and metabolites thereof in French urban sewage sludge. *Chemosphere* **69:**1368-1373 https://doi.org/10.1016/j.chemosphere.2007.05.022

Gibson, R., Wang, M-J., Padgett, E. & Beck, A.J. 2005. Analysis of 4-nonylphenols, phthalates, and polychlorinated biphenyls in soils and biosolids. *Chemosphere* **61:**1336-1344 https://doi.org/10.1016/j.chemosphere.2005.03.072

González, M., Martin, J., Santox, J., Aparicio, I. & Alonso, E. 2010. Occurrence and risk assessment of nonylphenol and nonylphenol ethoxylates in sewage sludge from different conventional treatment processes. *Science of the Total Environment* **408:**563-570 https://doi.org/10.1016/j.scitotenv.2009.10.027

Gu, J., Jing, Y., Ma, Y., Sun, F., Wang, L., Chen, J., Guo, H. & Ji, R. 2017. Effects of the earthworm *Metaphire guillelmi* on the mineralization, metabolism, and bound-residue formation of tetrabromobisphenol A (TBBPA) in soil. *Science of the Total Environment* **595:**528-536. https://doi.org/10.1016/j.scitotenv.2017.03.273

Hansch, C., Leo, A. & Hoekman, D. 1995. Exploring QSAR – Hydrophobic, electronic, and steric constants. Washington DC, American Chemical Society, 1995, ISBN 978-0841229914.

Hardy, M.L. & Smith, R.L. 1999. In American Chemical Society, Division of Environmental Chemistry, Preprint Extended Abstracts, 217th ACS National Meeting, V39: 191-4 https://www.tib.eu/en/search/id/TIBKAT:300343264?cHash=872929316ea20aca9f7f77ed3eec677b

Harrison, E.Z., Oakes, S.R., Hysell, M. & Hay, A. 2006. Organic chemicals in sewage sludges. *Science of the Total Environment* **367:**481-497 https://doi.org/10.1016/j.scitotenv.2006.04.002

Hinckley, D.A., Bidleman, T.F. & Foreman, W.T. 1990. Determination of vapor pressures for nonpolar and semipolar organic compounds from gas chromatographic retention data. *Journal of Chemical & Engineering Data* **35:**232-237 https://doi.org/10.1021/je00061a003

Hine, J. & Mookerjee, P.K. 1975. Structural effects on rates and equilibriums. XIX. The intrinsic hydrophilic character of organic compounds. Correlations in terms of structural contributions. *Journal of Organic Chemistry* **40:**292-298 https://doi.org/10.1021/jo00891a006

Holly, M.A., Gunn, K.M., Keymer, D. & Sanford, J.R. 2024. Evaluation of per- and polyfluoroalkyl substances leaching from biosolids and mitigation potential of biochar through undisturbed soil columns. *ACS ES&T Water* **4:**413-426 https://doi.org/10.1021/acsestwater.3c00414

Howard, P.H., Banerjee, S. & Robillard, K.H. 1985. Measurement of water solubilities, octanol/water partition coefficients and vapor pressures of commercial phthalate esters. *Environmental Toxicology and Chemistry* **4:**653-661 https://doi.org/10.1002/etc.5620040509

Hough, R.L., Booth, P., Avery, L.M., Rhind, S., Crews, C., Bacon, J., Campbell, C.D. & Tompkins, D. 2012. Risk assessment of the use of PAS100 green composts in sheep and cattle production in Scotland. *Waste Management* **32:**117-130 https://doi.org/10.1016/j.wasman.2011.08.008

Hoyer, H. & Peperle, W. 1958. Dampfdruckmessungen an organischen substanzen und ihre sublimationswärmen. *Berichte der Bunsengesellschaft für Physikalische Chemie* **62:**61-66 [in German] https://doi.org/10.1002/bbpc.19580620109

Huygens, D., Garcia-Guiterrez, P., Orvellion, G., Schillaci, C., Delre, A., Orgiazzi, A., Wojda, P., Tonini, D., Egle, L., Jones, A., Pistocchi, A. & Lugato, E. 2022. Screening risk assessment of organic pollutants and environmental impacts from sewage sludge management, EUR 31238 EN, Publications Office of the European Union, Luxembourg, 2022, JRC129690 https://dx.doi.org/10.2760/541579

Inoue, Y., Hashizume, N., Yakata, N., Murikami, H., Suzuki, Y., Kikushima, E. & Otsuka, M. 2012. Unique physicochemical properties of perfluorinated compounds and their bioconcentration in common carp *Cyprinus carpio* L. *Archives of Environmental Contamination and Toxicology* **62:**672-680. https://doi.org/10.1007/s00244-011-9730-7

Itokawa, H., Totsuka, N., Nakahara, K., Maezuru, M., Takeya, K., Kondo, M., Inamatsu, M. & Morita, H. 1989. A quantitative structure-activity relationship for antitumor activity of long-chain phenols from *Ginkgo biloba* L. *Chemical and Pharmaceutical Bulletin* **37:**1619-1621. https://doi.org/10.1248/cpb.37.1619

Janssen, M.P.M., Traas, T.P., Rila, J-P. & van Vlaardingen, P.L.A. 2004. Guidance for deriving Dutch environmental risk limits from EU-Risk assessment reports of existing substances. National Institute for Public Health and the Environment, Directorate-General for Environmental Protection, Directorate for Chemicals, Waste and Radiation. RIVM, Bilthoven, NL, RIVM Report 601501020/2004. https://www.rivm.nl/bibliotheek/rapporten/601501020.pdf

Jensen, J., Ingvertsen, S.T. & Magid, J. 2012. Risk evaluation of five groups of persistent organic contaminants in sewage sludge. Environmental Project No. 1406 2012. Danish Environmental Protection Agency. https://www2.mst.dk/udgiv/publikationer/2012/05/978-87-92779-69-4.pdf

Johnson, G.R. 2022. PFAS in soil and groundwater following historical land application of biosolids. *Water Research* **211:**118035. https://doi.org/10.1016/j.watres.2021.118035

Kesic, R., Elliott, J.E., Fremlin, K.M., Gauthier, L., Drouillard, K.G. & Bishop, C.A. 2021. Continuing persistence and biomagnification of DDT and metabolites in northern temperate fruit orchard avian food chains. *Environmental Toxicology and Chemistry* **40:**3379-3391. https://doi.org/10.1002/etc.5220

Kominko, H., Gorazda, K. & Wzorek, Z. 2024. Sewage sludge: A review of its risks and circular raw material potential. *Journal of Water Process Engineering* **63:**105522. https://doi.org/10.1016/j.jwpe.2024.105522

Košnár, Z., Mercl, F., Pierdonà, L., Chane, A.D., Míchal, P. & Tlustoš, P. 2023. Concentration of the main persistent organic pollutants in sewage sludge in relation to wastewater treatment plant parameters and sludge stabilization. *Environmental Pollution* **333:**122060. https://doi.org/10.1016/j.envpol.2023.122060

Kuramochi, H., Kawamoto, K., Miyazaki, K., Nagahama, K., Maeda, K., Li, X-W., Shibata, E., Nakamura, T. & Sakai, S-I. 2008. Determination of physicochemical properties of tetrabromobisphenol A. *Environmental Toxicology and Chemistry* **27:**2413-2418. https://doi.org/10.1897/07-472.1

Lamastra, L., Suciu, N.A. & Trevisan, M. 2018. Sewage sludge for sustainable agriculture: contaminants' contents and potential use as fertilizer. *Chemical and Biological Technologies in Agriculture* **5:**10 https://doi.org/10.1186/s40538-018-0122-3

Leslie, H.A., Brandsma, S.H., Barber, J.L., Gabrielsen, G.W., Bersuder, P., Barry, J., Shore, R.F., Walker, L.A. & de Boer, J. 2021. Decabromodiphenylether trends in the European environment: Bird eggs, sewage sludge and surficial sediments. *Science of the Total Environment* **774:**145174 https://doi.org/10.1016/j.scitotenv.2021.145174

Longhurst, P.J., Tompkins, D., Pollard, S.J.T., Hough, R.L., Chambers, B., Gale, P., Tyrrel, S., Villa, R., Taylor, M., Wu, S., Sakrabani, R., Litterick, R., Snary, Leinster, P. & Sweet, N. 2019. Risk assessments for quality-assured, source-segregated composts and anaerobic digestates for a circular bioeconomy in the UK. *Environment International* **127:**253-266. https://doi.org/10.1016/j.envint.2019.03.044

Lorber, M. & Cleverly, D. 2010. In: An exposure assessment of polybrominated diphenyl ethers. EPA/600/R-08/086F. National Centre for Environmental Assessment (NCEA), Office of Research and Development. Washington, D.C 2010. https://assessments.epa.gov/risk/document/&deid=210404

MacGregor, J.A. & Nixon, W.B. 1997. Hexabromocyclododecane (HBCD): Determination of n-octanol/water partition coefficient. Wildlife International LTD. Project No. 439C-104. Sponsored by Chemical Manufacturers Association, Brominated Flame Retardant Industry Panel, Arlington, VA.

Madrid, F., Rubio-Bellido, M. & Morillo, E. 2020. Extraction of nonylphenol, pyrene and phenanthrene from sewage sludge and composted biosolids by cyclodextrins and rhamnolipids. *Science of the Total Environment* **715:**136986 https://doi.org/10.1016/j.scitotenv.2020.136986

Marklund, A., Andersson, B. & Haglund, P. 2005. Organophosphorus flame retardants and plasticizers in Swedish sewage treatment plants. *Environmental Science & Technology* **39:**7423-7429 https://doi.org/10.1021/es051013l

May, W.E., Wasik, S.P., Miller, M.M., Tewarl, Y.B., Brown-Thomas, J.M. & Goldberg, R.N. 1983. Solution thermodynamics of some slightly soluble hydrocarbons in water. *Journal of Chemical & Engineering Data* **28:**197-200 https://doi.org/10.1021/je00032a021

McCarthy, L.H., Loyo-Rosales, J.E., Raby, M., Payne, M., Huber, A., Lavery, J., Caputa, K., Taylor, L. & Falls, A. 2015. Risks associated with application of municipal biosolids to agricultural lands in a Canadian context. Literature review prepared for Canadian Municipal Water Consortium. https://cwn-rce.ca/wp-content/uploads/2015/08/McCarthy-Risks-Biosolids-2015.pdf

McGrath, S.P., Chaudri, A.M. & Giller, K.E. 1995. Long-term effects of metals in sewage sludge on soils, microorganisms and plants. *Journal of Industrial Microbiology* **14:**94-104 https://doi.org/10.1007/BF01569890

Miller, M.M., Wasik, S.P., Huang, G-L., Shiu, W-Y. & MacKay, D. 1985. Relationships between octanol-water partition coefficients and aqueous solubility. *Environmental Science & Technology* **19:**522-529 https://doi.org/10.1021/es00136a007

Miyata, H., Takayama, K., Mimura, M., Kashimoto, T. & Fukushima, S. 1989. Specific congener profiles of polychlorinated dibenzo-p-dioxins and dibenzofurans in blue mussel in Osaka Bay in Japan: Aqueous solubilities of PCDDs and PCDFs. *Bulletin of Environmental Contamination and Toxicology* **43:**342-349 https://doi.org/10.1007/BF01701867

Morris, S., Allchin, C.R., Zegers, B.N., Haftka, J.J.H., Boon, J.P., Belpaire, C., Leonards, P.E.G., van Leeuwen, S.P.J. & de Boer, J. 2004. Distribution and fate of HBCD and TBBPA brominated flame retardants in north sea estuaries and aquatic food webs. *Environmental Science & Technology* **38:**5497-5504. https://doi.org/10.1021/es049640i

Murray, J.J., Pottie, R.F. & Pupp, C. 1974. The vapor pressures and enthalpies of sublimation of five polycyclic aromatic hydrocarbons. *Canadian Journal of Chemistry* **52:**557-563 https://doi.org/10.1139/v74-087

NCCT (National Center for Computational Toxicology). 2024. https://19january2017snapshot.epa.gov/aboutepa/about-national-center-computational-toxicology-ncct_.html

Navarro, I., de la Torre, A., Sanz, P., Pro, J., Carbonell. G. & de los Ángeles Martínez, M. 2016. Bioaccumulation of emerging organic compounds (perfluoroalkyl substances and halogenated flame retardants) by earthworm in biosolid amended soils. *Environmental Research* **149:**32-39. https://doi.org/10.1016/j.envres.2016.05.004

Nestrick, T.J., Lamparski, L.L. & Townsend, D.I. 1980. Identification of tetrachlorodibenzo-p-dioxin isomers at the 1-ng level by photolytic degradation and pattern recognition techniques. *Analytical Chemistry* **52:**1865-1874. https://doi.org/10.1021/ac50062a021

Öko-Onstitute. 2019. RHOS Annex II Dossier for TBBP-A Restriction proposal for substances in electrical and electronic equipment under RoHS. Report No. 2, Tetrabromobisphenol A (TBBP-A, flame retardant) (2,2',6,6'-tetrabromo-4,4'-isopropylidenediphenol) CAS 79-94-7, Version 2, 04/12/2019. TBBPA_RoHS_Dossier_V2_final_20191204.pdf (oeko.info)

Österås, A.H., Allmyr, M. & Sternbeck, J. 2015. Screening of organic pollutants in sewage sludge amended arable soils. Swedish Environmental Protection Agency, National Environmental Monitoring Commissioned by the Swedish EPA, WSP Sverige AB, Stockholm. naturvardsverket-rapport-screening-of-organic-pollutants-in-sewage-sludge-amended-arable-soils_151124-2.pdf (svensktvatten.se)

Pearlman, R.S., Yalkowsky, S.H. & Banerjee, S. 1984. Water solubilities of polynuclear aromatic and heteroaromatic compounds. *Journal of Physical and Chemical Reference Data*. **13:**555-562. https://doi.org/10.1063/1.555712

Plassmann, M.M., Meyer, T., Lei, Y.D., Wania, F., McLachlan, M.S. & Berger, U. 2011. Laboratory studies on the fate of perfluoroalkyl carboxylates and sulfonates during snowmelt. *Environmental Science & Technology* **45:**6872-6878. https://doi.org/10.1021/es201249d

PubChem. 2023. National Library of Medicine – National Center for Biotechnology Information. https://pubchem.ncbi.nlm.nih.gov/

Rede, D., Tiexeira, I., Delerue-Matos, C. & Fernandes, V.C. 2024. Assessing emerging and priority micropollutants in sewage sludge: environmental insights and analytical approaches. *Environmental Science and Pollution Research* **31:**3152-3168. https://doi.org/10.1007/s11356-023-30963-1

Rivier, P-A., Havranek, I., Coutris, C., Norli, H.R. & Joner, E.J. 2019. Transfer of organic pollutants from sewage sludge to earthworms and barley under field conditions. *Chemosphere* **222:**954-960. https://doi.org/10.1016/j.chemosphere.2019.02.010

Rordorf, B.F. 1987. Prediction of vapour pressures, boiling points and enthalpies of fusion for twenty-nine halogenated dibenzo-p-dioxins. *Thermochimica Acta* **112:**117-122. https://doi.org/10.1016/0040-6031(87)88090-5

Saliu, T.D. & Sauvé, S. 2024. A review of per- and polyfluoroalkyl substances in biosolids: geographical distribution and regulations. *Frontiers in Environmental Chemistry* **5:**1383185. https://www.frontiersin.org/journals/environmental-chemistry/articles/10.3389/fenvc.2024.1383185/full

Science for Environment Policy. 2017. Persistent organic pollutants: towards a POPs-free future; Future Brief 19. Brief produced for the European Commission DG Environment. Bristol: Science Communication Unit, UWE. https://docslib.org/doc/6143359/persistent-organic-pollutants-towards-a-pops-free-future#google_vignette

Scottish Government. 2018. The impacts on human health and environment arising from the spreading of sewage sludge to land CR/2016/23: research reports. https://www.gov.scot/collections/sewage-sludge-research/

Shiu, W.Y., Doucette, W., Gobas ,F.A.P.C., Andren, A. & MacKay, D. 1988. Physical-chemical properties of chlorinated dibenzo-*p*-dioxins. *Environmental Science & Technology* **22:**651-658. https://doi.org/10.1021/es00171a006

Sidhu, H., O'Connor, G., Ogram, A. & Kumar, K. 2019. Bioavailability of biosolids-borne ciprofloxacin and azithromycin to terrestrial organisms: Microbial toxicity and earthworm responses. *Science of the Total Environment* **650:**18-26. https://doi.org/10.1016/j.scitotenv.2018.09.004

Sims, R.C. & Overcash, M.R. 1983. Fate of polycyclic aromatic compounds (PNAs) in soil-plant systems. *Research Reviews* **88:**1-68. https://doi.org/10.1007/978-1-4612-5569-7_1

Šmídová, K., Šerá, J., Bielská, L. & Hofman, J. 2015. Influence of feeding and earthworm density on compound bioaccumulation in earthworms *Eisenia andrei*. *Environmental Pollution* **207:**168-175. https://doi.org/10.1016/j.envpol.2015.09.025

Sonnefeld, W.J., Zoller, W.H. & May, W.E. 1983. Dynamic coupled-column liquid chromatographic determination of ambient temperature vapour pressures of polynuclear aromatic hydrocarbons. *Analytical Chemistry* **55:**275-280. https://doi.org/10.1021/ac00253a022

Stańczyk-Mazanek, E., Stępniak, L. & Kępa, U. 2019. Analysis of migration of polycyclic aromatic hydrocarbons from sewage sludge used for fertilization to soils, surface waters, and plants. *Water* **11:**1270. https://doi.org/10.3390/w11061270

Stenzel, J.I. & Markley, B.J. 1997. Hexabromocyclododecane (HBCD): Determination of the water solubility. Wildlife International LTD. Project No. 439C-105. Sponsored by Chemical Manufacturers Association, Brominated Flame Retardant Industry Panel, Arlington, VA.

Stenzel, J.I. & Nixon, W.B. 1997. Hexabromocyclododecane (HBCD): Determination of the vapor pressure using a spinning rotor gauge. Wildlife International LTD. Project No. 439C-117. Sponsored by Chemical Manufacturers Association, Brominated Flame Retardant Industry Panel, Arlington, VA.

Stewart, M., Northcott, G., Gaw, S. & Tremblay, L.A. 2016. An update on emerging organic contaminants of concern for New Zealand with guidance on monitoring approaches for councils. Prepared by Streamlined Environmental Ltd., Northcott Research Consultants Ltd., University of Canterbury, Cawthron Institute and the University of Auckland for Auckland Council, Greater Wellington Regional Council and Environment Canterbury Regional Council. Auckland Council Technical Report TR2016/006. https://knowledgeauckland.org.nz/media/1271/tr2016-006-update-on-emerging-organic-contaminants-new-zealand.pdf

Stockholm Convention. 2018. Risk profile on perfluorohexane sulfonic acid (PFHxS), its salts and PFHxS-related compounds. Addendum. Stockholm Convention on Persistent Organic Pollutants, Persistent Organic Pollutants Review Committee, Fourteenth Meeting, Rome, 17-21 September 2018, UNEP/POPS/POPRC.14/6/Add.1 https://www.meti.go.jp/shingikai/kagakubusshitsu/shinsa/pdf/238_s06_00.pdf

Sun, S-J., Zhao, Z-B., Li, B., Ma, L-X., Fu, D-L., Sun, X-Z., Thapa, S., Shen, J-M., Qi, H. & Wu, Y-N. 2019. Occurrence, composition profiles and risk assessment of polycyclic aromatic hydrocarbons in municipal sewage sludge in China. *Environmental Pollution* **245:**764-770. https://doi.org/10.1016/j.envpol.2018.11.067

Thomaidi, V.S., Stasinakis, A.S., Borova, V.L. & Thomaidis, N.S. 2016. Assessing the risk associated with the presence of emerging organic contaminants in sludge-amended soil: A county-level analysis. *Science of the Total Environment* **548-549:**280-288. https://doi.org/10.1016/j.scitotenv.2016.01.043

Umlauf, G., Christoph, E.H., Lanzini, L., Savolainen, R., Skejo, H., Bidoglio, G., Clemens, J., Goldbach, H. & Scherer, H. 2011. PCDD/F and dioxin-like PCB profiles in soils amended with sewage sludge, compost, farmyard manure, and mineral fertilizer since 1962. *Environmental Science and Pollution Research* **18:**461-470. https://doi.org/10.1007/s11356-010-0389-9

USEPA. 1987. Health and environmental effects profile for benzo(k)fluoranthene. ECAO-CIN-P229. 1987. Cincinnati, OH: USEPA, Environmental Criteria Assessment Office. https://www.epa.gov/tsca-screening-tools/epi-suitetm-estimation-program-interface

USEPA. 2012. Estimation Program Interface (EPI) Suite. Version 4.11. November 2012. Current version available from https://www.epa.gov/tsca-screening-tools/epi-suitetm-estimation-program-interface

USEPA. 2014. Supplemental File 2: TBBPA Environmental and Wildlife Monitoring Data. Prepared by Versar Incorporated under EPA contract EP-W-10-005. https://www.epa.gov/sites/default/files/2015-09/documents/supplemental_file_2_tbbpa_environmental_monitoring.pdf

USEPA OPERA. 2024. https://repository.qsartoolbox.org/Tools/Details/6703ab01-9529-4f86-814f-6efc49e1f59c

Verbruggen, E.M.J. 2012. Environmental risk limits for polycyclic aromatic hydrocarbons (PAHs) for direct aquatic, benthic, and terrestrial toxicity. National Institute for Public Health and the Environment, Ministry of Health, Welfare and Sport, Netherlands. RIVM Report 607711007/2012. https://www.rivm.nl/bibliotheek/rapporten/607711007.pdf

Vodyanitskii, Y.N. & Yakovlev, A.S. 2016. Contamination of soils and groundwater with new organic micropollutants: A review. *Eurasian Soil Science* **49:**560-569. https://doi.org/10.1134/S1064229316050148

Wang, H., Liu, S., Zhang, C., Wan, Y. & Chang, H. 2020. Occurrence and mass balance of emerging brominated flame retardants in a municipal wastewater treatment plant. *Water Research* **185:**116298. https://doi.org/10.1016/j.watres.2020.116298

Wang, L., Wang, X., Xu, O. & Tian, L. 1986. Determination of the *n*-octanol/water partition coefficients of polycyclic aromatic hydrocarbons be HPLC and estimation of their aqueous solubilities. *Huanjing Kexue Xuebao* **6:**491-497.

Wluka, A-K., Huang, Y., Coenen, L., Dsikowitzky, L. & Schwarzbauer, J. 2021. Structural diversity of organic contaminants in sewage sludge: a comparison of sewage fingerprints from Germany and China. *Discover Water* **1:**4. https://doi.org/10.1007/s43832-021-00004-4

WRAP. 2016. Risk assessment for the use of PAS100 green composts in Scottish livestock production. Report OAV021-004. Waste and Resources Action Programme, Banbury, UK; available on request from risk.assessments@wrap.org.uk

Wydro, U., Wołejko, E., Łozowicka, B. & Jabłońska-Trypuć, A. 2021. Microbial diversity and P content changes after the application of sewage sludge and glyphosate to soil. *Minerals* **11:**1423. https://doi.org/10.3390/min11121423

Xiao, R., Kuang, B., Hu, Y., Wang, Y., Bai, J., Zhang, L., Wang, C., Wei, Z., Zhang, K., Jorquera, M.A., Campos, M., Acuña, J.J. & Pan, W. 2023. Ecological risk assessment of glyphosate and its possible effect on bacterial community in surface sediments of a typical shallow lake, northern China. *Ecotoxicology and Environmental Safety* **249:**114445. https://doi.org/10.1016/j.ecoenv.2022.114445

Yalkowski, S.H., He, Yan, & Jain, P. 2010. Handbook of aqueous solubility data second edition. CRC Press, Boca Raton, FL 2010 ISBN 9780367384173.

Yalkowsky, S.H. & Dannenfelser, R.M. 1992. Aquasol Database of Aqueous Solubility. Version 5 (1992), College of Pharmacy, University of Arizona. https://semspub.epa.gov/work/10/500018326.pdf

Zarietalabad, P., Siemens, J., Hamer, M. & Amelung, W. 2013. Perfluorooctanoic acid (PFOA) and perfluorooctanesulfonic acid (PFOS) in surface waters, sediments, soils and wastewater – A review on concentrations and distribution coefficients. *Chemosphere* **91:**725-732. https://doi.org/10.1016/j.chemosphere.2013.02.024

Zhao, W., Teng, M., Zhang, J., Wang, K., Zhang, J., Xu, Y. & Wang, C. 2022. Insights into the mechanisms of organic pollutant toxicity to earthworms: Advances and perspectives. Environmental Pollution **303**:119120. https://doi.org/10.1016/j.envpol.2022.119120.

Zhou, T., Li, X., Liu, H., Dong, S., Zhang, Z., Wang, Z., Li, J., Nghiem, L.D., Khan, S.J. & Wang, Q. 2024. Occurrence, fate and remediation for per- and polyfluoroalkyl substances (PFAS) in sewage sludge: A comprehensive review. *Journal of Hazardous Materials* **466:**133637. https://doi.org/10.1016/j.jhazmat.2024.133637

Zhou, W., Teng, M., Zhang, J., Wang, K., Zhang, J., Xu, Y. & Wang, C. 2022. Insights into the mechanisms of organic pollutant toxicity to earthworms: Advances and perspectives. *Environmental Pollution* **303:**119120. https://doi.org/10.1016/j.envpol.2022.119120

4. Pharmaceuticals and Personal Care Products

4.1. Background

Pharmaceuticals and personal care products (PPCPs) are used for health or cosmetic purposes, and include products used on both animals and humans. A diverse collection of tens of thousands of chemical substances can be classed as PPCPs, comprising prescription and over-the-counter medicines, veterinary medicines, fragrances and cosmetics.

Pharmaceuticals are designed to modify biochemical and physiological functions of biological systems in humans and animals – these properties can unintentionally influence the behaviour and fate of microorganisms in the soil or water should their habitats become contaminated with these compounds. Major groups of pharmaceuticals include antibiotics including antimicrobials, antivirals, etc., disinfectants, steroids, hormones and nutraceuticals (health benefit products derived from food sources e.g. dietary supplements), as well as their various degradation products. Some pharmaceutical compounds are easily broken down and processed in the human body, whilst in other cases, a significant proportion of the compounds and their metabolites are eliminated through urine or faeces. Through these routes, some pharmaceuticals and their degradation products enter the wastewater stream, and many compounds have been detected in sewage sludge and biosolids produced by wastewater treatment.

Unlike pharmaceuticals, personal care products such as cosmetics, shampoos and lotions, are directly washed into wastewater during showering and bathing, and hence enter wastewater treatment. Many of these compounds and their degradation products have been detected in sewage sludge (e.g. Richardson et al., 2005). Once present in sewage sludge, there are both theoretical and measured pathways by which soil biota can become exposed to PPCPs (Keerthanan et al. 2021; Verlicchi & Zambello, 2015).

Numerous studies report a range of different classes of pharmaceuticals in domestic sewage, including antibiotics, antiepileptics, anticoagulants, analgesics and anti-inflammatories, lipid regulators, steroidal compounds, cosmetics, psychostimulants (Pérez-Lemus et al. 2022; Mejias et al. 2021; Luo et al., 2014). Several studies have showed that the fate and transport of these compounds varies during wastewater treatment, with some compounds completely degraded and some only partly degraded (WHO, 2012).

The half-lives of various PPCPs in sewage sludge applied to land have been investigated by several authors. Walters et al. (2010) collated data from previous studies and compared these with their own empirical evidence from uncontrolled outdoor pot incubation studies. The results indicated that the 'environmental half-lives' in uncontrolled incubation experiments were greater than those from controlled laboratory incubation experiments, as well as values calculated in chemical fate models. This was thought to be due to a variety of factors:

1. Binding of the compounds within the sewage sludge matrix, reducing their bioavailability
2. The presence of complex pharmaceutical mixtures that may inhibit sludge and soil microbial activity and limit degradation
3. The uncontrolled water content of the sludge matrix and amended soils
4. The quantity of readily available nutrients within the sludge matrix and amended soils

5. Uncontrolled ambient temperatures experienced
6. Microbial adaptation / acclimatisation to the compounds of interest
7. The initial concentration of the compounds of interest

To compensate for this potential underestimation in fluxes of PPCPs from the multimedia fugacity modelling used in this study (Section 1.2.2), the exposure assessment of the risk modelling has adopted '*reasonable worst-case*' assumptions as a precautionary approach to estimating exposure (see Section 1.2).

Ecotoxicologic impacts of PCPPs added to soil via sewage sludge are evident and can be seen as an indication of potential impacts on living organisms, soil health, and indirect impacts on humans (Aydin et al. 2022). For example, Carter et al. (2016) showed that earthworms could accumulate selected pharmaceutical compounds (fluoxetine, carbamazepine, diclofenac and orlistat), while Konradi & Vogel (2013) studied cirpofloxacin, sulfamtheoxazole, ofloxcin and clarithromycin; recommending that the presence of these and other antibiotics in sewage sludge is monitored to mitigate risks of change to soil microbial populations. During the 2018 assessment (Scottish Government, 2018) many toxicity and fate data were either absent or insufficient to undertake full quantitative risk assessment for many emerging PPCPs of concern (Scottish Government, 2018; Higgins et al., 2010). While there has been some limited improvement over the past five years, this is largely still the case and the scope and range of risk assessments already undertaken for PCPPs remains similar. Lack of data does not equate to lack of risk.

4.2. Previous risk assessments

Several previous large-scale generalised risk assessments (Gibbs & Jones, 2017 (CIP2 Biosolids assessment); Healy et al., 2017; Jensen et al., 2012; Clarke & Smith, 2011; WEAO 2010; Eriksen et al., 2009; Smith 2009; WEAO 2001) were reviewed as part of the 2018 assessment (Scottish Government, 2018). To this list, the National Chemical Investigations Programme 2020-2022 (CIP3) assessment of emerging contaminants in biosolids (Thornton & Yates, 2023) was added. It was considered that while some of the contaminants listed in the older studies might have changed position on these lists in terms of their relative priority, the basic lists still provided a sound basis for inclusion within the assessment. It should be noted that where an organic chemical is primarily used for non-PCPP purposes, this chemical was not included in the risk assessment presented here, but in the assessment for organic and other emerging chemicals (Section 3). A summary of the previous risk assessments is provided in Table 4.1 below.

Table 4.1 – Chemicals present in sewage sludge and considered in selected risk assessment studies

Imperial College Reviews		**Danish Risk Assessment**	**Norwegian Risk Assessment**	**WEAO Reports**	**EPA Research Report**	**UKWIR National Chemical Investigations Programme CIP2 & CIP3**	
Smith 2009[a]	Clarke & Smith 2011[a]	Jensen et al. 2012	Eriksen et al. 2009	WEAO 2010, 2001	Healy et al. 2017	Gibbs & Jones 2017 (CIP2)	Thornton & Yates 2023 (CIP3)
>1000: Linear alkylbenzene sulphonates (LASs) >100<1000: Nonylphenols Nonylphenol esters >1 <100 Tricosan Triclocarbans Musks	>1000: Steroids: Cholesterol Coprostanol Epicoprostanol Quaternary ammonium compounds (QACs) >100 <1000 Polychlorinated alkanes Polydimethylsiloxanes Steroids: Campesterol Stigmasterol >1 <100 Steroids: E1, E2, E3, EE2	Musks Pharmaceuticals	OP/OPEs NP/NPEs LASs Pharmaceuticals: Atorvastatin Carisoprodol Chloprothixene Dipyridamole Fexofenadine Gabapentin Levetiracetam Losartan Mesalazine MTP Ranitidine Sotalol, TC Chlorophenols[b] Chlorobenzenes[b] TCS[b] Musks (galaxolide, tonalide)[b] BHT[b]	VOCs LASs Estrogenic hormones PCDD/Fs Pharmaceuticals PBDEs Musks Triclosan Triclocarbans	Triclosan Triclocarbans	Diclofenac Ibuprofen Atorvastatin Ortho-hyroxyatorvastatin Para-hydroxyatorvastatin Propranolol Atenolol Erythromycin Norerythromycin Azithromycin Clarithromycin Ciprofloxacin Metformin Ranitidine Carbamazepine Epoxycarbamazepine Sertraline Norsertraline Fluoxetine Tamoxifen Trixylenyl phosphate	Triclosan Galaxolide Tonalide Celestolide Phantolide Traeolide N-siloxanes Atorvastatin Ortho-hyroxyatorvastatin Para-hydroxyatorvastatin Azithromycin Epoxycarbamazepine Ciprofloxacin Climbazole Fluoxetine Norfloxacin Ofloxacin

[a]Values correspond to average concentrations in sewage sludge (mg kg^{-1} DW)
[b]These compounds were not formally risk assessed due to data limitations

4.3. Risk Assessment

Chemicals listed in Table 4.1 were included in the risk assessment if; (i) there primary application was for pharmaceutical or personal care products (PPCPs), (ii) if reliable distribution coefficients were available that satisfied the requirements of the multi-media fugacity models (Section 1.2.2), and (iii) if reliable predicted no effect concentration (PNEC) values were available, at least for aquatic environments. The full list of PPCPs, their physical-chemical properties and PNEC values are listed in Table 4.2.

Table 4.2 – Physico-chemical properties and PNEC values for pharmaceutical and personal care products (PPCPs) entered into the risk assessment.

PPCP name (IUPAC)	Water solubility (mg l^{-1} @ 25°C)	Vapour preddure (mm Hg @ 25°C)	Henry's constant (atm-m^3 mol^{-1})	Log Kow	PNEC (mg kg^{-1})
Triclocarban (3-(4-chlorophenyl)-1-(3,4-dichlorophenyl)urea)	$2.37x10^{-3}$ (USEPA, 2012)	$3.60x10^{-9}$ (USEPA, 2012)	$4.50x10^{-11}$ (NCCT, 2024)	4.90 (USEPA, 2012)	$4.00x10^{-2}$ (Musee, 2018)
Triclosan (5-chloro-2-(2,4-dichlorophenoxy)phenol)	$1.00x10^{1}$ (Yalkowsky et al., 2010)	$4.60x10^{-6}$ (USEPA, 2012)[a]	$2.10x10^{-8}$ (NCCT, 2024)	4.76 (NITE 2012)	$2.00x10^{-2}$ (Musee, 2018)
Carbamazepine (5H-dibenz[b,f]azepine-5-carboxamide)	$1.80x10^{1}$ (USEPA, 2012)	$1.84x10^{-7}$ (USEPA, 2012)	$1.10x10^{-10}$ (NCCT, 2024)	2.45 (Dal Pozzo et al., 1989)	$5.00x10^{-2}$ (Biel-Maeso et al., 2018)
Cyclomethicone 5 (decamethyl-1,3,5,7,9,2,4,6,8,10-pentaoxapentasilicane)	$1.70x10^{-2}$ (Kochetkov et al., 2001)	$3.00x10^{-1}$ (ECHA, 2015)	$3.30x10^{1}$ (NCCT, 2024)	8.06 (Xu et al., 2014)	$1.15x10^{-2}$ (Huygens et al. 2022)
Cyclomethicone 6 (decamethylcyclohexasiloxane)	$5.10x10^{-3}$ (Varaprath et al., 1996)[b]	$1.69x10^{-2}$ (Lei et al., 2010)	$2.50x10^{1}$ (NCCT, 2024)	8.87 (Xu et al., 2014)	$1.15x10^{-2}$ (Huygens et al., 2022)
Caffeine (1,3,7-trimethylxanthine)	$2.16x10^{4}$ (Yalkowsky et al., 2010)	$9.00x10^{-7}$ (Emel'yanenko & Verevkin 2008)	$1.10x10^{-11}$ (NCCT, 2024)	-0.07 (Hansch et al., 1995)	$1.03x10^{-1}$ (Mejías et al., 2021)
Diclofenac (2-{2-[(2,6-dichlorophenyl)amino]phenyl}acetic acid)	$2.37x10^{0}$ (Fini et al., 1986)	$6.14x10^{-8}$ (USEPA, 2012)	$1.55x10^{-10}$ (USEPA OPERA, 2024)	4.51 (Avdeef, 1987)	$3.42x10^{-2}$ (Mejías et al., 2021)
Ibuprofen ((RS)-2-(4-(2-methylpropyl)phenyl)propanoic acid)	$2.10x10^{1}$ (Yalkowsky & Dannenfelser 1992)	$4.74x10^{-5}$ (Daubert & Danner 1989)	$1.50x10^{-7}$ (NCCT, 2024)	3.97 (Avdeef 1993)	$4.65x10^{-2}$ (Mejías et al., 2021)
Atorvastatin ((3R,5R)-7-[2-(4-fluorophenyl)-3-phenyl-4-(phenylcarbamoyl)-5-(propan-2-yl)-1H-pyrrol-1-yl]-3,5-dihydroxyheptanoic acid)	$1.12x10^{-3}$ (USEPA, 2012)	$6.56x10^{-10}$ (USEPA OPERA, 2024)	$2.40x10^{-23}$ (NCCT, 2024)	6.36 (USEPA, 2012)	$6.89x10^{-2}$ (Mejías et al., 2021)
Atenolol ((2RS)-2-[4-(2-hydroxy-3-isopropylaminopropoxy)phenyl]propan-2-ol)	$1.33x10^{4}$ (McFarland et al., 2001)	$1.11x10^{-9}$ (USEPA OPERA, 2024)	$4.35x10^{-10}$ (USEPA OPERA, 2024)	0.16 (Hansch et al., 1995)	$3.75x10^{-1}$ (Mejías et al., 2021)
Erythromycin ((3R,4S,5S,6R,7R,9R,11R,12R,13S,14R)-6-{[(2S,3R,4S,6R)-4-(dimethylamino)phenyl]methyl}oxane-2-carboxylic acid)	$2.00x10^{3}$ (Merck, 2024)	$2.12x10^{-25}$ (USEPA, 2012)	$1.28x10^{-11}$ (USEPA OPERA, 2024)	4.02 (McFarland et al., 1997)	$4.08x10^{-3}$ (Mejías et al., 2021)

Azithromycin (9-deoxo-9a-aza-9a-methyl-9a-homoerythromycin A)	$2.37x10^{0}$ (USEPA, 2012)	$2.65x10^{-24}$ (USEPA, 2012)	$1.33x10^{-11}$ (USEPA OPERA, 2024)	4.02 (McFarland et al., 1997)	$4.10x10^{1}$ (Mejías et al., 2021)
Clarithromycin ((3R,4S,5S,6R)-6-O-methylerythromycin A)	$1.69x10^{0}$ (USEPA, 2012)	$2.32x10^{-25}$ (USEPA, 2012)	$1.01x10^{-10}$ (USEPA OPERA, 2024)	3.16 (McFarland et al 1997)	$6.72x10^{-3}$ (Mejías et al., 2021)
Ciprofloxacin (1-cyclopropyl-6-fluoro-4-oxo-7-(piperazin-1-yl)-1,4-dihydroquinoline-3-carboxylic acid)	$3.00x10^{4}$ (Nowara et al., 1997)	$2.85x10^{-13}$ (USEPA, 2012)	$9.46x10^{-12}$ (USEPA OPERA, 2024)	0.28 (Takács - Novák et al., 1992)	$2.14x10^{-3}$ (Mejías et al., 2021)
Metformin (N,N-dimethylimidodicarbonimidic diamide)	$1.06x10^{6}$ (USEPA, 2012)	$7.58x10^{-5}$ (USEPA, 2012)	$7.60x10^{-16}$ (NCCT, 2024)	-2.64 (USEPA, 2012)	$1.34x10^{1}$ (Mejías et al., 2021)
Ranitidine (N-(2-[(5-[(dimethylamino)methyl]furan-2-yl)methylthio]ethyl)-N'-methyl-2-nitroethene-1,1-diamine)	$2.47x10^{1}$ (Ley, 2000)	$2.99x10^{-9}$ (USEPA OPERA, 2024)	$7.29x10^{-9}$ (USEPA OPERA, 2024)	0.22 (USEPA OPERA, 2024)	$2.90x10^{-4}$ (Helwig et al., 2015)[c]
Sertraline ((1S,4S)-4-(3,4-dichlorophenyl)-N-methyl-1,2,3,4-tetrahydronaphthalen-1-amine)	$3.50x10^{-2}$ (Drugbank, 2024)	$3.36x10^{-6}$ (USEPA OPERA, 2024)	$1.15x10^{-6}$ (USEPA OPERA, 2024)	4.55 (USEPA OPERA, 2024)	$1.58x10^{-3}$ (Mejías et al., 2021)
Norsertraline ((1S,4S)-4-(3,4-dichlorophenyl)-N-methyl-1,2,3,4-tetrahydronaphthalen-1-amine)	$1.04x10^{-6}$ (USEPA OPERA, 2024)	$1.07x10^{-9}$ (USEPA OPERA, 2024)	$6.43x10^{-7}$ (USEPA OPERA, 2024)	4.94 (USEPA OPERA, 2024)	$1.58x10^{-3}$ (Mejías et al., 2021)
Tamoxifen ((Z)-2-[4-(1,2-diphenyl-1-buten-1-yl)phenoxy]-N,N-dimethylethanamine)	$1.67x10^{1}$ (USEPA, 2012)	$3.46x10^{-8}$ (USEPA, 2012)	$2.21x10^{-8}$ (USEPA OPERA, 2024)	6.30 (USEPA, 2012)	$8.10x10^{-5}$ (Orias et al., 2015)[c]
Dipyridamole (2-({6-[bis(2-hydroxyethyl)amino]-4,8-bis(piperidin-1-yl)-[1,3]diazino[5,4-d]pyrimidin-2-yl}(2-hydroxyethyl)amino)ethan-1-ol)	$9.22x10^{2}$ (Human Metabolome Database, 2024)	$9.30x10^{-10}$ (USEPA OPERA, 2024)	$1.21x10^{-11}$ (USEPA OPERA, 2024)	2.06 (USEPA OPERA, 2024)	ND
Fexofenadine ((±)-4-[1-hydroxy-4-[4-hydroxydiphenylmethyl)-1-piperidinyl]-butyl]-α,α-dimethylbenzeneacetic acid)	$2.40x10^{-2}$ (UESPA, 2012)	$2.56x10^{-9}$ (USEPA OPERA, 2024)	$2.51x10^{-8}$ (USEPA OPERA, 2024)	2.81 (USEPA, 2012)	$4.00x10^{-2}$ (Jonsson et al., 2014)[c]
Gabapentin (2-[1-(aminomethyl)cyclohexyl]acetic acid)	$4.49x10^{3}$ (USEPA, 2012)	$2.94x10^{-10}$ (USEPA, 2012)	$4.87x10^{-8}$ (USEPA OPERA, 2024)	-1.10 (Sangster, 2005)	$1.00x10^{0}$ (He et al., 2019; Minguez et al., 2016)[c]
Levetiracetam ((2S)-2-(2-oxypyrrolodin-1-yl)butanamide)	$1.04x10^{5}$ (PDR, 2024)	$3.50x10^{-6}$ (USEPA, 2012)	$1.77x10^{-9}$ (USEPA OPERA, 2024)	-0.49 (USEPA, 2004)	$5.74x10^{-4}$ (Minguez et al., 2016)[c]
Sotalol (N-(4-{1-hydroxy-2-[(propan-2-yl)amino]ethyl}phenyl)methanesulfonamide)	$5.51x10^{3}$ (Drugbank, 2024)	$2.19x10^{-10}$ (USEPA OPERA, 2024)	$1.25x10^{-9}$ (USEPA OPERA, 2024)	0.55 (USEPA OPERA, 2024)	$1.24x10^{-4}$ (Minguez et al., 2016)[c]
Benzothiazole (benzothiazole)	$4.30x10^{0}$ (Human Metabolome Database, 2024)	$1.60x10^{-2}$ (USEPA OPERA, 2024)	$9.53x10^{-6}$ (USEPA OPERA, 2024)	2.05 (USEPA OPERA, 2024)	$7.01x10^{-2}$ (EU, 2008)

[(a)]at 20°C; [(b)]at 23°C; [(c)]derived from an aquatic PNEC

Initial concentrations of PPCPs were set based on the '*realistic worst-case*' paradigm (Longhurst et al., 2019; Scottish Government 2018; Hough et al. 2012). The data used in the 2018 assessment (Scottish Government 2018) was used as an initial starting point, and these data were added to with recent data published since 2018. Concentrations used in the current assessment were derived from 90%ile values from the combined data sets (Table 4.3).

Table 4.3 – Initial concentrations of pharmaceutical and personal care products (PPCPs) in sewage sludge prior to spreading.

Chemical (IUPAC)	Concentration in sewage sludge (mg kg^{-1})	Reference(s)
Triclocarban (3-(4-chlorophenyl)-1-(3,4-dichlorophenyl)urea)	$1.12x10^{2}$	Zhu et al., 2019 USEPA, 2012
Triclosan (5-chloro-2-(2,4-dichlorophenoxy)phenol)	$3.20x10^{0}$	Zhu et al., 2019 USEPA, 2012 Stasinakis et al., 2008
Carbamazepine (5H-dibenz[b,f]azepine-5-carboxamide)	$4.85x10^{-2}$	Pagaling et al., 2023 Thornton & Yates 2023 Aydin et al., 2022 Kodešová et al., 2019 Gibbs & Jones 2017 USEPA, 2012 UKWIR, 2012 JRC, 2012
Cyclomethicone 5 (decamethyl-1,3,5,7,9,2,4,6,8,10-pentaoxapentasilicane)	$2.58x10^{3}$	Harrison et al., 2006
Cyclomethicone 6 (decamethylcyclohexasiloxane)	$2.58x10^{3}$	Harrison et al., 2006
Caffeine (1,3,7-trimethylxanthine)	$4.96x10^{-2}$	USEPA, 2012 JRC, 2012
Diclofenac (2-{2-[(2,6-dichlorophenyl)amino]phenyl}acetic acid)	$2.47x10^{-2}$	Pagaling et al., 2023 Thornton & Yates 2023 Aydin et al., 2022 Gibbs & Jones 2017 UKWIR, 2012 JRC, 2012
Ibuprofen ((RS)-2-(4-(2-methylpropyl)phenyl)propanoic acid)	$2.64x10^{-2}$	Pagaling et al., 2023 Thornton & Yates 2023 Aydin et al., 2022 Gibbs & Jones 2017 UKWIR, 2012 JRC, 2012 USEPA, 2012 Gomez et al., 2007 Carballa, 2004
Atorvastatin	$8.73x10^{-2}$	Thornton & Yates 2023 Gibbs & Jones 2017

((3R,5R)-7-[2-(4-fluorophenyl)-3-phenyl-4-(phenylcarbamoyl)-5-(propan-2-yl)-1H-pyrrol-1-yl]-3,5-dihydroxyheptanoic acid)		
Atenolol ((2RS)-2-[4-(2-hydroxy-3-isopropylaminopropoxy)phenyl]propan-2-ol)	$2.06x10^{-1}$	Pagaling et al., 2023 Thornton & Yates 2023 Aydin et al., 2022 Gibbs & Jones 2017 UKWIR, 2012
Erythromycin ((3R,4S,5S,6R,7R,9R,11R,12R,13S,14R)-6-{[(2S,3R,4S,6R)-4-(dimethylamino)phenyl]methyl}oxane-2-carboxylic acid)	$4.48x10^{-2}$	Pagaling et al., 2023 Thornton & Yates 2023 Aydin et al., 2022 Gibbs & Jones 2017 USEPA, 2012 UKWIR, 2012
Azithromycin (9-deoxo-9a-aza-9a-methyl-9a-homoerythromycin A)	$1.27x10^{-1}$	Thornton & Yates 2023 Aydin et al., 2022 Gibbs & Jones 2017 USEPA, 2012
Clarithromycin ((3R,4S,5S,6R)-6-O-methylerythromycin A)	$1.43x10^{-1}$	Thornton & Yates 2023 Aydin et al., 2022 Gibbs & Jones 2017 USEPA, 2012 UKWIR, 2012
Ciprofloxacin (1-cyclopropyl-6-fluoro-4-oxo-7-(piperazin-1-yl)-1,4-dihydroquinoline-3-carboxylic acid)	$3.71x10^{-1}$	Thornton & Yates 2023 Aydin et al., 2022 Gibbs & Jones 2017 USEPA, 2012 UKWIR, 2012 Vieno et al., 2006 Golet et al., 2003
Metformin (N,N-dimethylimidodicarbonimidic diamide)	$4.00x10^{1}$	Thornton & Yates 2023 Gibbs & Jones 2017 USEPA, 2012 Eggen et al., 2011
Ranitidine (N-(2-[(5-[(dimethylamino)methyl]furan-2-yl)methylthio]ethyl)-N'-methyl-2-nitroethene-1,1-diamine)	$9.00x10^{-2}$	Thornton & Yates 2023 Gibbs & Jones 2017 USEPA, 2012
Sertraline ((1S,4S)-4-(3,4-dichlorophenyl)-N-methyl-1,2,3,4-tetrahydronaphthalen-1-amine)	$1.20x10^{0}$	Thornton & Yates 2023 Gibbs & Jones 2017
Norsertraline ((1S,4S)-4-(3,4-dichlorophenyl)-N-methyl-1,2,3,4-tetrahydronaphthalen-1-amine)	$7.60x10^{-1}$	Thornton & Yates 2023 Gibbs & Jones 2017
Tamoxifen ((Z)-2-[4-(1,2-diphenyl-1-buten-1-yl)phenoxy]-N,N-dimethylethanamine)	$2.35x10^{-2}$	Thornton & Yates 2023 Aydin et al., 2022 Gibbs & Jones 2017
Dipyridamole (2-({6-[bis(2-hydroxyethyl)amino]-4,8-bis(piperidin-1-yl)-[1,3]diazino[5,4-d]pyrimidin-2-yl}(2-hydroxyethyl)amino)ethan-1-ol)	$2.48x10^{-1}$	Okuda et al., 2008
Fexofenadine	$1.70x10^{-4}$	Golovko et al., 2014

((±)-4-[1-hydroxy-4-[4-hydroxydiphenylmethyl)-1-piperidinyl]-butyl]-α,α-dimethylbenzeneacetic acid)		
Gabapentin (2-[1-(aminomethyl)cyclohexyl]acetic acid)	$1.00x10^{-3}$	Writer et al., 2013
Levetiracetam ((2S)-2-(2-oxypyrrolodin-1-yl)butanamide)	$1.25x10^{-2}$	Gurke et al., 2015
Sotalol (N-(4-{1-hydroxy-2-[(propan-2-yl)amino]ethyl}phenyl)methanesulfonamide)	$1.58x10^{-2}$	Aydin et al .,2022 Radjenović et al., 2009
Benzothiazole (benzothiazole)	$6.44x10^{1}$	Harrison et al., 2006

Combining partition coefficients (Table 4.2) with initial concentrations in sewage sludge (Table 4.3) within equations 1.1 – 1.7 (section 1.2.2), it was possible to estimate the concentrations of each PPCP that would remain in the soil amended with sewage sludge, as opposed to migrating to soil pore water or soil pore air (Figure 4.1). Those contaminants with the greatest propensity to partition to the soil pore water are subsequently more likely to leach to surface/groundwater or be taken up by plants including food crops.

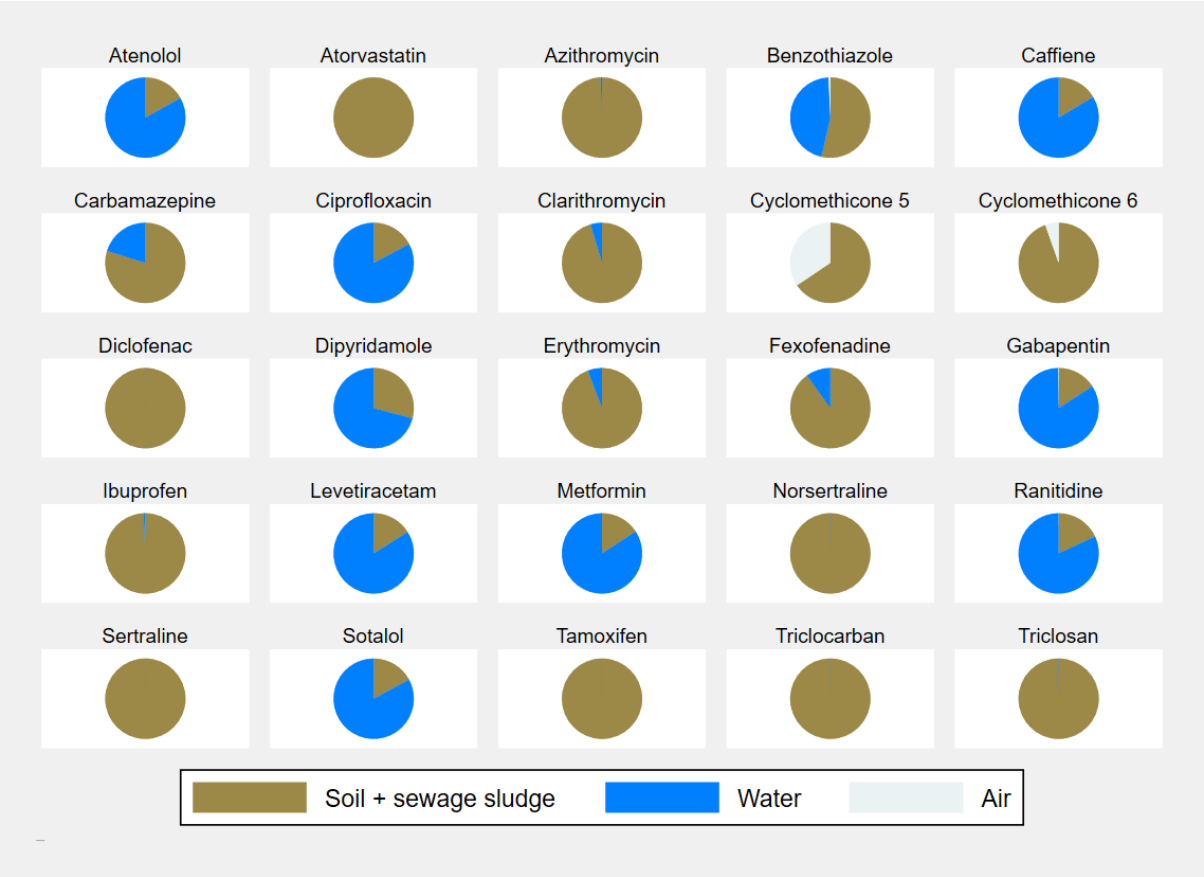

Figure 4.1 – Partitioning [%, ppm] of pharmaceutical and personal care products (PCPPs) between soil amended with sewage sludge, soil pore water, and soil pore air spaces given initial starting concentrations in sewage sludge (Table 4.3)

4.3.1. Risk Characterisation

The current assessment focuses on ecological impacts of the contaminants of interest on soil biota. To this end, the predicted concentrations of each PPCP in the sewage-amended soil fraction were assumed to be equivalent to predicted effect concentrations (PEC) to which soil biota are exposed. Relative risks (or risk ratios; RR) were then derived as the comparison of the PEC to published predicted no-effect level concentrations (PNEC) for soil (Table 4.2, see Section 1.2.4). Figure 4.2 provides a visualisation of RR values for the PPCPs, here using a $\log_{10}$ scale. Thus, any PPCP returning a $\log_{10}$(RR) greater than zero is indicative that we may expect to see an appreciable impact on soil biota as a result of spreading sewage sludge on the land. For convenience, the un-logged values of RR are also listed in Table 4.4.

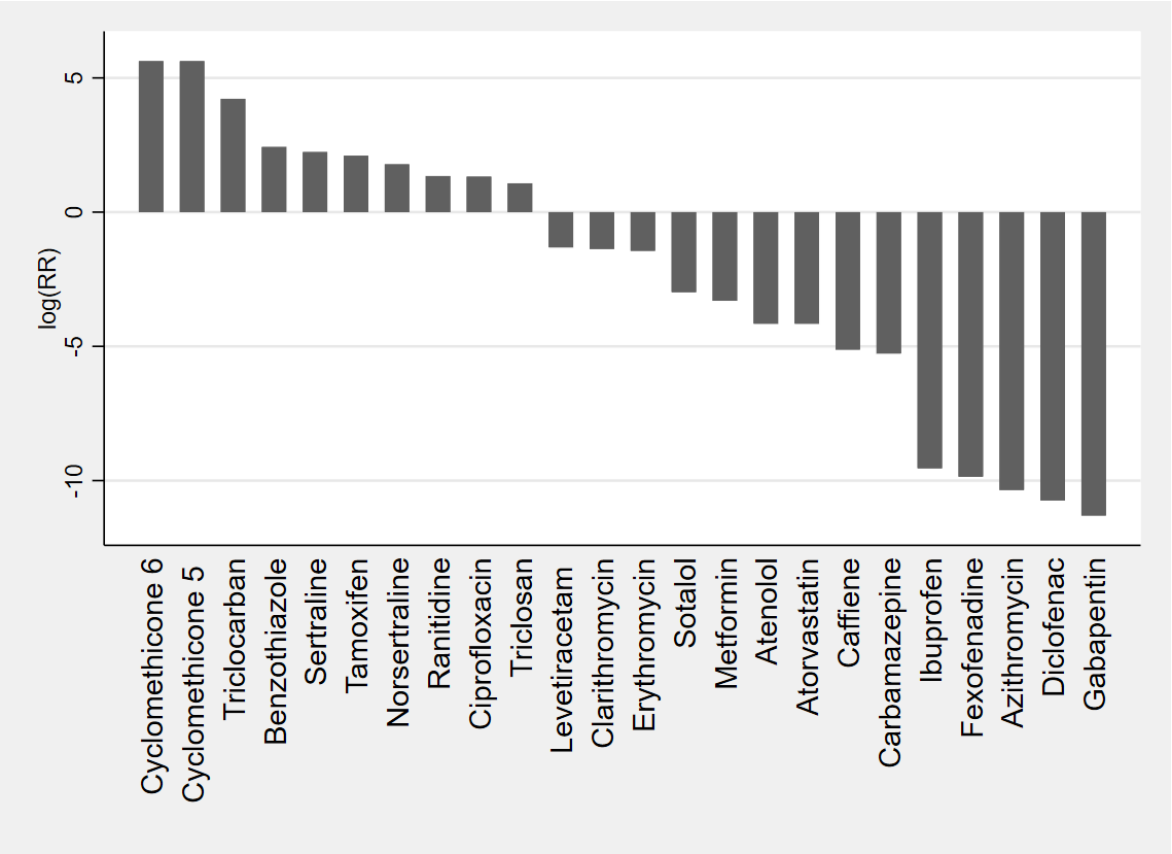

Figure 4.2 – Estimates of relative risk (or risk ratios) presented using a logarithmic scale, log(RR), for each pharmaceutical and personal care product (PPCP) based on the ratio of the predicted effect concentration (PEC; see Figure 4.1) to the predicted no-effect concentration (PNEC; see Table 4.2). These estimates are based on a single application of sewage sludge at 50 t ha^{-1}. A log(RR) > 0 indicates that further investigation may be required.

Table 4.4 – Estimates of relative risk or risk ratio (RR) for each organic contaminant based on the ratio of predicted effect concentration (PEC; see Figure 4.1) to the predicted no-effect concentration (PNEC; see Table 4.2). These estimates are based on a single application of sewage sludge at 50 t $^{ha-1}$. An RR > 1 (shaded in blue) indicates that further investigation may be required.

Contaminant	Risk Ratio, RR
Cyclomethicone 5	2.77×10^{2}
Cyclomethicone 6	2.77×10^{2}
Triclocarban	6.81×10^{1}
Benzothiazole	1.13×10^{1}
Sertraline	9.38×10^{0}
Tamoxifen	8.17×10^{0}
Norsertraline	5.94×10^{0}
Ranitidine	3.83×10^{0}
Ciprofloxacin	3.75×10^{0}
Triclosan	2.93×10^{0}
Levetiracetam	2.69×10^{-1}
Clarithromycin	2.54×10^{-1}
Erythromycin	2.37×10^{-1}
Sotalol	5.07×10^{-2}
Metformin	3.68×10^{-2}
Atenolol	1.57×10^{-2}
Atorvastatin	1.57×10^{-2}
Caffiene	5.96×10^{-3}
Carbamazepine	5.16×10^{-3}
Ibuprofen	7.16×10^{-5}
Fexofenadine	5.25×10^{-5}
Azithromycin	3.19×10^{-5}
Diclofenac	2.17×10^{-5}
Gabapentin	1.23×10^{-5}

4.4. Conclusions

Under a scenario that supports precautionary decision making with a single application of sewage sludge (50 t ha^{-1}), 10 out of 24 PPCP compounds returned a relative risk (or risk ratio) greater than unity (note that it was not possible to assess dipyridamole as no PNEC values were available) and some demonstrating RR values as high as 277. The remaining 14 compounds posed little appreciable risk to soil biota.

Six of the compounds identified as posing a significant risk were also highlighted by the 2018 assessment (Scottish Government 2018), namely, cyclomethicone 5 & 6, triclocarban, benzothiazole, sertraline and tamoxifen. While the current assessment has a different focus to the 2018 assessment (direct ecological impact as opposed to indirect impact on human health via the terrestrial food chain), returning the same compounds as being prominent provides some confidence that these contaminants could have detrimental impacts on environmental and human health. In addition to the six compounds listed above, norsertraline, ranitidine, ciprofloxacin, and triclosan also returned an appreciable risk. Apart from norsertraline, these

four compounds did not feature very highly in the 2018 human health assessment, however it should be noted that for ranitidine and tamoxifen no measured PNEC values were available for the soil environment (Table 4.2).

These findings make it clear that a number of PPCPs are expected to have a detrimental impact on soil health. This assessment cannot ascertain the nature of these impacts (short- or long-term, reversible or irreversible) therefore further empirical evidence gathering is required. As well as impacts on soil biota, human and animal health concerns over increased antimicrobial resistance are noteworthy. The release of persistent pharmaceuticals into the environment has been linked to the acceleration and spread of antimicrobial resistance where species of bacteria, fungi and viruses become immune to commonly used antimicrobial medicines (Frascaroli et al. 2021).

The relative risk estimates presented here are based on a scenario that supports precautionary decision making by using a 'heavy' application of sewage sludge of 50 t ha^{-1}. Typical application rates of sewage sludge should vary from 5 to 20 t ha^{-1} depending on crop nutrient requirements (BAS, 2019). Assuming sludge providers and farmers are following guidance and testing sludges and receiving soils, the majority of agricultural soils are likely to experience lower levels of sewage sludge application The risks highlighted here are important considerations on a site-specific basis. Farm site-specific factors such as cropping system, climate, soil types, land use and historic land management practices and sources of soil contamination should also be considered. Risks could be reduced by using alternative, presumably less contaminated organic fertilisers such as green manures, animal manures or digestates/composts derived from them. This is particularly so where the fertiliser has no uncontrolled elements (cf. on-farm composts with on-farm sourced feedstocks which is highly controlled to source-segregated municipal composts with mixed uncontrolled domestic feedstocks). Similarly, many WWTPs treat industrial wastewater alongside domestic wastewater will result in a more contaminated sewage sludge product. Risks could also potentially be reduced through improved nutrient management if crop nutrient requirements can be met using lower sewage sludge application.

The *realistic worst-case* approach adopted here aims to support precautionary decision making. It uses legitimate maximum values, thus highlights where current legislation is not protective. Hence this approach will derive appreciable risk estimates where an 'average' or 'typical' scenario might not. This approach is helpful as it gives a level of confidence where we identify PPCPs that based on assumptions pose limited risk to soil biota. The cocktail effect caused by multiple exposures to multiple agents is also an important consideration. Little is known about how the compounds assessed here combine, degrade, and change over time. The soil biota are experiencing exposure to this multitude of chemicals and knowledge of how this affects toxicity and impacts is extremely limited. However, it is most likely that the cocktail effect increases the magnitude of risk. Given this, the use of a realistic worst case as opposed to an average or typical scenario is again appropriate, as it is more precautionary and therefore accounts for this uncertainty to some extent.

It should be noted that the assessment has only considered a single application of sewage sludge for the assessment of PPCPs. The 26 PPCPs assessed here (25 where data were sufficient to undertake a full assessment) all behave differently in the environment, some will persist and accumulate, some will degrade, some will move away from the exposure site but will still remain within the wider environment. For some of the PPCPs, these behaviours are known well enough to simulate, but for others, they are not. A simple assessment, where repeated applications are taken as additive will over-estimate risk so was not considered appropriate. For

these reasons, a single application of sewage sludge has been assessed here as was the case in the 2018 assessment (Scottish Government 2018).

As mentioned in Section 3, there are currently no long-term sewage sludge experimental farm plots in the UK, and previous ones have focussed on *traditional* contaminants such as the heavy metals regulated under The Sludge (Use in Agriculture) regulations (e.g. Cd, Pb, Zn). Long-term monitoring data of PCPPs in sludge amended soils is lacking but using the precautionary approach, based on such evidence gathered from previous long-term experiments conducted in the UK, Sweden, Germany and the USA, where metals were found to have adverse effects on soil microbial parameters (cite McGrath et al., 1995) would support soil protection legislation and help inform the safe and sustainable management of sewage sludge. Findings from the ongoing 2-year UK Water Industry Research Chemical Investigations Programme Phase 4 (CIP4 - Introducing the 4th Phase of the Chemical Investigations Programme - Jenni Hughes, UKWIR Strategic Programme Manager) investigating the fate and behaviour of metals, nutrients, and a selection of persistent organic pollutants, microplastics, as well as PCPPs (including antibiotics) will be valuable. The development of new treatment technologies that can effectively remove PPCPs from wastewater e.g., Nyex™ water treatment (Arvia, 2024), and improved source control for PPCPs is also important e.g., manufacturers signing up to initiatives such as the MADE SAFE™ WISE list (MADE SAFE™, 2024).

4.5. References

Arvia. 2024. Nyex™ water treatment. https://arviatechnology.com/arvia-technology-products/nyex-a/

Avdeef, A. 1993. pH-metric log *P*. II: Refinement of partition coefficients and ionization constants of multiprotic substances. *Journal of Pharmaceutical Sciences* **82:**183-190 https://doi.org/10.1002/jps.2600820214

Aydın, S., Ulvi, A., Bedük, F. & Aydın, M.E. 2022. Pharmaceutical residues in digested sewage sludge: Occurrence, seasonal variation and risk assessment for soil. *Science of the Total Environment* **817:**152864 https://doi.org/10.1016/j.scitotenv.2021.152864

BAS. 2019. Biosolids Agricultural Good Practice Guidance Leaflet. Biosolids Assurance Scheme, 2019 Biosolids-Agric-Good-Practice-Guidance-January-2019.pdf (assuredbiosolids.co.uk)

Biel-Maeso, M., Corada-Fernández, C. & Lara-Martín, P.A. 2018. Monitoring the occurrence of pharmaceuticals in soils irrigated with reclaimed wastewater. *Environmental Pollution* **235:**312-321 https://doi.org/10.1016/j.envpol.2017.12.085

Carballa, M., Omif, F., Lema, J.M., Llompart, M., Garcia-Jates, C., Rodriguez, I., Gomez, M. & Ternes, T. 2004. Behaviour of pharmaceuticals, cosmetics and hormones in a sewage treatment plant. *Water Research* **38:**2918-2926 https://doi.org/10.1016/j.watres.2004.03.029

Carter, L.J., Ryan, J.J. & Boxall, A.B.A. 2016. Effects of soil properties on the uptake of pharmaceuticals into earthworms. *Environmental Pollution* **213:**922-931 https://doi.org/10.1016/j.envpol.2016.03.044

Clarke, B.O. & Smith, S.R. 2011. Review of 'emerging' organic contaminants in biosolids and assessment of international research priorities for the agricultural use of biosolids. *Environment International* **37:**226-247 https://doi.org/10.1016/j.envint.2010.06.004

Dal Pozzo, A., Donzelli, G., Rodriquez, L. & Tajana, A. 1989. "In vitro" model for the evaluation of drug distribution and plasma protein-binding relationships. *International Journal of Pharmaceutics* **50:**97-101 https://doi.org/10.1016/0378-5173(89)90133-6

Daubert, T.E. & Danner, R.P. 1989. Physical and thermodynamic properties of pure chemicals data compilation. Washington DC, Taylor & Frances, 1989 ISBN 0891169482:9780891169482

Drugbank. 2024. Database for Drug and Drug Target Info. https://go.drugbank.com/

ECHA. 2015. Search for chemicals. Decamethylcyclopentasiloxane (CAS 541-02-6) Registered Substances Dossier. European Chemical Agency. https://echa.europa.eu/fr/registration-dossier/-/registered-dossier/14807

Eggen, T., Asp, T.N., Grave, K. & Hormazabal, V. 2011. Uptake and translocation of metformin, ciprofloxacin and narasin in forage- and crop plants. *Chemosphere* **85:**26-33 https://doi.org/10.1016/j.chemosphere.2011.06.041

Emel'yanenko, V.N. & Verevkin, S.P. 2008. Thermodynamic properties of caffeine: Reconciliation of available experimental data. *Journal of Chemical Thermodynamics* **40:**1661-1665 https://doi.org/10.1016/j.jct.2008.07.002

Eriksen, G.S., Amundsen, C.E., Bernhoft, A., Eggen, T., Grave, K., Halling-Sørensen, B., Källqvist, T., Sogn, T. & Sverdrup, L. 2009. Risk assessment of contaminants in sewage sludge applied on Norwegian soils. Opinion of the panel on contaminants in the Norwegian Scientific Committee for Food Safety. Norwegian Scientific Committee for Food Safety (VKM). 05/511-22-final https://vkm.no/download/18.645b840415d03a2fe8f1293/1501260413588/2ae7f1b4e3.pdf

EU. 2008. N-Cyclohexylbenzothiazol-2-sulphenamide. European Union Risk Assessment Report. Risk Assessment, May 2008. https://echa.europa.eu/documents/10162/52baf757-f74c-4993-84c8-3bb72195cf55

Fini, A., Fazio, G. & Rabasco, A.M. 1989. /Partición 1-octanol/agua de sales de diclofenac. *Acta Technologiae et Legis Medicamenti* **4:**33-44 [in Italian]

Frascaroli, G., Reid, G., Hunter, C., Roberts, J., Helwig, K., Spencer, J. & Escudero, A. 2021. Pharmaceuticals in wastewater treatment plants: A systematic review on the substances of greatest concern responsible for the development of antimicrobial resistance. *Applied Sciences* **11:**6670 https://doi.org/10.3390/app11156670

Gibbs, H. & Jones, V. 2017. CIP2 Sludge Investigations, UK Water Industry Research 2017 https://ukwir.org/cip2-information

Golet, E.M., Xifra, I., Siegrist, H., Alder, A., Giger, W. 2003. Environmental exposure assessment of fluroquinolone antibacterial agents from sewage to soil. *Environmental Science & Technology* **37:**3324-3249 https://doi.org/10.1021/es0264448

Golovko, O., Kumar, V., Fedorova, G., Randak, T. & Grabic, R. 2014. Seasonal changes in antibiotics, antidepressants/psychiatric drugs, antihistamines and lipid regulators in a wastewater treatment plant. *Chemosphere* **111:**418-426 https://doi.org/10.1016/j.chemosphere.2014.03.132

Gomez, M.J., Martinez Bueno, M.J., Lacorte, S., Fernandez-Alba, A.R. & Aguera, A. 2007. Pilot survey monitoring pharmaceuticals and related compounds in a sewage treatment plant located on the Mediterranean coast. *Chemosphere* **66:**993-1002 https://doi.org/10.1016/j.chemosphere.2006.07.051

Gurke, R., Rößler, M., Marx, C., Diamond, S., Schubert, S., Oertel, R. & Fauler, J. 2015. Occurrence and removal of frequently prescribed pharmaceuticals and corresponding metabolites in wastewater of a sewage treatment plant. *Science of the Total Environment* **532:**762-770 https://doi.org/10.1016/j.scitotenv.2015.06.067

Hansch, C., Leo, A. & Hoekman, D. 1995. Exploring QSAR – Hydrophobic, electronic, and steric constants. Washington DC, American Chemical Society, 1995, ISBN 978-0841229914.

Harrison, E.Z., Oakes, S.R., Hysell, M. & Hay, A. 2006. Organic chemicals in sewage sludges. *Science of the Total Environment* **367:**481-497. https://doi.org/10.1016/j.scitotenv.2006.04.002

Healy, M.G., Fenton, O., Cummins, E., Clarke, R., Peyton, D., Fleming, G., Wall, D., Morrison, L. & Cormican, M. 2017. Health and water quality impacts arising from land spreading of biosolids. EPA Research Report 200. https://www.epa.ie/publications/research/land-use-soils-and-transport/EPA-RR-200_web-Essentra.pdf

Helwig, K., Hunter, C., McNaughton, M., Roberts, J. & Pahl, O. 2015. Ranking prescribed pharmaceuticals in terms of environmental risk: Inclusion of hospital data and the importance of regular review. *Environmental Toxicology and Chemistry* **35:**1043-1050. https://doi.org/10.1002/etc.3302

He, Y., Li, X., Jia, D., Zhang, W., Zhang, T., Yu, Y., Xu, Y. & Zhang, Y. 2019. A transcriptomics-based analysis of the toxicity mechanisms of gabapentin to zebrafish embryos at realistic environmental concentrations. *Environmental Pollution* **251:**746-755. https://doi.org/10.1016/j.envpol.2019.05.063

Higgins, C.P., Sharp, J.O., Sepulvado, J.G., Littrell, B., O'Connor, G., Snyder, E. & McAvoy, D. 2010. Trace organic chemicals in biosolids-amended soils: State-of-the-science review. Water Environment Research Foundation, IWA Publishing. SRSK5T09. https://www.waterrf.org/research/projects/trace-organic-chemicals-biosolids-amended-soils-state-science-review

Hough, R.L., Booth, P., Avery, L.M., Rhind, S., Crews, C., Bacon, J., Campbell, C.D. & Tompkins, D. 2012. Risk assessment of the use of PAS100 green composts in sheep and cattle production in Scotland. *Waste Management* **32:**117-130. https://doi.org/10.1016/j.wasman.2011.08.008

Human Metabolome Database. 2024. HMDB Version 5.0. https://hmdb.ca/

Huygens, D., Garcia-Guiterrez, P., Orvellion, G., Schillaci, C., Delre, A., Orgiazzi, A., Wojda, P., Tonini, D., Egle, L., Jones, A., Pistocchi, A. & Lugato, E. 2022. Screening risk assessment of organic pollutants and environmental impacts from sewage sludge management, EUR 31238 EN, Publications Office of the European Union, Luxembourg, 2022, JRC129690. https://dx.doi.org/10.2760/541579

Jensen, J., Ingvertsen, S.T. & Magid, J. 2012. Risk evaluation of five groups of persistent organic contaminants in sewage sludge. Environmental Project No. 1406 2012. Danish Environmental Protection Agency. https://www2.mst.dk/udgiv/publikationer/2012/05/978-87-92779-69-4.pdf

Jonsson, M., Fick, J., Klaminder, J. & Brodin, T. 2014. Antihistamines and aquatic insects: Bioconcentration and impacts on behavior in damselfly larvae (Zygoptera). *Science of the Total Environment* **472:**108-111. https://doi.org/10.1016/j.scitotenv.2013.10.104

JRC. 2012. Occurrence and levels of selected compounds in European sewage sludge samples – Results of a pan-European screening exercise (FATE SEES). Joint Research Centre. https://op.europa.eu/en/publication-detail/-/publication/ee674d82-c3bf-4686-a272-3e2d0da6b13e

Keerthanan, S., Jayasinghe, C., Biswas, J.K. & Vithanage, M. 2021. Pharmaceutical and personal care products (PPCPs) in the environment: Plant uptake, translocation, bioaccumulation, and human health risks. *Critical Reviews in Environmental Science and Technology* **51:**1221-1258. https://doi.org/10.1080/10643389.2020.1753634

Kochetkov, A., Smith, J.S., Ravikerishna, R., Valsaraj, K.T. & Thibodeaux, L.J. 2001. Air-water partition constants for volatile methyl siloxanes. *Environmental Toxicology and Chemistry* **20:**2184-2188. https://doi.org/10.1002/etc.5620201008

Kodešová, R., Klement, A., Golovko, O., Fér, M., Kočárek, M. 2019. Soil influences on uptake and transfer of pharmaceuticals from sewage sludge amended soils to spinach. *Journal of Environmental Management* **250:**109407 https://doi.org/10.1016/j.jenvman.2019.109407

Konradi, S. & Vogel, I. 2013. Fate of pharmaceuticals (PhCs) in sewage sludge and proposal of indicator substances for monitoring. https://www.umweltbundesamt.de/sites/default/files/medien/376/dokumente/konradi-presentation.pdf

Lei, Y.D., Wania, F. & Mathers, D. 2010. Temperature-dependent vapor pressure of selected cyclic and linear polydimethylsiloxane oligomers. *Journal of Chemical & Engineering Data* **55:**5868-5873. https://doi.org/10.1021/je100835n

Ley, C. (exec ed.). 2000. Ullmann's Encyclopedia of Industrial Chemistry. 7th Edition. Wiley-VCH Verlag GmbH & Co. KGaA. https://onlinelibrary.wiley.com/doi/book/10.1002/14356007

Longhurst, P.J., Tompkins, D., Pollard, S.J.T., Hough, R.L., Chambers, B., Gale, P., Tyrrel, S., Villa, R., Taylor, M., Wu, S., Sakrabani, R., Litterick, R., Snary, Leinster, P. & Sweet, N. 2019. Risk assessments for quality-assured, source-segregated composts and anaerobic digestates for a circular bioeconomy in the UK. *Environment International* **127:**253-266. https://doi.org/10.1016/j.envint.2019.03.044

Luo, Y., Guo, W. & Ngo, H.H., Nghiem, L.D., Hai, F.I., Zhang, J., Liang, S. & Wang, X.C. 2014. A review on the occurrence of micropollutants in the aquatic environment and their fate and removal during wastewater treatment. *Science of the Total Environment* **473-474:**619-641. https://doi.org/10.1016/j.scitotenv.2013.12.065

MADE SAFE™. 2024. WISE List. https://madesafe.org/pages/wise-list-powered-by-made-safe#:~:text=The%20Wise%20List%20bans%20or%20restricts%20over%2015%2C000,-%20Neurotoxicants%20-%20Formaldehyde%20releasers%20-%20Heavy%20metals

McFarland, J.W., Avdeef, A., Berger, C.M. & Raevsky, O.A. 2001. Estimating the water solubilities of crystalline compounds from their chemical structures alone. *Journal of Chemical Information and Computer Sciences* **41:**1355-1359. https://doi.org/10.1021/ci0102822

McFarland, J.W., Berger, C.M., Froshauer, S.A., Hayashi, S.F., Hecker, S.J., Jaynes, B.H., Jefson, M.R., Kamicker, B.J., Lipinski, C.A., Lundy, K.M., Reese, C.P. & Vu, C.B. 1997. Quantitative structure-activity relationships among macrolide antibacterial agents: In vitro and in vivo potency against *Pasteurella multocida*. *Journal of Medicinal Chemistry* **40:**1340-1346. https://doi.org/10.1021/jm960436i

McGrath, S.P., Chaudri, A.M. & Giller, K.E. 1995. Long-term effects of metals in sewage sludge on soils, microorganisms and plants. *Journal of Industrial Microbiology* **14:**94-104. https://doi.org/10.1007/BF01569890

Minguez, L., Pedelucq, J., Farcy, E., Ballandonne, C., Budzinski, H. & Halm-Lemeille, M-P. 2016. Toxicities of 48 pharmaceuticals and their freshwater and marine environmental assessment in northwestern France. *Environmental Science and Pollution Research* **23:**4992-5001. https://doi.org/10.1007/s11356-014-3662-5

Mejías, C., Martín, J., Santos, J.L., Aparicio, I. & Alonso, E. 2021. Occurrence of pharmaceuticals and their metabolites in sewage sludge and soil: A review on their distribution and environmental risk assessment. *Trends in Environmental Analytical Chemistry* **30:**e00125. https://doi.org/10.1016/j.teac.2021.e00125

Merck. 2024. The Merck Index – An encyclopedia of chemicals, drugs, and biologicals. On-line version, Royal Society of Chemistry. https://merckindex.rsc.org/

Musee, N. 2018. Environmental risk assessment of triclosan and triclocarban from personal care products in South Africa. *Environmental Pollution* **242(Part A):**827-838. https://doi.org/10.1016/j.envpol.2018.06.106

NCCT (National Center for Computational Toxicology). 2024. https://19january2017snapshot.epa.gov/aboutepa/about-national-center-computational-toxicology-ncct_.html

NITE. 2012. Chemical Risk Information Platform (CHRIP). Biodegradation and bioconcentration. Tokyo, Japan. https://www.nite.go.jp/en/chem/shiryo/chrip-new-pamph_e.pdf

Nowara, A., Burhenne, J. & Spiteller, M. 1997. Binding of fluoroquinolone carboxylic acid derivatives to clay minerals. *Journal of Agricultural and Food Chemistry* **45:**1459-1463. https://doi.org/10.1021/jf960215l

Okuda, T., Kobayashi, Y., Nagao, R., Yamashita, N., Tanaka, H., Fujii, S., Konishi, C. & Houwa, I. 2008. Removal efficiency of 66 pharmaceuticals during wastewater treatment process in Japan. *Water Science & Technology* **57:**65-71. https://doi.org/10.2166/wst.2008.822

Orias, F., Bony, S., Devaux, A., Durrieu, C., Aubrat, M., Hombert, T., Wigh, A. & Perrodin, Y. 2015. Tamoxifen ecotoxicity and resulting risks for aquatic ecosystems. *Chemosphere* **128:**79-84. https://doi.org/10.1016/j.chemosphere.2015.01.002

Pagaling, E., Troldborg, M. & Zhang, Z. 2023. The transport, fate and impact of pharmaceuticals in the environment in Scotland. Scottish Government, SEFARI 2023. https://sefari.scot/sites/default/files/documents/The%20Transport%2C%20Fate%20and%20Impact%20of%20Pharmacueticals%20in%20the%20Environment%20in%20Scotland.pdf

PDR. 2024. Physicians desk reference. PDR Drug Information. https://www.pdr.net/

Pérez-Lemus, N., López-Serna, R., Pérez-Elvira, S.I. & Barrado, E. 2022. Analysis of 60 pharmaceuticals and personal care products in sewage sludge by ultra-high performance liquid chromatography and tandem mass spectroscopy. *Microchemical Journal* **175:**107148. https://doi.org/10.1016/j.microc.2021.107148

Radjenović, J., Petrović, M. & Barceló, D. 2009. Fate and distribution of pharmaceuticals in wastewater and sewage sludge of the conventional activated sludge (CAS) and advanced membrane bioreactor (MBR) treatment. *Water Research* **43:**831-841. https://doi.org/10.1016/j.watres.2008.11.043

Richardson, B.J., Lam, P.K.S. & Martin, M. 2005. Emerging chemicals of concern: Pharmaceuticals and personal care products (PPCPs) in Asia, with particular reference to Southern China. *Marine Pollution Bulletin* **50:**1230-1237. https://doi.org/10.1016/j.marpolbul.2005.06.034

Sangster, J. 2005. LOGKOW Database. A databank of evaluated octanol-water partition coefficients (Log P). http://www.tds-tds.com/c5/application/files/7314/7360/9080/LOGKOWFS2016e.pdf

Scottish Government. 2018. The impacts on human health and environment arising from the spreading of sewage sludge to land CR/2016/23: research reports. https://www.gov.scot/collections/sewage-sludge-research/

Smith, S.R. 2009. Organic contaminants in sewage sludge (biosolids) and their significance for agricultural recycling. *Philosophical Transactions of the Royal Society* **367:**4005-4041. https://doi.org/10.1098/rsta.2009.0154

Stasinakis, A.S., Gatidou, G., Mamais, D., Thomaidi, N.S. & Lekkas, T.D. 2008. Occurrence and fate of endocrine disrupters in Greek sewage treatment plants. *Water Research* **42:**1796-1804. https://doi.org/10.1016/j.watres.2007.11.003

Takács-Novák, K., Józan, M., Hermecz, I. & Szász, G. 1992. Lipophilicity of antibacterial fluoroquinolones. *International Journal of Pharmaceutics* **79:**89-96. https://doi.org/10.1016/0378-5173(92)90099-N Thornton, A. & Yates, C. 2023. The National Chemical Investigations Programme 2020-2022 Volume 6 – Biosolids products data report. UK Water Industry Research (UKWIR) 22/EQ/01/27. https://ukwir.org/the-national-chemical-investigations-programme-2020-2022-volume-6-biosolids-products-data-report

UKWIR. 2012. Investigation of organic compounds in sludge – Persistent organic pollutants; steroid oestrogens; and selected pharmaceuticals. United Kingdon Water Industry Research 2012.

USEPA. 2012. Estimation Program Interface (EPI) Suite. Version 4.11. November 2012. Current version available from https://www.epa.gov/tsca-screening-tools/epi-suitetm-estimation-program-interface

Varaprath, S., Frye, C.L. & Hamelink, J. 1996. Aqueous solubility of permethylsiloxanes (silicones). *Environmental Toxicology and Chemistry* **15:**1263-1265. https://doi.org/10.1002/etc.5620150803

Verlicchi, P. & Zambello, E. 2015. Pharmaceuticals and personal care products in untreated and treated sewage sludge: Occurrence and environmental risk in the case of application on soil – A critical review. *Science of the Total Environment* **538:**750-767. https://doi.org/10.1016/j.scitotenv.2015.08.108

Vieno, N.M., Tuhkanen, T. & Kronberg, L. 2006. Analysis of neutral and basic pharmaceuticals in sewage treatment plants and in recipient rivers using solid phase extraction and liquid chromatography-tandem mass spectrometry detection. *Journal of Chromatography A* **1134:**101-111. https://doi.org/10.1016/j.chroma.2006.08.077

Walters, E. ,McClellan, K. & Halden, R.U. 2010. Occurrence and loss over three years of 72 pharmaceuticals and personal care products from biosolids-soil mixtures in outdoor mesocosms. *Water Research* **44:**6011-6020. https://doi.org/10.1016/j.watres.2010.07.051

WEAO. 2010. Assessing the fate and significance of microconstituents and pathogens in sewage biosolids. Update to the 2001 WEAO report on fate and significance. Water Environment Association of Ontario. https://www.biosolids.com.au/wp-content/uploads/Assessing-the-Fate-and-Significance-of-Microconstituents-and-Pathogens.pdf

WEAO. 2001. Fate and significance of selected contaminants in sewage biosolids applied to agricultural land through literature review and consultation with stakeholder groups. Water Environment Association of Ontario.

WHO. 2012. Pharmaceuticals in drinking water. World Health Organization, Geneva. https://www.who.int/publications/i/item/9789241502085

Writer, J.H., Ferrer, I., Barber, L.B. & Thurman, E.M. 2013. Widespread occurrence of neuro-active pharmaceuticals and metabolites in 24 Minnesota rivers and wastewaters. *Science of the Total Environment* **461-462:**519-527 https://doi.org/10.1016/j.scitotenv.2013.04.099

Xu, S., Kozerski, G. & MacKay, D. 2014. Critical review and interpretation of environmental data for volatile methylsiloxanes: Partition properties. *Environmental Science & Technology* **48:**11748-11759 https://doi.org/10.1021/es503465b

Yalkowski, S.H., He, Yan, & Jain, P. 2010. Handbook of aqueous solubility data second edition. CRC Press, Boca Raton, FL 2010 ISBN 9780367384173

Yalkowsky, S.H. & Dannenfelser, R.M. 1992. Aquasol Database of Aqueous Solubility. Version 5 (1992), College of Pharmacy, University of Arizona. https://semspub.epa.gov/work/10/500018326.pdf

Zhu, Q., Jia, J., Wang, Y., Zhang, K., Zhang, H., Liao, C. & Jiang, G. 2019. Spatial distribution of parabens, triclocarban, triclosan, bisphenols, and tetrabromobisphenol A and its alternatives in municipal sewage sludges in China. *Science of the Total Environment* **679:**61-69 https://doi.org/10.1016/j.scitotenv.2019.05.059

5. Synthesis

In this section, common threads that cut across all the contaminants assessed in Sections 2 – 4 are discussed.

5.1. Pressures on agricultural soils

Agricultural soils face numerous pressures that threaten their ability to support sustainable food production and maintain ecosystem services. A recent report by Environmental Standards Scotland, highlighted the various risks and threats to our soil resource, including risks presented by current legislation being either disjointed, fragmented or contradictory (Environmental Standards Scotland 2024).

One of the most significant issues is biodiversity loss within soil ecosystems, hence why this is the focus of the current assessment. Intensive farming practices, including monoculture cropping, heavy tillage, and excessive use of agrochemicals have contributed to biodiversity decline (e.g., Osumanu & Kosoe 2023; Stein-Bachinger et al. 2022; Frøslev et al. 2022). The decline in biodiversity affects crucial soil functions such as nutrient cycling, organic matter decomposition, and pest control. Ultimately, reduced soil function impacts negatively on soil health and crop productivity (Parikh & James 2012).

Soil erosion is another major concern for agricultural land. Conventional tillage practices, especially on sloped terrain, expose soil to wind and water erosion. The extent of which is soil type dependent (Montgomery 2007). Soil erosion strips away the nutrient rich topsoil, reducing soil fertility and organic matter content. Soil erosion not only diminished on-site productivity but also causes off-site environmental problems such as sedimentation in water bodies and increased flood risk (Mahabaleshwara & Nagabhushan, 2014). The rate of soil loss often exceeds the natural rate of soil formation, making erosion a critical threat to long-term agricultural stability (Parikh & James 2012). With respect to the current assessment, soil erosion presents the mass transportation of large amounts of contaminants from soils into the aquatic environment and is particularly pertinent to those chemicals with long half-lives including so called *forever chemicals* such as PFAS (Sima & Jaffé 2021).

Nutrient depletion is a growing problem in many agricultural systems, particularly in regions with intensive farming. Continuous cropping without adequate nutrient management leads to exhaustion of soil nutrients, decreased crop yields, and quality (Parikh & James 2012). Sewage sludge and other fertilisers can temporarily address this issue but as we have seen in this assessment, their use can lead to environmental pollution. Organic matter is crucial for nutrient retention, thus organic fertilisers such as sewage sludges, manures, composts do provide some advantages over inorganic alternatives. However, as this assessment shows, it is a delicate balance between maintaining soil nutrient status while minimising negative impacts on soil function.

Soil contamination itself poses significant threats to agricultural soil functions. The accumulation of metals, pesticides and other organic contaminants, pharmaceuticals, plastics, and other pollutants can have long-lasting and irreversible impacts on soil health and food safety (see below). Sources of contamination include agricultural amendments such as sewage sludge, manures, composts, digestates, etc. and industrial activities, waste disposal practices, and overuse of agrochemicals. Agricultural amendments that have an uncontrolled

component, such as being sourced from waste streams that have been source segregated by the general population, or where multiple waste streams are combined for treatment, are likely to be most contaminated. Soils that are contaminated not only pose risks to human health through the food chain but also negatively affect soil biodiversity and overall ecosystem functioning (Parikh & James 2012). Addressing soil contamination often requires complex and costly remediation efforts, highlighting the importance of prevention and a precautionary approach to soil management (Pettersson & Johansson, 2022).

5.2. The *cocktail* effect

While the assessment presented here considered each chemical agent or group of agents in isolation of each other, in reality, application of sewage sludge to agricultural soils introduces these chemicals as a complex mixture that can have synergistic effects on soil biota. This chemical *cocktail* includes microplastics, organic and emerging chemicals, pharmaceutical and personal care products, as well as other agents not covered by the current assessment (e.g., pathogens, resistance genes for antibiotics and agrichemicals, metals including technology critical elements; Pozzebon & Seifert 2023). The combined impact of these contaminants on soil ecosystems can be more significant than the additive effects of the individual compounds alone which is one reason the current assessment has adopted an approach that supports precautionary decision making. For instance, the combined effects of PFAS have been found to alter microbial community functions and reduce the biodiversity and connectivity of soil bacteria (Cao et al. 2022; Wu et al. 2022). Similarly, the presence of microplastics can act as carriers for other environmental contaminants including PCBs, dioxins, PAHs, etc. (e.g., Shi et al. 2020). Microplastics facilitate transport of chemicals from different environmental compartments (soil to water, etc.) and the small size of microplastics allows them to be easily ingested by a wide range of organisms (Godoy et al. 2019). The ability of microplastics to carry chemicals deep into organisms is particularly concerning. The smallest plastics, often called *nanoplastics* can potentially cross biological barriers, including cell membranes (Ašmonaitė et al. 2020). Overall, these *cocktail* effects are difficult to identify and dose-response data that are useable within a risk assessment framework are essentially non-existent. For these reasons, while the *cocktail* effect must be acknowledged, we cannot currently assess it with any suitable level of certainty. Evidence as indicated above suggests that as sewage sludge does include a wide variety of contaminants (many of which are unable to be tested here due to lack of data), the combined effects of chemicals will inevitably raise the risk to the health of soil biota.

5.3. Repeated application of sewage sludge

Repeated application of sewage sludge on agricultural land can have various significant impacts. Primarily, concerns are around the accumulation of contaminants, and this issue was first observed with elevated soil levels of trace metals such as cadmium, copper, mercury, nickel and zinc (Purves 1986). Indeed, the Sludge (Use in Agriculture) regulations and associated Code of Practice was developed bearing in mind the management of heavy metal accumulation in topsoil (Sewage sludge in agriculture: code of practice for England, Wales and Northern Ireland - GOV.UK (www.gov.uk)). In addition, those organic pollutants and pharmaceuticals that display strong sorption behaviours to soil particles (see Figures 3.1 and

4.1), and are persistent (i.e., are slow to degrade) can also accumulate in soil following repeated applications of sewage sludge (e.g., Magid et al. 2020; Pulkrabová et al. 2019). Indeed, some organic chemicals are very persistent with some PFAS taking over 1000 years to degrade in soil (Russell et al. 2008). Microplastics, by their nature will accumulate in soil following repeated applications of material containing MPs (Bondarczuk et al. 2016). There are also reports that repeated applications of sewage sludge can alter microbial communities and enhance the spread of anti-microbial resistance (Bondarczuk et al. 2016). Emerging contaminants, including pharmaceuticals and pharmaceutical residues are poorly understood with regards their long-term impact on soil ecosystems (Bolesta et al. 2022).

A proportion of this accumulation will be irreversible (>lifetime), though there is a tendency for reduced bioavailability of contaminants over time. However, this is only really understood for metals with minimal evidence for organic, pharmaceutical, and plastic pollution (Purves 1986). The soil itself has significant buffering capacity, and this plays a crucial role in mitigating the potential impacts of pollutants introduced through sewage sludge application and other routes. Buffering is particularly high in clay, calcareous, and organic-rich soils, and indeed the organic matter introduced by the application of sewage sludge will also enhance the soils buffering capacity. However, decomposition of organic matter over time will also act to release sorbed contaminants enhancing their availability to organisms. It is well established that conventional soil management techniques such as ploughing enhance degradation of soil organic content (e.g., Stockfisch et al. 1999). The complexity of the chemical mixtures present in sludge and other agricultural amendments (manures, slurries, composts, etc.) makes them difficult to assess and regulate with respect to overall impact on soil and ecosystem health.

In the current assessment, accumulation over repeated applications of sewage sludge was only assessed for microplastics. To achieve this, it was assumed that plastic particles were inert thus the only degradation was via movement out of the system (i.e., by leaching and run-off to surface water). On this basis, the current assessment indicates that microplastics have similar accumulative characteristics to heavy metals.

The current assessment was unable to evaluate accumulation of organic and pharmaceutical chemicals due to data limitations. To account for this, the assessment was undertaken on a single application at the highest legitimate loading of sewage sludge to some extent account for uncertainties.

To try to address some of the data limitations, and unknowns about long term use of sewage sludge, UKWIR have been conducting long-term field experiments on sewage sludge applications. Aspects included have been the effects on soil chemical, biological and physical properties, crop yield and quality, and benefits to soil quality, fertility, and nutrient supply. Similarly, long-term field trials in Sweden (which may be relevant to UK conditions) have been on-going since 1981. These trials have examined the effects of repeated applications of sewage sludge on crop yields, accumulation of organic contaminants including PFAS and brominated flame retardants and impacts on soil organisms such as earthworms.

The UK government is reviewing regulations concerning land application of sewage sludge, septage, and biosolids and there are plans to incorporate the Safe Sludge Matrix UK level voluntary agreement into the new UK Agricultural Use of Sewage Sludge Regulations. Currently, UK legislation (and most other countries) only require testing for a limited set of potentially toxic elements (heavy metals and metalloids) in sewage sludge. There is recognition that this list needs expansion to include contaminants such as PFAS and microplastics.

5.4. Impacts on water quality

As intimated in Sections 2 – 4, the application of sewage sludge to land can have several impacts on water quality. In these chapters, the focus has been on the contaminants of interest, and in this section wider impacts on water quality will be summarised.

Sewage sludge is a fertilizer and is rich in plant nutrients. Excess nutrients, particularly nitrogen and phosphorus, from sewage sludge can leach into groundwater or run off into surface waters. This can lead to eutrophication of water bodies, causing algal blooms and potential oxygen depletion (Sing & Agrawal, 2008).

As discussed above (Section 5.2), many contaminants (metals, organic and emerging chemicals, pharmaceuticals, microplastics and microorganisms/pathogens) will accumulate in soils over time and can migrate to water sources posing risks to aquatic ecosystems and human health (Corradini et al. 2019; Antoniadis et al. 2017; Clarke & Smith 2011; Gerba & Smith 2005). Figures 3.1 and 4.1 indicate the partitioning behaviour of the organic and pharmaceutical contaminants. Those chemicals with a greater propensity to partition to the water phase, will also be those that are most likely to migrate to aquatic environments. Similarly, in Section 2, leaching of microplastics from the soil system was assumed to be 0.2 % d^{-1} (Zhang et al., 2022). These plastics are thus most likely to end up in aquatic ecosystems where they have potential to impact negatively on aquatic organisms. If these microplastics also have chemicals sorbed to their surfaces, changes in chemistry in surface water or inside organisms may result in desorption thus delivering chemical pollutants at different exposure sites (see Section 5.1 above). Effects on water quality may not be immediately apparent but can accumulate over time with repeated applications of sewage sludge (Smith 2009).

The persistence of contaminants is just as relevant in the water environment as it is in the soil environment. Some of the contaminants of concern, such as PFAS, can have incredibly long half-lives reaching 100s if not 1000s of years. This has led the phrase ‘forever chemicals’ to be coined, where *forever* is indicative of >lifetime persistence.

Linking back to the cocktail effect, the complex mixture of interacting chemicals produces various degradation by-products. Some of these will be already present in the wastewater stream prior to water treatment, others will arise during sludge processing, and further degradation by-products will arise once the sewage sludge has been applied to the soil. In nearly all cases, it is difficult to know where or how specific degradation products have arisen unless the process is monitored. Some degradation products will be more soluble that the parent compounds, thus increasing the impact on water quality. One example is TFA which is the extremely persistent degradation product of PFAS. While TFA is already present in wastewater streams prior to treatment and sludge manufacture, further TFA will arise over lifetime timescales as it is the terminal product of many PFAS degradation pathways (Arp et al. 2024). Because of this, while noting that wastewater is only one source of environmental PFAS, TFA has been seen to accumulate in aquatic systems and now accounts for over 90% (by mass) of PFAS in drinking water sources (Neuwald et al. 2022). Another example is aminomethylphosphic acid (AMPA) which is a degradation by-product of glyphosate degradation. The AMPA is significantly more soluble than glyphosate, with global estimates suggesting that over 80% of river inputs (some 4000 tonnes per annum globally) are in the form of AMPA with the other 20% being glyphosate (Zhang et al. 2024). The impact of contamination in sewage sludge is not therefore limited to the site of application, indeed the impacts are far wider once contaminants and their degradation by-products reach the aquatic environment.

5.5. References

Antoniadis, V., Levizou, E., Shaheen, S.M., Ok, Y.S., Sebastian, A., Baum, C., Prasad, M.N.V., Wenzel, W.W. & Rinklebe, J. 2017. Trace elements in the soil-plant interface: Phytoavailability, translocation, and phytoremediation – A review. *Earth Science Reviews* **171:**621-645 https://doi.org/10.1016/j.earscirev.2017.06.005

Arp, H.P.H., Gredelj, A., Glüge, J., Scheringer, M. & Cousins, I.T. 2024. The global threat from the irreversible accumulation of trifluoroacetic acid. *Environmental Science & Technology* **58:**19925–19935. https://doi.org/10.1021/acs.est.4c06189

Ašmonaitė, G., Tivefälth, M., Westberg, E., Magnér, J., Backhaus, T. & Almroth, B.C. 2020. Microplastics as a vector for exposure to hydrophobic organic chemicals in fish: A comparison of two polymers and silica particles spiked with three model compounds. *Frontiers in Environmental Science* **8.** https://doi.org/10.3389/fenvs.2020.00087

Bolesta, W., Glodniok, M. & Styszko, K. 2022. From sewage sludge to the soil – Transfer of pharmaceuticals: A review. *International Journal of Environmental Research and Public Health* **19:**10246. https://doi.org/10.3390%2Fijerph191610246

Bondarczuk, K., Markowicz, A. & Piotrowska-Seget, Z. 2016. The urgent need for risk assessment on the antibiotic resistance spread via sewage sludge land application. *Environment International* **87:**49-55. https://doi.org/10.1016/j.envint.2015.11.011

Cao, L., Xu, W., Wan, Z., Li, G. & Zhang, F. 2022. Occurrence of PFASs and its effect on soil bacteria at a fire-training area using PFOS-restricted aqueous film-forming foams. *iScience* **25:**104084. https://doi.org/10.1016/j.isci.2022.104084

Clarke, B.O. & Smith, S.R. 2011. Review of 'emerging' organic contaminants in biosolids and assessment of international research priorities for the agricultural use of biosolids. *Environment International* **37:**226-247. https://doi.org/10.1016/j.envint.2010.06.004

Corradini, F., Meza, P., Eguiluz, R., Casado, F., Huerta-Lwanga, E. & Geissen, V. 2019. Evidence of microplastic accumulation in agricultural soils from sewage sludge disposal. *Science of the Total Environment* **671:**411-420. https://doi.org/10.1016/j.scitotenv.2019.03.368

Environmental Standards Scotland. 2024. The risks to Scotland's soils: a scoping report. https://environmentalstandards.scot/wp-content/uploads/2024/10/The-risks-to-Scotlands-soils-a-scoping-report-October-2024.pdf

Frøslev, T.G., Nielsen, I.B., Santos, S.S., Barnes, C.J., Bruun, H.H. & Ejrnæs, R. 2022. The biodiversity effect of reduced tillage on soil microbiota. *Ambio* **51:**1022-1033. https://doi.org/10.1007/s13280-021-01611-0

Gerba, C.P. & Smith, J.E. 2005. Sources of oathogenic microorganisms and their fate during land application of wastes. *Journal of Environmental Quality* **34:**42-48. https://doi.org/10.2134/jeq2005.0042a

Godoy, V., Blázques, G., Calero, M., Quesada, L. & Martín-Lara, M.A. 2019. The potential of microplastics as carriers of metals. *Environmental Pollution* **255:**113363. https://doi.org/10.1016/j.envpol.2019.113363

Magid, J., Pedersen, K.E., Hansen, M., Cedergreen, N. & Brandt, K.K. 2020. Chapter Six – Comparative assessment of the risks associated with use of manure and sewage sludge in Danish agriculture. *Advances in Agronomy* **164:**289-334. https://doi.org/10.1016/bs.agron.2020.06.006

Mahabaleshwara, H. & Nagabhushan, H.M. 2014. A study on soil erosion and its impacts on floods and sedimentation. *International Journal of Research in Engineering and Technology* **3:**443-451. https://ijret.org/volumes/2014v03/i15/IJRET20140315086.pdf

Montgomery, D.R. 2007. Soil erosion and agricultural sustainability. *Proceedings of the National Academy of Sciences* **104:**13268-13272. https://doi.org/10.1073/pnas.0611508104

Neuwald, I.J., Hübner, D., Wiegand, H.L., Valkov, V., Borchers, U., Nödler, K., Scheurer, M., Hale, S.E., Arp, H.P.H. & Zahn, D. 2022. Ultra-short-chain PFASs in the sources of German drinking water: Prevalent, overlooked, difficult to remove, unregulated. *Environmental Science and Technology* **56:**6380-6390. https://doi.org/10.1021/acs.est.1c07949

Osumanu, I.K. & Kosoe, E.A. 2023. Global biodiversity deline and loss from agricultural intensification through agrochemical application. In: Ogwu, M.C. & Chibueze Izah, S. (eds.) One health implications of agrochemicals and their sustainable alternatives. *Sustainable Development and Biodiversity* **34:**77-103. https://doi.org/10.1007/978-981-99-3439-3_3

Parikh, S.J. & James, B.R. 2012. Soil: The foundation of agriculture. *Nature Education Knowledge* **3:**2 https://www.nature.com/scitable/knowledge/library/soil-the-foundation-of-agriculture-84224268/

Pettersson, M. & Johansson, O. 2022. How cautious should we be? The role of the precautionary principle in the regulation of sewage sludge in Sweden. *Detritus* **21:**105-113 https://doi.org/10.31025/2611-4135/2022.16227

Pozzebon, E.A. & Seifert, L. 2023. Emerging environmental health risks associated with the land application of biosolids: a scoping review. *Environmental Health* **22:**57 https://doi.org/10.1186/s12940-023-01008-4

Pulkrabová, J., Černy, J., Száková, J., Švarcová, A., Gramblička, T., Hajšlová, J., Balík, J. & Tlustoš, P. 2019. Is the long-term application of sewage sludge turning soil into a sink for organic pollutants?: evidence from field studies in the Czech Republic. *Journal of Soils and Sediments* **19:**2445-2458 https://doi.org/10.1007/s11368-019-02265-y

Purves, D. 1986. Long-term effects of application of sewage sludge to soil on composition of herbage with respect to potentially toxic elements. *Environmental Geochemistry and Health* **8:**11-13 https://doi.org/10.1007/BF02280115

Russell, M.H., Berti, W.R., Szostek, B. & Buck, R.C. 2008. Investigation of the biodegradation potential of a fluoroacrylate polymer product in aerobic soils. *Environmental Science & Technology* **42:**800-807. https://doi.org/10.1021/es0710499

Shi, J., Sanganyado, E., Wang, L., Li, P., Li, X. & Liu, W. 2020. Organic pollutants in sedimentary microplastics from eastern Guangdong: Spatial distribution and source identification. *Ecotoxicology and Environmental Safety* **193:**110356 https://doi.org/10.1016/j.ecoenv.2020.110356

Sima, M.W. & Jaffé, P.R. 2021. A critical review of modeling poly- and perfluoroalkyl substances (PFAS) in the soil-water environment. *Science of the Total Environment* **757:**143793 https://doi.org/10.1016/j.scitotenv.2020.143793

Singh, R.P. & Agrawal, M. 2008. Potential benefits and risks of land application of sewage sludge. *Waste Management* **28:**347-358 https://doi.org/10.1016/j.wasman.2006.12.010

Smith, S.R. 2009. A critical review of the bioavailability and impacts of heavy metals in municipal solid waste composts compared to sewage sludge. *Environment International* **35:**142-156 https://doi.org/10.1016/j.envint.2008.06.009

Stein-Bachinger, K., Preißel, S., Kühne, S. & Reckling, M. 2022. More diverse but less intensive farming enhances biodiversity. *Trends in Ecology & Evolution* **35:**395-396 https://doi.org/10.1016/j.tree.2022.01.008

Stockfisch, N., Forsteruter, T. & Ehlers, W. 1999. Ploughing effects on soil organic matter after twenty years of conservation tillage in Lower Saxony, Germany. *Soil and Tillage Research* **52:**91-101 https://doi.org/10.1016/S0167-1987(99)00063-X

Wu, J-Y., Ding, F-G., Shen, Z-W., Hua, Z.L. & Gu, L. 2022. Linking microbiomes with per- and poly- fluoroalkyl substances (PFASs) in soil ecosystems: Microbial community assembly, stability, and trophic phylosymbiosis. *Chemosphere* **305:**135403 https://doi.org/10.1016/j.chemosphere.2022.135403

Zhang, Q., Li, Y., Kroeze, C., Xu, W., Gai, L., Vitsas, M., Ma, L., Zhang, F. & Strokal, M. 2024. A global assessment of glyphosate and AMPA inputs into rivers: Over half of the pollutants are from corn and soybean production. *Water Research* **261:**121986 https://doi.org/10.1016/j.watres.2024.121986

Zhang, X., Chen, Y., Li, X., Zhang, Y., Gao, W., Jiang, J., Mo, A. & He, D. 2022. Size/shape-dependent migration of microplastics in agricultural soil under simulative and natural rainfall. *Science of the Total Environment* **815:**152507. http://dx.doi.org/10.1016/j.scitotenv.2021.152507

6. Overall conclusions

This assessment focussed on the impacts of sewage sludge on soil biota resident in the receiving soil. As such, this assessment represents a more direct exposure compared to the 2018 assessment (Scottish Government 2018) that was focussed on *downstream* impacts of sewage sludge use in agriculture on human health via the food chain. It would therefore be expected that impacts, when present, would be more acute. This was certainly the case using scenarios designed to simulate *realistic worst-case* scenarios where 21 potentially hazardous agents were estimated to pose a risk to soil biota. Those hazards that were prominent in the 2018 assessment, returned a higher magnitude of risk in the current assessment. This makes sense given the more direct nature of the exposure and the possible increased vulnerability of soil biota compared to human beings. Note that risks to human health also merit re-assessment in the future, especially considering new scientific evidence that supports the need for updated human health risk assessments.

The 21 highlighted agents are listed in table 5.1 along with possible mitigation measures. It should be noted that the chemicals included in the assessment were those that were possible to assess in a quantitative manner, with significantly more chemicals remaining unknown. It should also be noted that these potential risks have been identified as part of a robust theoretical mathematical exercise, successfully used in previous risk assessments (Scottish Government 2018; Longhurst et al., 2019: Hough et al., 2012).

Table 5.1 – List of potentially hazardous agents for which a quantitative estimate of appreciable risk to soil biota was possible

Potentially hazardous agent	Relative risk outcome	Magnitude of risk (realistic worst case)	Uncertainty	Possible mitigation
Microplastics	RR>1 after 3 – 10 applications	Medium	Medium	Improved wastewater treatment processes – advanced membrane bioreactor technology can be effective
PCBs 118, 180	RR>1 after 1 application	Medium - High	Medium	Manufacture of PCBs already banned
PCBs 95, 52, 174	RR>1 after 1 application	Medium - Low	Medium	Manufacture of PCBs already banned
Nonylphenol	RR>1 after 1 application	High	Medium	• AD may reduce concentrations, but evidence compounded by ready transformation of NP2EO to NP • Activated carbon or biochar promising sorbant
Nonylphenol-diethoxylate	RR>1 after 1 application	High	High	
PFOS	RR>1 after 1 application	low	Medium	

PFHxS	RR>1 after 1 application	low	High	Improved treatment technologies all show promise: • Activated carbon or biochar • Ion Exchange • Membrane filtration • Advance oxidation • Electrochemical treatment
PFOA	RR>1 after 1 application	low	Medium	
Cyclomethicone 5	RR>1 after 1 application	Medium - High	Medium	• Cycolmethicone 5 readily degraded by aerobic treatment and AD • Cyclomethicone 6 more likely to remain in sludges
Cyclomethicone 6	RR>1 after 1 application	Medium - High	Medium	
Triclocarban	RR>1 after 1 application	Medium	Medium	Partial removal by thermal hydrolysis
Benzothiazole	RR>1 after 1 application	Medium	High	Should be removed during aerobic treatment
Sertraline	RR>1 after 1 application	Medium	Medium	Nyex™ water treatment process is considered industry leading solution (Arvia, 2024)
Norsertraline	RR>1 after 1 application	Low - Medium	Medium	
Tamoxifen	RR>1 after 1 application	Medium	High	Ozonation effectively removes tamoxifen from wastewater
Ranitidine	RR>1 after 1 application	Low	High	• Activated carbon/biochar • Advanced oxidation / ozonation • Membrane filtration
Ciprofloxacin	RR>1 after 1 application	Low	Medium	• Sorption via ceramsite or magnetic metal-organic frameworks • Advanced oxidation / ozonation
Triclosan	RR>1 after 1 application	Low	Medium	Activated sludge treatment is generally effective for removal

This assessment has focussed on three priority contaminant groups, microplastics, organic and other emerging chemicals, and PPCPs, in support of the scientific interests of Fidra and emerging best practice and legislature. This analysis has not attempted to update all the hazard categories covered by the 2018 assessment (Scottish Government, 2018) and thus cannot draw any conclusions about other areas such as pathogens, anti-microbial resistance, and emerging

unregulated metals (such as the technology-critical elements). Sewage sludge is a very complex medium and its composition changes over time. Thus, the catalogue and magnitude of hazards is not fixed which makes sewage sludge particularly challenging to assess in a definitive way. As water treatment technology improves, a different set of hazards are likely to become priority and the current sludge use in agriculture regulations must be updated. Protecting environmental health is an on-going challenge.

Overall, the direct exposure of soil and water ecosystems to contaminated sewage sludge can likely lead to acute impacts. These impacts are not exclusive to sewage sludge and can also be caused by the various other agricultural amendments containing the same variety and degree of potential hazards. Sewage sludge does have the potential to be a useful circular resource, adding nutrients and organic matter to the soil which benefits soil biota. However, how much these benefits outweigh the impacts from chemical and physical exposures can only currently be estimated. Whether any appreciable risks are long-lasting or reversible is unknown, as are the impacts from multiple applications over the years. There is also limited data available to determine long-term impacts on the soil ecosystem, which, paired with the unknown impacts of the 'cocktail effect' of these potential hazards and assumed persistence, indicates that a precautionary approach may be required.

6.1. Recommendations

1. **Adopt the Precautionary Principle:** Given the persistence, bioaccumulation and emerging nature of many contaminants, the report advocates for the adoption of precautionary measures to protect soil health and the wider environment. This may involve limiting the application of sewage sludge on agricultural land until more comprehensive risk data are available and includes adopting recommendations 2 – 7.
2. **Improve Wastewater Treatment:** The report calls for the enhancement of wastewater treatment processes to reduce the levels of contaminants, especially microplastics, organic contaminants, and PPCPs, in sewage sludge. This includes upgrading treatment facilities and introducing stricter regulations on pollutant discharge.
3. **Increase Research on Emerging Contaminants:** The study emphasizes the need for further research into the environmental risks posed by unregulated microplastic and emerging chemical contaminants. More data are needed to understand the long-term impacts of these substances on soil biota and the wider environment.
4. **Regular Monitoring of Contaminant Levels:** The report recommends ongoing monitoring of chemical and microplastic contaminant levels in sewage sludge and agricultural soils. This will help identify trends in contamination and enable timely interventions to minimise environmental damage.
5. **Develop Alternative Treatment Technologies:** To address the limitations of current wastewater treatment systems, the report suggests exploring new technologies for removing persistent and other contaminants from sewage sludge.
6. **Develop Enhanced Quality Standards**: Encourage upstream source control solutions for reducing and/or eliminating contaminants in sewage sludge before recycling to land.
7. **Improved Sludge Use in Agriculture Policies**: Widen scope of regulations to enforce appropriate management strategies and best practice to ensure soil health and the wider environment are protected from a more comprehensive range of contaminants. Build in flexibility and review processes, so that regulations adapt to changes in the contamination profile in a timely manner.

6.2. References

Arvia. 2024. Nyex™ water treatment. https://arviatechnology.com/arvia-technology-products/nyex-a/

Hough, R.L., Booth, P., Avery, L.M., Rhind, S., Crews, C., Bacon, J., Campbell, C.D. & Tompkins, D. 2012. Risk assessment of the use of PAS100 green composts in sheep and cattle production in Scotland. *Waste Management* **32:**117-130 https://doi.org/10.1016/j.wasman.2011.08.008

Longhurst, P.J., Tompkins, D., Pollard, S.J.T., Hough, R.L., Chambers, B., Gale, P., Tyrrel, S., Villa, R., Taylor, M., Wu, S., Sakrabani, R., Litterick, R., Snary, Leinster, P. & Sweet, N. 2019. Risk assessments for quality-assured, source-segregated composts and anaerobic digestates for a circular bioeconomy in the UK. *Environment International* **127:**253-266 https://doi.org/10.1016/j.envint.2019.03.044

Scottish Government. 2018. The impacts on human health and environment arising from the spreading of sewage sludge to land CR/2016/23: research reports. https://www.gov.scot/collections/sewage-sludge-research/