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A review of properties, occurrence, fate, and transportation mechanisms of contaminants of emerging concern in sewage sludge, biosolids, and soils: recent advances and future trends

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Treated sewage sludge, or biosolids, are used worldwide to improve soil quality and fertility due to their high organic matter content and micro- and macronutrients. However, biosolids also introduce contaminants into the environment. This review assessed recent publications on the occurrence, environmental fate, and transportation mechanisms of 419 contaminants of emerging concern (CEC) across four matrices: sewage sludge, biosolids, soils, and dust. Among these, 229 CECs were positively detected at least once. The review focuses on various CECs, including 18 therapeutical classes of pharmaceutical products, personal care products (parabens and chlorophenolic antimicrobials), hormones, plastic-related compounds (bisphenols and phthalates), rubber antioxidants (substituted diphenylamines and para-phenylenediamines), and neonicotinoid insecticides. Phthalates dominate biosolid composition, accounting for over 97% of the total weight of CECs investigated, followed by pharmaceuticals (1.87%), personal care products (0.57%), hormones (0.09%), antioxidants (0.07%), and bisphenols (0.05%). The 50 most prevalent contaminants include phthalates [e.g., di (2-ethylhexyl) phthalate-DEHP, butyl benzyl phthalate-BBzP], antifungals (e.g., miconazole-MCZ, ketoconazole-KTZ), chlorophenolic antimicrobials (e.g., triclocarban-TCC, triclosan-TCS), fluoroquinolone antibiotics (e.g., ofloxacin-OFL, ciprofloxacin-CPF), cardiovascular medications (e.g., telmisartan-TMS, propafenone-PPF), analgesics (e.g., salicylic acid-SA, naproxen-NPX), hormones (e.g., mestranol-EEME, progesterone-P), antidepressants (e.g., sertraline-SRT, amitriptyline-AMT), and lipid regulators (e.g., fenofibrate-FNF). Analytical techniques like matrix-assisted laser desorption/ionization mass spectrometry imaging (MALDI-MSI) have emerged as a valuable tool for qualitative analysis, allowing for identifying CECs in biosolids. The environmental fate and transport dynamics of studied CECs are influenced by their physicochemical properties (water solubility, volatility, degradation time, sorption capacity, and bioaccumulation potential) and environmental conditions (temperature, pH, and moisture content). Furthermore, soil characteristics, such as composition, organic matter, and microbial activity, play key roles in their adsorption, degradation, and persistence in soil environments. Additionally,

wastewater treatment processes impact the transformation and removal of CECs, affecting their degradation and partitioning between treated effluents and sewage sludge.

KEYWORDS

sewage sludge, biosolids, soil, emerging contaminants, physicochemical properties, occurrence, fate and transport

1 Introduction

In recent years, a group of contaminants referred to as emerging contaminants (ECs) or contaminants of emerging concern (CECs) has generated extensive research interest because of their toxic potential for humans and the ecosystem, especially since they are usually not regulated (Sauvé and Desrosiers, 2014). These CECs are ubiquitous because they have been detected in several environmental compartments, starting from simple matrices like air (Cao et al., 2022), drinking water (Teymoorian et al., 2023; Wu et al., 2023), surface water (Goeury et al., 2019; Zhong et al., 2022), moderately dense media exemplified by road dusts (Chae et al., 2024) and suspended sediments (Fenni et al., 2022; Shang et al., 2023), but also including more complex substrates such as manure (Solliec et al., 2016; Zahedi et al., 2022), composts (Huang et al., 2023; Tran et al., 2023), municipal wastewaters (Rabii et al., 2014; Vaudreuil et al., 2022), agricultural and urban soils (Cao et al., 2022; Rødland et al., 2023; Zhang et al., 2023), sewage sludge (Boix et al., 2016) and biosolids (Larivière et al., 2017; Pérez-Lemus et al., 2019; Kumar et al., 2022; Munoz et al., 2022; Saliu and Sauvé, 2024).

The sources of CECs encompass various nonpoint and point sources of pollution (Stefanakis and Becker, 2015), including air deposition (Barroso et al., 2019), sweeping runoff (Sidhu et al., 2013), agricultural enrichment, and pesticide application (He et al., 2015), plastic mulching (Hofmann et al., 2023), sediments, soils (vadose and phreatic zones), surface water bodies (rivers, lakes, seas, oceans), groundwater, drinking water, wetlands, humans, and biota (Ashraf et al., 2023). Additional documented sources are landfill sites, septic tanks (Ashraf et al., 2023), domestic and industrial discharges (Smith, 2009), hospital outflows (Vaudreuil et al., 2022), wastewater treatment plants (WWTP) influents and effluents (Rout et al., 2021; Vaudreuil et al., 2022; Ashraf et al., 2023), as well as soil manure and biosolids applications (Hanselman et al., 2003; He et al., 2015; Larivière et al., 2017; Pérez-Lemus et al., 2019).

Biosolids represent nutrient-rich final product of sewage sludge stabilization, achieved through various wastewater techniques including composting, aerobic or anaerobic digestion, lime treatment, or dewatering techniques such as air drying, vacuum filtering, and centrifugation. This process yields an organic solid product that is deemed safe for agricultural applications (Zorpas and Inglezakis, 2012). The United States Environmental Protection Agency (USEPA) categorizes biosolids into two distinct classes based on their pathogen content: Class A, which comprises biosolids that are virtually free of pathogens and can be applied without any site-specific restrictions, and Class B, which may harbor pathogens and thus mandates the implementation of management practices and site limitations to safeguard public health (USEPA, 1994). In Europe, the regulations pertaining to soil protection advocate for the treatment of sewage sludge prior to its application in agricultural practices, without designating it as biosolids (European Communities, 1986).

While the meticulous application of biosolids for soil quality and fertility improvement has been documented in California since 1920 and in Japan since 1950 (Kalavrouziotis et al., 2023), their regulation and control have garnered substantial government attention in the not-so-distant past (USEPA, 1994) given the risks to the ecological and public health that they pose (Girovich, 1996; McBride, 1995; Mannina et al., 2023). From an agricultural stance, amending soils with biosolids supplements organic matter (Zhang, 2017; Kumar et al., 2022; EPA, 2023), macro- (e.g., N, P, S, Ca, Mg) and micro-nutrients (e.g., Mn, Cu, Zn, Mo, K) to cultivable lands (Kim and Owens, 2010; Wijesekara et al., 2016). In contrast, these substances harbor a concentrated presence of contaminants that have adsorbed onto the soil post-wastewater treatment (Ashraf et al., 2023). The chemical composition and characteristics of biosolids (processed sewage) are widely variable and largely influenced by various factors such as their origin, treatment techniques, age, moisture levels, and temperature (Wijesekara et al., 2016; Wang et al., 2009; Zhang, 2017).

Latest literature reviews on the occurrence of CECs in sewage sludge, biosolids, and soils have mainly concentrated on the assessment of particular classes, namely, pharmaceutical products (PPs) (Larivière et al., 2017), pharmaceutical products and personal care products (PPCPs) (Pérez-Lemus et al., 2019; Riva et al., 2021; Ashraf et al., 2023), phthalates (He et al., 2015; Gao et al., 2018), per-and poly-fluoroalkyl substances (PFAS) (Riva et al., 2021; Kumar et al., 2022; Saliu and Sauvé, 2024), endocrine disrupting chemicals (EDCs), surfactants and flame retardants (Kumar et al., 2022), and plastic leachates and particles (Kumar et al., 2022; Hofmann et al., 2023). Others focused on analytical techniques for characterizing them into those complex matrices (Larivière et al., 2017; Martín-Pozo et al., 2021; Kumar et al., 2022), while others aimed at their occurrence, fate, or removal during drinking water and wastewater treatment (Kumar et al., 2022; Yang et al., 2017). This literature review's original contribution is to annotate the occurrence of twenty-seven classes of CECs: 18 therapeutic classes of PPs, two classes of PCPs, natural and synthetic hormones, two classes of plastic-related compounds (bisphenols and phthalates), two classes of rubber and polymer antioxidants (aromatic secondary amines), and neonicotinoid insecticides in complex matrices, namely, sewage sludge, biosolids, soils, and dusts. The review will also present the physicochemical properties of studied CECs, discuss factors determining their behaviors during the wastewater treatment (WWT) process, and debate their environmental fate and transportation mechanisms in soils.

2 Materials and methods

To examine the occurrence of specified CECs in aforesaid environmental compartments, a thorough examination of

scientific literature published between January 2018 and July 2023 was conducted across four reputable academic databases (Web of Science, PubMed, Reaxys, and SciFinder). However, a thorough analysis of articles and reports from before and after this timeframe was also conducted to provide a comprehensive elucidation of the characteristics of CECs, their means of mobility, and their eventual destiny in the environment. The review was carried out through cross-referencing search terms organized into three distinct clusters: the types of matrices under investigation (sewage, biosolids, soil), the main groups of targeted CECs, and thematic research areas (occurrence, transport, fate). The search query was tailored to exclude references related specifically to wastewater influents and effluents to enhance precision and clarity in the retrieved citations. After deduplicating the retrieved articles, a thorough examination of their titles and abstracts was conducted, followed by a classification based on thematic relevance. Finally, we surveyed the references within the acquired publications and performed a targeted search on Google Scholar to obtain any missing information. A total of 151 articles were retained for a thorough and detailed review of the occurrence, environmental transport, and transformation processes of target groups of CECs. Figure 1 summarizes the approach for this literature review.

3 Pharmaceutical products

3.1 Overview

PPs, also known as medicines, medications, or drugs, are preparations administered in conventional and traditional medicine to diagnose, protect (public health), prevent (prophylactic), treat (therapeutic) humans or animals against disease or health conditions, or to restore, correct, and alter organ functions (Daughton and Ternes, 1999; WHO, 2024; Larousse, 2024). PPs are often categorized based on their mechanisms of action or therapeutic applications, and this review examines the presence of PPs across 18 therapeutic classes within sewage sludge and biosolids. These classes are analgesics (narcotic and non-narcotic analgesics, non-steroidal anti-inflammatory drugs-NSAIDs, antipyretics, stimulants, and their metabolites), antibiotics (sulfonamides, quinolones and fluoroquinolones, β-lactams, macrolides, tetracyclines, nitroimidazoles, lincosamides, nitrofurans, oxazolidinones, and ionophores), anticonvulsants and antiepileptics, antidepressants (comprising antipsychotics and anxiolytics), cardiac care medications (antiarrhythmics, antiplatelets, antihypertensives, β-blockers, diuretics, calcium channel blockers), antifungals, lipid regulators (statins and fibrates), antihistamines and bronchodilators, antineoplastics (listing alkylating agents, antimetabolites, and immunosuppressants), erectile dysfunction agents, x-ray contrast media, antidiabetics, antivirals, muscles relaxants, anesthetics, cholinergic drugs, antihelminthics, and antiemetics). A complete enumeration of the covered PPs and their respective therapeutic classifications is available in the Supplementary Appendix B: Sheet 2-Analytes.

3.2 Physical and chemical characteristics: indices of fate and transport

From a chemical standpoint, PPs are engineered to be moderately water-soluble to facilitate physiological distribution, and to be lipophilic to ensure their bioavailability and biological activity (Ikehata et al., 2006). The physicochemical properties of PPs exhibit considerable variability, with aspects such as structural simplicity seen, for instance, in fluorouracil and metformin

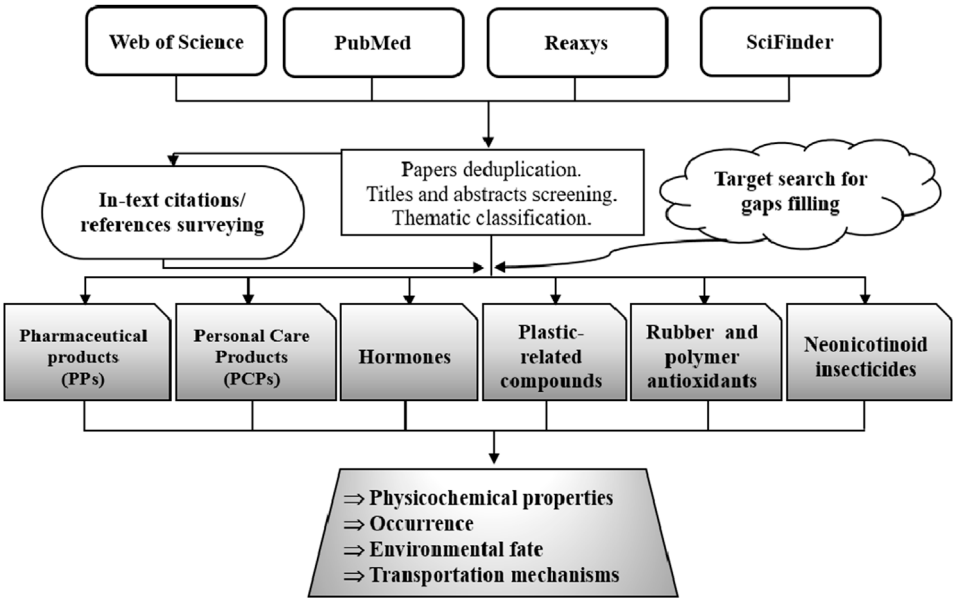

FIGURE 1
Literature review approach.

(MTF), versus structural complexity, as perceived in aclarubicin and roxithromycin. Additionally, these characteristics include molecular weight variations, exemplified by MTF having a molecular weight of 129.16 g/mol and tylosin at 916.10 g/mol. Furthermore, the acid dissociation constant (pK_a) of PPs fluctuates remarkably, as in the case of oxazepam with a pK_a of 1.63 (Shayesteh et al., 2015), oxycodone presenting a pK_a of 8.5 (Kinnunen et al., 2019), and simvastatin with a pK_a of 14.91 (Shaheen et al., 2022). The n-octanol-water partitioning coefficient (log K_{ow}) values also demonstrate notable differences; for example, tetracycline exhibits a log K_{ow} of −1.37 (Chabilan et al., 2022), gemfibrozil (GMF) with a log K_{ow} of 4.39 (Pérez et al., 2023), while mefenamic acid displays a log K_{ow} of 5.12 (Feng et al., 2013). The presence of more than one functional group in a drug's chemical architecture normally causes it to have more than one pK_a value. Gao et al. (2019) proposed a hydrophilicity-hydrophobicity scale for organic contaminants, predicated on the log K_{ow} value. They categorized organic contaminants into hydrophilic (log K_{ow} < 3), moderately hydrophobic (3 < log K_{ow} < 5), and strongly hydrophobic (log K_{ow} > 5). In parallel, the Stockholm Convention on Persistent Organic Pollutants (POPs) designates a log K_{ow} value greater than five to indicate a pollutant's potential for bioaccumulation in aquatic organisms (UNEP, 2019). Table 1 presents important physicochemical properties (solubility, volatility, degradation time, sorption, and bioaccumulation) of organic contaminants, predictors of their environmental destiny, and behaviors.

3.3 Occurrence in sewage sludge and biosolids

In June 2019, the German Environment Agency (UBA) conducted a comprehensive review of the presence of PPs in various environmental matrices to facilitate an update of the Measured Environmental Concentration Database (MEC DB). That review reported a total of 54 environmental matrices

TABLE 1 Physicochemical properties indicative of the transport mechanism and environmental fate of organic contaminants.

Water solubility (*S_w*) FAO (2000)	Solubility (mg/L at 20°C)	Classification
	<0.1	Not soluble
	0.1–1	Slightly soluble
	1–10	Moderately soluble
	10–100	Readily soluble
	>100	Highly soluble
Henry's Law Volatility Constant (*Volatility-H_v*) Howard (2017) Sander et al. (2022)	Volatility (atm-m^3/mol)	Classification[a]
	$<10^{-3}$	Highly volatile
	10^{-3}–10^{-7}	Moderately volatile
	$>10^{-7}$	Slightly volatile
Biological and physicochemical degradation (*Half-life degradation time-DT_{50}*) FAO (2000)	DT_{50} (days)	Classification
	<20	Readily degradable
	20–60	Fairly degradable
	60–180	Slightly degradable
	>180	Very slightly degradable
Sorption (*Organic carbon-water partitioning coefficient-log K_{oc}*) FAO (2000)	Log K_{oc}	Classification
	<1	Highly mobile
	1–2	Mobile
	2–3	Moderately mobile
	3–4	Slightly mobile
	4–5	Hardly mobile
	>5	Immobile
Bioaccumulation (*n-Octanol-water partitioning coefficient-log K_{ow}*) Gao et al. (2019) and UNEP (2019)	Log K_{ow}	Classification
	<3	Moderately bioaccumulative
	3–5	Bioaccumulative
	>5	Highly bioaccumulative

[a]Scale adjusted in three levels

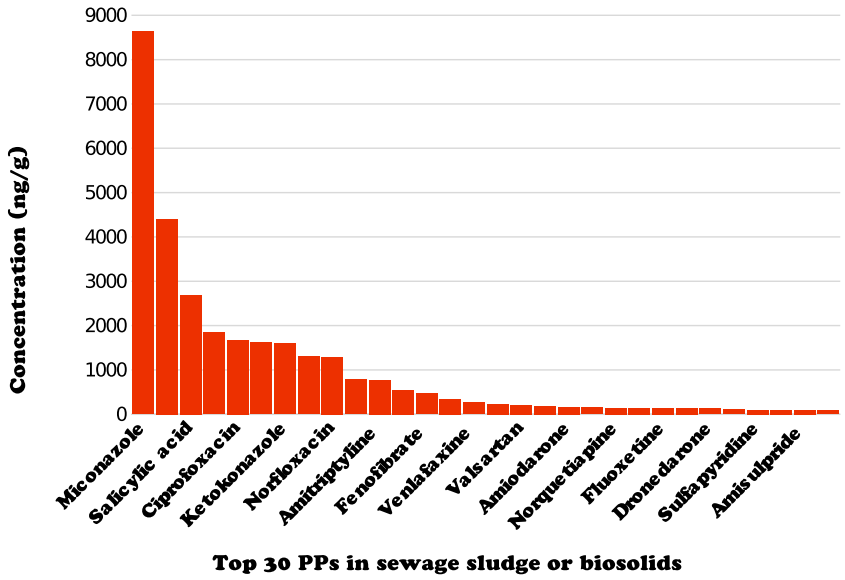

FIGURE 2
Thirty most prevalent PPs in sewage sludge and biosolids.

clustered into liquid emissions (e.g., reclaimed water, hospital sewage), liquid immissions (e.g., drinking water, groundwater), solid emissions (mainly sewage sludge and biosolids), and solid immissions (e.g., soil, lake sediments). The analysis positively (>LOD) determined from seven solid emissions 145 PPs and metabolites across 75 countries (Eike et al., 2019). A subsequent update released in December 2021 broadened this assessment in 89 countries and incorporated an additional category of composts in solid emissions, leading to the identification of 337 PPs from eight solid matrices (Graumnitz and Jungmann, 2021). A comparative analysis between the initial report, which reviewed scientific literature from 2010 to 2016, and the most recent one, which focused on publications from 2017 to 2020, reveals that an additional 192 PPs have been newly and positively identified as originating from solid emissions.

This review encompasses an analysis of academic articles discussing PPs in sewage sludge or biosolids. Moreover, publications about the fate and transport of PPs were examined to clarify their ecological destiny and migration routes during WWT processes and within soil matrices. From a total of 301 PPs and degradation products analyzed, 112 compounds were positively detected at least once in one of the two matrices under consideration. The most concentrated pharmaceuticals are dominated by over-the-counter medicines (antifungals and analgesics), followed by antibiotics and antidepressants. Firstly, miconazole (MCZ: 10,382 ng/g) (Li et al., 2021), followed by naproxen (NPX: 9,355 ng/g) (Pérez-Lemus et al., 2020), ciprofloxacin (CPF: 4,889 ng/g) and ofloxacin (OFL: 4,673 ng/g) (Riva et al., 2021), norfloxacin (NRF: 3,359 ng/g) (Camotti Bastos et al., 2020), ketoconazole (KTZ: 3,009 ng/g) (Svahn and Björklund, 2019), salicylic acid (SA: 2,695 ng/g) (Pérez-Lemus et al., 2020), amitriptyline (AMT: 1,980 ng/g) (Costa Junior et al., 2020), diclofenac (DCF: 1,620 ng/g) (Camotti Bastos et al., 2020), and caffeine (CAF: 1,620 ng/g) (Costa Junior et al., 2020). Figure 2 reveals the mean concentration of the 30 most prevalent PPs in sewage sludge and biosolids. The cumulative mean concentration of various classes of PPs in sewage sludge is primarily dominated by antibiotics (7,689 ng/g), analgesics (4,989 ng/g), antidepressants (2,607 ng/g), cardiovascular medications (1,723 ng/g), and antifungals (1,598 ng/g). Antifungals (8,633 ng/g) are the main class of PPs found in biosolids, followed by cardiovascular agents (3,501 ng/g), antibiotics (1,824 ng/g), antidepressants (1,073 ng/g), and lipid regulators (622 ng/g). The cumulative mean concentration of PPs in sewage sludge (18,949 ng/g) does not substantially differ from that in biosolids (16,324 ng/g). For the summary statistics of various therapeutic classes, refer to Table S1 in Supplementary Appendices A, B-Sheet 3-PPs.

3.4 Environmental fate and transportation mechanisms

In the context of WWT, the characteristics of PPs, the concentration of dissolved organic matter (DOM) that enhances their leaching into effluent, and pH levels emerged as critical factors influencing the environmental fate and transport mechanisms of PPs (Zhang et al., 2014; Silva et al., 2021). Despite the limited availability of treatment technologies specifically addressing PPs and the inadequacy of standard WWT processes in effectively eliminating them, various advanced treatment methods have significantly improved the removal of pharmaceutical compounds during WWT. They include conventional activated sludge, moving bed biofilm reactors (MBBR) (Tisler et al., 2021), ultraviolet C (UV-C) radiation (Grgić et al., 2021), advanced oxidation such as electrochemical oxidation, the use of microalgae and fungal strains, anaerobic membrane bioreactors, as well as membrane separation techniques including nanofiltration (NF) and reverse

osmosis (RO), alongside Fenton and photo-Fenton processes (Shahid et al., 2021).

Interactions of pharmaceuticals with soil matrices depend on their properties such as molecular structure and spatial arrangement, hydrophobicity, polarity, polarizability (Zhi et al., 2019), polar surface area (PSA), as well as soil characteristics like soil type and pH, soil organic matter (SOM), and coexisting ions (Silva et al., 2021). Polar pharmaceutical sorbates mostly interact with SOM through H-bonds and van der Waals forces, while hydrophobic PPs do so through hydrophobic interactions, such as π–π bonds and aromatic ring alignment (Ahmed et al., 2015). Applying the experimental log K_{ow} values for three sulfonamides at pH 5 (for sulfadiazine: 0.14, sulfamethazine: 0.27, and sulfachloropyridazine: 0.69) reported by Carda-Broch and Berthod (2004), Conde-Cid et al. (2020) illustrated that hydrophobicity can better predict the transportation behavior of sulfonamide antibiotics compared to pK_a/ionic species. Movement and transportation pathways within a more homogeneous group of PPs may not be applicable across different classes of PPs, and other characteristics like functional groups of the compounds can significantly influence their environmental persistence and mobility. The pronounced polarity and presence of multi-ionic groups are primarily responsible for the significant soil adsorption capacities exhibited by fluoroquinolones and tetracyclines (Zhi et al., 2019). On the other hand, the PSA can serve as a critical metric for evaluating the capacity of PPs to permeate cellular membranes. Silva et al. (2021) state that compounds exhibiting a PSA greater than 140 square angstroms ($Å^2$) demonstrate diminished permeability across cellular barriers. Erythromycin, which has a PSA of 194 $Å^2$, exemplifies this reduced permeability.

Cationic pharmaceutical species exhibit greater retardation compared to their acidic and neutral counterparts due to the predominantly negatively charged nature of soil surfaces (Schaffer et al., 2012). Various soil parameters significantly influence the adsorption capacities of PPs within soil environments; these include soil texture, clay content, cation exchange capacity (CEC), and anion exchange capacity (AEC). Several soil parameters influence the adsorption capacities of PPs to the soil, including soil texture, clay content and composition, and CEC or AEC. For example, due to greater surface area and CEC, enrofloxacin showed stronger adsorption to montmorillonite (expanded 2:1 clay minerals) than to kaolinite (1:1 clay minerals). The observed reduction in pharmaceuticals sorption with increasing pH can be elucidated through two mechanisms: alterations in charge dynamics (Srinivasan et al., 2013), and modifications in the sorption mechanism (Klement et al., 2018). In the former, as pH rises, sulfamethoxazole (SMX) loses some of its positive charges, and electrostatic repulsion reduces the sorption action; in the latter, in acidic soils, cation bridging and surface complexation of irbesartan (IRB) take the place of the CEC-driven mechanism. As a result, the repulsion of negative charges on the compounds and the soil surface greatly decreased sorption (Xu et al., 2021b).

Numerous investigations demonstrated the toxicity and bioaccumulation potential of certain PPs (Carter et al., 2016; Pullagurala et al., 2018; Almeida and Nunes, 2019). Antibiotics can harm cells, the circulatory system, the metabolism, and the development of fish, they have shown the ability to bioaccumulate in fish liver, lungs, and fillet and they have been detected in dairy products and milk samples and crops can absorb them (Bansal, 2022). Different fish species across various trophic levels selectively absorbed antidepressants. The most significant bioaccumulation occurred in the brain and was directly correlated with WWT effluent exposure. Following the brain, bioaccumulation was observed in the gonads, liver, and muscle tissues (Arnnok et al., 2017). Pérez et al. (2023) upheld pharmaceutical load as a useful bioaccumulation metric in the tissues of bulrushes (*Typha latifolia*). They showed that fluoxetine (FLX) accumulated more in the stem, carbamazepine (CBZ) and FLX in the leaf, and GMF in the root. The pharmaceutical mass load was related to log K_{ow} in the root and rhizome, while in the leaf, it was correlated with pK_a and plant transpiration.

4 Personal care products

Two categories of PCPs and their degradation byproducts were reviewed: parabens and chlorophenolic antimicrobials.

4.1 Overview

4.1.1 Parabens

Parabens, or para-hydroxybenzoic acid esters, are synthetic compounds used as preservatives in cosmetics, pharmaceuticals, and food items since the 1920s. These substances effectively inhibit microbial proliferation and prolong shelf life (Ma et al., 2018; Chen et al., 2021; Stoiber, 2019). Their antimicrobial efficacy is particularly pronounced against molds, yeasts (Ye et al., 2006), and Gram-positive bacteria (Stoiber, 2019). Parabens are EDCs with steroidogenesis disturbance and estrogenic effects (Liang et al., 2023) that can permeate the human body through dermal exposure when employing various products such as moisturizers, facial and body cleansers, UV sunscreens, deodorants, shaving creams, toothpaste, makeup, and other items that contain these substances (Stoiber, 2019), although environmental exposure has also been documented (Chen et al., 2021).

4.1.2 Chlorophenolic antimicrobials

Like parabens, chlorophenolic antimicrobials triclosan (TCS), also known as irgasan, and triclocarban (TCC) have garnered significant attention from regulatory authorities due to their potential as EDCs (Chen et al., 2008) and for their contribution to the spread of multidrug resistance, including antibiotic (Carey and McNamara, 2014; Carey et al., 2016). For this reason, a phase-out notice of TCS and TCC from consumer antiseptic washes in the United States, and the ban on product-type 1 items (human hygiene biocidal products for skin or scalps) in Europe were issued in 2016 by the United States Food and Drug Administration, and the European Commission, respectively (USFDA, 2016; EC, 2016). The Food and Drugs Act in Canada authorizes TCS use in PPs at concentrations ranging from 0.1% to 1.0%. To safeguard human health, TCS is permitted at a concentration not exceeding 0.03% in mouthwashes, and at or below 0.3% in topical formulations and dentifrices (Health Canada, 2024). In October 2020, the Canadian

federal government issued a directive mandating that entities utilizing or importing 100 kg or more of TCS annually must develop a pollution prevention plan specifically for TCS. Individuals subject to the Final Notice in 2020 and 2021 are required to achieve 30% and 95% reductions, respectively, in the total mass of TCS utilized compared to levels established in the base year (Environment Canada, 2020). Regarding TCC, in light of the assessment that the risk it poses to human health is low, the government is contemplating the establishment of a monitoring plan for surface water and sediment, as part of the chemical management plan (CMP) (Environment Canada and Health Canada, 2023).

4.2 Physical and chemical characteristics: indices of fate and transport

4.2.1 Parabens

Parabens are characterized by a benzoic acid backbone featuring a hydroxyl group positioned at the para-position. This structural configuration arises from the esterification of the carboxylic acid function with diverse alcohols (Soni et al., 2005; Ma et al., 2018), thereby contributing to their hydrophobic properties, which in turn influence their solubility (Ma et al., 2018; Chen et al., 2021; Li et al., 2020). The presence of hydroxyl group facilitates H-bonding, thereby enhancing their interactions with biological molecules and potentially contributing to their endocrine-disrupting effects. The significance of parabens as CECs lies in their ability to act as EDCs (Ma et al., 2018). Their ester bonds are attributed to their chemical stability as they can withstand hydrolysis in some situations, although they may break down in extremely acidic or alkaline conditions (Chen et al., 2021). Each paraben exhibits variability in the length and branching of its alkyl chains, which significantly affects its physical and chemical behavior (Chen et al., 2021). Generally, parabens display moderate solubility in water, a property that is contingent upon the length of the alkyl chain; short-chain parabens tend to exhibit greater solubility compared to their long-chain counterparts. This solubility characteristic has implications for their distribution and persistence within aquatic ecosystems and wastewater treatment systems. Additionally, these compounds possess relatively low molecular weights, which can influence their volatility and environmental mobility (Ma et al., 2018). The melting and boiling points of parabens are determined by their molecular structure and chain length, typically increasing with longer alkyl chains. While parabens are subject to degradation in environment, their persistence and behavior can be affected by various factors including temperature, pH levels, and microbial activity (Li et al., 2020).

4.2.2 Chlorophenolic antimicrobials

TCS and TCC are chlorinated aromatic compounds characterized by monochlorophenyl and dichlorophenyl moieties. Both compounds exhibit low solubility in aqueous environments and possess high log K_{ow} values, with TCS at 4.8 and TCC at 4.9 (Coogan et al., 2007; Kwon and Xia, 2012). These properties contribute to their pronounced lipophilicity while reducing their hydrophilicity, indicating a pronounced inclination for partitioning into organic fractions rather than remaining in the aqueous phase (Carey and McNamara, 2014; Armstrong et al., 2019).

4.3 Occurrence in sewage sludge and soils

The occurrence of twenty PCPs grouped into preservatives (eight parabens and four degradation products) and antimicrobials (TCS, TCC, and six metabolites) was evaluated in sewage sludge and soils.

4.3.1 Parabens

The presence of six widely used parabens and 4 byproducts in WWT process and sewage sludge was examined by Ma et al. (2018). The analyzed paraben compounds are methylparaben (MeP), ethylparaben (EtP), propylparaben (PrP), butylparaben (BuP), benzyl paraben (BzP), and heptyl paraben (HepP), and the four metabolites are 4-hydroxybenzoic acid (4-HB), 3,4-dihydroxybenzoic acid (3,4-DHB), methyl protocatechuate (OH-MeP) and ethyl protocatechuate (OH-EtP). Parabens have been detected in wastewater and sludge, indicating frequent contamination. Chen et al. (2021) investigated eight parabens –MeP, EtP, PrP, BuP, BzP, HpP, isopropyl paraben (isoPrP), and isobutyl paraben (isoBuP), and identified their presence in various outdoor environmental media, including soil samples. The prevalence of parabens is predominantly characterized by isobutyl-paraben (iso-BuP), succeeded by 4-hydroxybenzoic acid (4-HB) and 3,4-dihydroxybenzoic acid (3,4-DHB), with respective concentrations measured at 131 ng/g, 100 ng/g, and 86 ng/g d. w.

4.3.2 Chlorophenolic antimicrobials

TCS and TCC were investigated by numerous authors in sewage sludge (Moško et al., 2021; Li et al., 2021; Chen et al., 2019; Malvar et al., 2020b; Malvar et al., 2020a) and soil (Mercl et al., 2021; Malvar et al., 2020b; Chen et al., 2021; Abril et al., 2018). TCC and TCS are the most prevalent PCPs in sewage sludge, with a median concentration of 1,710 and 1,165 ng/g, respectively. Additionally, degradation byproducts such as monocarbanilide (MCC), 2′-hydroxy-triclocarban (2-OH-TCC), carbanilide (CBN), and 3,3′,4,4′-tetrachlorocarbanilide (TCCC) exhibited median concentrations of 520 ng/g, 180 ng/g, 102 ng/g, and 91 ng/g, respectively. These concentrations are within the quantifiable range of their parent compounds, emphasizing the necessity to investigate both the parent substances and their metabolites, as highlighted by Chen et al. (2019). The concentration ranges for examined parabens and antimicrobials in sewage are observed to be between 380 and 81,750 ng/g; notably, antimicrobials predominate over parabens, with the concentration range for the former extending from 132 to 79,985 ng/g. TCS and TCC in soil were below the quantitation limits in the researched studies. Statistics on the individual and cumulative occurrence of PCPs in sewage sludge and soil can be found in Table S2 of Supplementary Appendices A, B- Sheet 4-PCPs.

4.4 Environmental fate and transport mechanisms

4.4.1 Parabens

The main ways that parabens enter the environment are through wastewater and runoff from homes where these compounds are utilized in various domestic items. In the WWT process, parabens can be successfully eliminated by the cyclic-activated sludge

technology and anaerobic-oxic (A/O) treatment methods (Ma et al., 2018). It has also been demonstrated that temperature and pH impact the biodegradation of parabens using aerobic-activated sludge. Indeed, parabens maintain stability in neutral to slightly acidic pH ranges; however, they are susceptible to degradation under extreme pH conditions, which is relevant in WWT processes. Increased residual quantities of parabens were reported at pH 10.0 compared to neutral or slightly acidic conditions, indicating decreased degradation efficiency (Ma et al., 2018). Higher temperatures cause the aqueous phase's residual concentrations to drop. Possibility of aerobic-activated sludge biodegrading parabens, emphasizing the role of this process in WWT (Lu et al., 2018). Parabens are readily biodegradable in soil and unlikely to threaten earthworms' survival and reproduction (Arachchige Chamila Samarasinghe et al., 2021). Nonetheless, due to their widespread use and pseudo-persistence, they are present in the environment and may be more dangerous for other species or toxicological endpoints. Parabens exhibit varied environmental fate as influenced by biodegradation, photodegradation, and sorption to particulate matter (Chen et al., 2021), and organic matter (Mercl et al., 2021). Their transport mechanisms primarily involve water pathways, volatilization into the atmosphere (Chen et al., 2021; Li et al., 2020) with potential contributions from atmospheric and soil movement. Some parabens may bioaccumulate, leading to higher concentrations in the tissues of biological organisms.

4.4.2 Chlorophenolic antimicrobials

The characteristics of TCS and TCC significantly influence their removal efficiency during WWT processes. Kwon and Xia (2012) indicate that, owing to their high log K_{ow}, up to 90% of TCS and TCC entering WWTPs are adsorbed onto the sludge during the treatment processes. In a simulation study, it was shown that variations in hydraulic retention time, sludge retention time (SRT), and temperature influence the removal and transformation of TCS and TCC throughout WWT (Armstrong et al., 2018). Apparently, both antimicrobials tend to adsorb onto sludge, resulting in minimal concentrations remaining in the effluent. Furthermore, their degradation rates increased with elevated temperatures, with TCS exhibiting a more rapid degradation than TCC. To learn more about the destiny of these two antimicrobials, a second WWT simulation was performed under nitrifying conditions. The findings indicated a rapid degradation of TCS into methyl triclosan (MeTCS), which is becoming increasingly pronounced in alkaline environments (Armstrong et al., 2019). Conversely, TCC exhibited significant resistance to this treatment. The emergence of MeTCS, a degradation product of TCS with a log K_{ow} value of 5.2 (Coogan et al., 2007), suggests heightened concern due to its greater potential for bioaccumulation compared to the parent compound. Alongside WWT transformation products like carbanilides, MeTCS is thought to retain endocrine disruption capabilities (Ahn et al., 2008), raising additional ecological concerns.

Li et al. (2020) discussed the stability of TCS under typical environmental conditions but highlighted the possibility of photodegradation when exposed to ultraviolet light. In contrast, TCC is acknowledged for its lasting presence in the environment; it demonstrates resistance to degradation processes and has the capacity to accumulate in aquatic and terrestrial ecosystems following release via WWT operations (Carey and McNamara, 2014; Armstrong et al., 2019; Abril et al., 2018). As specified above, the presence of TCS and TCC was below the quantitation limits. This phenomenon may be attributed to their degradation potential, as postulated by Lozano et al. (2010), who observed a substantial reduction in TCS levels in agricultural soils 16 months following the application of biosolids. In a similar context, Anand et al. (2022) highlighted the biotransformation of parent CECs, as opposed to their elimination or biodegradation, as a plausible mechanism for elevating the concentration of total contaminants in effluent relative to the influent. Degradation byproducts of antimicrobials were not measured in the consulted studies. Lastly, the presence of parabens, TCS, and TCC has been determined in urine (Shin et al., 2019), human serum (Li et al., 2020; Assens et al., 2019), and in adipose tissue (Artacho-Cordón et al., 2018), highlighting their bioaccumulation potential. Pérez et al. (2023) demonstrated an elevated bioaccumulation of TCS in bulrush (*T. latifolia*) roots. In a separate study by Cavanagh et al. (2018), TCS demonstrated considerable potential for endocrine disruption, notably through its binding affinity to the transthyretin (TTR) receptor. It was also highlighted for its anti-androgenic properties, suggesting a capacity to interfere with normal developmental processes and male reproductive health.

5 Hormones

5.1 Overview

Natural and synthetic hormones are increasingly recognized as CECs due to their potential impacts on environmental and human health. Natural hormones, such as estrogens and androgens, are excreted by humans and animals and can enter water systems through wastewater (Adeel et al., 2017; Naldi et al., 2016). Additionally, synthetic hormones, such as those utilized in contraceptives, hormone replacement therapies (HRT) (Viglino et al., 2008; Koubovec et al., 2005), anabolic agents for muscle growth and strength enhancement (Liu and Wu, 2019; Bond et al., 2022), cancer therapy (Johnston and Cheung, 2018; Sartor and Gillessen, 2014), and various agricultural applications (Xuan et al., 2008), further contribute to environmental contamination. Hormones are biologically active at very low concentrations and can disrupt endocrine systems in wildlife, leading to reproductive and developmental abnormalities (Diamanti-Kandarakis et al., 2009). For example, exposure to estrogen hormones in aquatic environments has been linked to feminization of male fish (Jobling et al., 2006). Thus, hormones are a growing focus of environmental monitoring and regulatory efforts to mitigate their ecological and health impacts.

5.2 Physical and chemical characteristics: indices of fate and transport

Based on their chemical structure, hormones are threefold: lipid-derived hormones (or lipid-soluble hormones), amino acid-derived hormones, and peptide hormones. Most lipid-soluble hormones are

derived from cholesterol and exhibit structural similarities. Steroid hormones are the main class of lipid hormones in humans. Steroid hormones estradiol (estrogen) and testosterone (androgen), which are associated with female and male sexual traits, respectively, are in this class. Chemically, they are categorized as ketones or alcohols. The amino acid-derived hormones are made of tryptophan and tyrosine amino acids, which are comparatively small molecules. Hormones in this group include thyroxine, which is produced by the thyroid gland, and adrenal gland medulla-made norepinephrine and epinephrine. In contrast, peptide hormones are characterized by their polypeptide chain structure, which consists of amino acids. They include oxytocin and antidiuretic hormones, which are synthesized in the brain and released into the bloodstream by the posterior pituitary gland. This category also encompasses smaller proteins, such as growth hormones produced by the pituitary gland, as well as larger glycoproteins like follicle-stimulating hormones (Fowler et al., 2018).

5.3 Occurrence in sewage sludge, biosolids, and soils

Research has demonstrated the occurrence of hormones in various environmental matrices, including soil (Kumirska et al., 2019), wastewater (Gewurtz et al., 2022) and biosolids (Lorenzen et al., 2004). Here, we analyzed studies focusing on hormonal compounds present in sewage sludge (Silva et al., 2021; Riva et al., 2021; Moško et al., 2021; Svahn and Björklund, 2019), biosolids (Gewurtz et al., 2022), and soil (Kumirska et al., 2019). The hormones under investigation include eleven naturally occurring hormones: androstenedione (A4), androsterone (AN), cortisone (E), estrone (E1), 17 β-estradiol (E2), 17 α-estradiol (17 α-E2), estriol (E3), progesterone (P), testosterone (T), equilin (EQL), and equilenin (EQN); as well as ten synthetic hormones: 17 α-ethinylestradiol (EE2), allyl trenbolone or altrenogest (ALT), desogestrel (DSG), diethylstilbestrol (DES), gestodene (GST), melengestrol acetate (MGA), mestranol (EEME), norethindrone (NRT), norgestrel (NRG), 17 α-dihydroequilin (2H-EQL). The quantification of four hormones in sewage sludge reveals a predominance of GST, with a concentration of 41 ng/g, followed by E1 at 17 ng/g, and E2 at 16 ng/g. In biosolids, the hormonal composition is primarily characterized by synthetic hormones, specifically DSG at 252 ng/g and EEME at 176 ng/g, while natural hormones are represented by E3 at 95 ng/g, AN at 54 ng/g, and P at 28.5 ng/g. Out of five hormones investigated in soils by Kumirska et al. (2019), E2, EE2, and DES appeared below the quantitation limits, while E1 and E3 showed respective median concentrations of 6.3 ng/g and 1.9 ng/g. Supplementary Table S3 gives detailed information on the occurrence of hormonal compounds in examined matrices.

5.4 Environmental fate and transport mechanisms

According to Limpiyakorn et al. (2009), peptide and steroid hormones represent significant environmental concerns. Hanselman et al. (2003) assert that endogenous hormones may exhibit extraordinary potency in disrupting the endocrine systems of organisms, potentially exceeding the effects of exogenous hormones by a factor ranging from 10,000 to 100,000. Various factors, including soil characteristics, organic matter content, pH levels, and microbial activity, can influence the sorption capacity of hormones to solid particles within WWT systems, thereby affecting their transport dynamics and potential removal (Ying and Kookana, 2005). The fate of these hormones is additionally modulated by residence time and temperature within WWT systems (Viglino et al., 2008). Numerous hormones, including both naturally occurring and synthetic variants, are not entirely eliminated during conventional WWT processes (Bai and Acharya, 2019), permitting their entry into adjacent water bodies and groundwater sources (Zhou et al., 2012). To cite an example, Huang et al. (2014) evaluated the removal efficacy of one progestogen (P), three androgens (A4, T, and dihydrotestosterone (DHT)), and four estrogens (E1, E2, E3, and EE2) throughout the WWT process. Their findings indicated a remarkable removal efficiency for androgens and progestogens, ranging from 86% to 100%. In contrast, the removal efficiency observed for estrogens was comparatively lower, falling within the 75%–92% range.

It is established that natural hormones are excreted by living organisms in conjugated forms that remain undetectable by analytical methods designed for unbound hormones. Given that conjugation can be reversed by naturally occurring enzymes present in WWTPs and the environment, the ecological fate and migration mechanisms of these hormones are notably affected by both conjugation/deconjugation processes as well as the specific wastewater treatment methodologies employed (Gewurtz et al., 2022). Deconjugation of some of the hormones excreted in urine seems to occur within a time frame of hours to days, within the period when the WWT process may have a significant impact on deconjugation (Naldi et al., 2016). Hormones demonstrate persistence against various treatment approaches (e.g., sedimentation, biological processing, and disinfection), leading to their detection in effluents released into ecosystems. This situation raises concerns over EDCs like E2, which can pose substantial ecological risks, including fish feminization and disruption of aquatic ecosystems, even at concentrations as minimal as 2 ng/L (Bai and Acharya, 2019). However, when wastewater undergoes treatment through techniques such as ultrafiltration (UF) or ozonation, research has indicated varying degrees of efficacy regarding hormone removal. For instance, the algae-mediated degradation exhibited a notable removal efficiency of approximately 60% for certain hormones like E2 and EE2. Gewurtz's study (2022) explained that WWT techniques such as lagoon systems alongside secondary and advanced treatment facilities could initiate hormonal deconjugation processes impacting concentrations of hormones like EEME and T. Furthermore, this investigation highlights that upon discharge into aquatic environments or application of treated biosolids on agricultural lands, these conjugated hormonal forms may subsequently undergo deconjugation in the natural environment. Such processes could enhance hormonal bioavailability, allowing for increased interactions with aquatic ecosystems and soils, which might affect terrestrial flora and fauna uptake.

The transport dynamics of hormones within soil matrices are contingent upon their solubility profiles, adsorption affinities to soil

particles, as well as uptake mechanisms involving plants or microorganisms (Bai and Acharya, 2019). The deconjugated form of these hormones exhibits lipophilicity, whereas their conjugated counterparts are hydrophilic (Gewurtz et al., 2022); this distinction illustrates how hormonal compounds may leach into groundwater or adhere to soil particles. Hormones may be either adsorbed onto particulate matter or dissolved within soil solution. Extended retention times in soils can allow for the degradation of certain compounds. Still, retention on soil surfaces would reduce the solution concentration of hormones and slow down their degradation. Furthermore, hormones may also migrate through leaching or surface runoff pathways, leading to potential groundwater contamination or infiltration into surface water bodies. In environmental contexts, plant systems can absorb hormones, potentially influencing physiological functions such as root development, shoot growth patterns, flowering events, and germination processes (Shi et al., 2010). Moreover, hormonal compounds have been documented to bioaccumulate in species such as quagga mussels (*Dreissena bugensis*), indicating their propensity to infiltrate food webs (Bai and Acharya, 2019), a phenomenon posing considerable risks to ecological health. Microbial transformations also play a crucial role in facilitating hormone degradation or transformation, including analogous deconjugation phenomena observed within WWTPs (Gewurtz et al., 2022; Olsen et al., 2007). Czajka and Londry (2006) highlighted the susceptibility of hormones to microbial degradation within soil environments depending on specific bacterial populations and other variables such as moisture levels and soil composition types; they further documented hormones' anaerobic transformation capabilities under certain conditions within soil matrices. Bartelt-Hunt et al. (2012) emphasized concerns associated with runoff occurrences in regions characterized by intensive animal husbandry practices where manure applications could directly introduce hormones into local ecosystems.

6 Plastic-related compounds

Bisphenols (BPs) and phthalic acid esters (PAEs) are two classes of plastic-related compounds under scrutiny. The literature examined documented the presence of twenty BP analogs and fifteen PAEs within the soil and sewage sludge matrices.

6.1 Overview

6.1.1 Bisphenols

Phenolic plastic-related compounds, such as bisphenol A (BPA), are synthetic compounds employed to enhance the flexibility and durability of plastics (Kubwabo et al., 2009; NIEHS, 2023). These substances are common in polycarbonate plastics and epoxy resins, utilized across various consumer products, including water bottles, food packaging, and medical devices (EFSA, 2023; Park et al., 2018). Research has raised concerns about the potential health repercussions of bisphenols, particularly BPA. This compound has been implicated in endocrine disruption, reproductive disorders, and an elevated risk of certain cancers (vom Saal et al., 2007; Lang et al., 2008). Furthermore, studies have demonstrated that BPA can leach from plastic materials into consumables, thereby facilitating human exposure (FDA, 2023). In light of these apprehensions, several states have instituted bans on using BPA in specific products such as baby bottles and sippy cups (EC, 2011a; EC, 2011b). Concurrently, various manufacturers have proactively substituted BPA with alternative bisphenols like bisphenol S (BPS) or bisphenol F (BPF). Nevertheless, research has also raised alarms about the potential health implications associated with these alternative bisphenols. A recent study correlated obesity in young people to exposure to BPS and BPF (Jacobson et al., 2019). Another study examined the endocrine disruption potential of six commercially available BPA alternatives: BPS, BPF, bisphenol AP (BPAP), bisphenol AF (BPAF), bisphenol Z (BPZ), and bisphenol B (BPB). All 6 BPA substitutes were found to be no less estrogenic than BPA in human breast cancer cells, with BPAF, BPB, and BPZ being even more estrogenic (Mesnage et al., 2017). As a consequence, experts are advocating for minimizing exposure to all bisphenols while promoting safer alternatives (Rochester and Bolden, 2015; Viñas and Watson, 2013).

6.1.2 Phthalic acid esters

PAEs, called phthalates, are synthetic compounds widely employed as plasticizers across various consumer products. These include flexible polyvinyl chloride (PVC) items such as vinyl flooring, wallpaper, and apparel; personal care items encompassing cosmetics, fragrances, and lotions; medical apparatus including tubing, catheters, and gloves; as well as food packaging and processing equipment (Wang and Qian, 2021; Lee et al., 2019; Net et al., 2015; Li et al., 2017). The mechanism by which phthalates function involves enhancing the flexibility and durability of plastics, thereby rendering them more malleable and resistant to fracturing. Nonetheless, these substances are not chemically integrated within the plastic, which renders them susceptible to leaching into the environment and potentially into foodstuffs and human organisms via ingestion, inhalation, or dermal contact matrix (Wang et al., 2022). Such leaching may result in human exposure that raises significant health concerns attributed to their potential endocrine-disrupting characteristics, which can interfere with human hormonal systems (Zhang et al., 2021a; Zhou et al., 2019). Empirical studies have established associations between phthalate exposure and adverse reproductive outcomes, congenital anomalies, as well as developmental disorders (Swan Shanna et al., 2005). For this reason, regulatory entities such as the European Chemicals Agency (ECHA) and the USEPA have prohibited or subjected PAEs to restrictions (ECHA, 2022; USEPA, 2023a).

6.2 Physical and chemical characteristics: indices of fate and transport

6.2.1 Bisphenols

The chemical structure of BPs consists of two phenolic groups connected by a bridging group (such as carbon or sulfur) (Sánchez-Piñero et al., 2020; Xu et al., 2021a). This structure contributes to their chemical reactivity and ability to form polymers, which accounts for their extensive application as plastic-related

compounds (Xu et al., 2021a). BPs tend to be soluble in organic solvents but exhibit limited solubility in water (Zhu et al., 2019; Sun et al., 2018; Huang et al., 2020). This hydrophobic behavior affects their distribution in sludge and increases their potential for adsorption onto particulate matter (Zhu et al., 2019).

6.2.2 Phthalic acid esters

PAEs constitute a group of industrial compounds distinguished by their chemical composition comprising two alkyl or aryl groups bonded to a phthalate group. Their unique physicochemical characteristics render them suitable for diverse applications. Predominantly employed as plasticizers or additives in consumer goods, PAEs can be classified based on their molecular weight. Higher molecular weight phthalates are typically utilized as plastic additives to enhance the pliability and durability of polyvinyl chloride (PVC) products, exemplified by DEHP. Lower molecular weight phthalates fulfill various functions (e.g., solvents, emulsifiers, stabilizers) in consumer items like cosmetics, personal care products (PCPs), insecticides, adhesives, pharmaceuticals, and solvents - examples being diethyl phthalate (DEP), di-n-butyl phthalate (DnBP), and dimethyl phthalate (DMP) (Koniecki et al., 2011; Tao et al., 2023).

6.3 Occurrence in sewage sludge, and soils

6.3.1 Bisphenols

Publications on twenty BPs in sewage sludge and the soil were examined. BPA and BPF are the most dominant sewage sludge among BP analogs, with 179 ng/g and 165 ng/g median concentrations, respectively. BPF (3.4 ng/g) and BPA (2.3 ng/g) dominate again in the soil. The median total concentration of twenty bisphenol analogs in sewage is 388 ng/g, and 9.6 ng/g in the soil. Detailed data are found in Supplementary Table S4.

6.3.2 Phthalic acid esters

Fifteen PAEs were analyzed in sewage sludge or soil by several authors. PAEs represent the most dominant class of CECs occurring in sewage sludge and soil compared to all other classes reviewed in this paper. In fact, the median cumulative concentration of 15 PAEs in sewage sludge is 734 μg/g, with minimum and maximum values ranging from 107 μg/g to 2,618 μg/g. The concentrations of PAEs are consistently elevated in sewage sludge when compared to those found in soils. The median concentration of PAEs measured in the soil is 2.99 μg/g, with the lower and upper optima of 0.28 μg/g and 634 μg/g, respectively. DEHP registered the highest median concentration in sewage (224 μg/g), followed by BBzP (202 μg/g), and dibutyl phthalate (DBP) (182 μg/g). In the soil, DEHP dominates (1.73 μg/g), followed by DiBP (0.53 μg/g), and DBP (0.29 μg/g). Additional information is in Table S6 and Supplementary Appendix B-Sheet 7-Phthalates.

6.4 Environmental fate and transport mechanisms

6.4.1 Bisphenols

Understanding the environmental behaviors and mobility of BPs is essential for assessing their environmental impact and developing strategies to mitigate their adverse effects. The fate of BPs in the WWT process, sewage sludge, and soil is influenced by a complex interplay of factors, and it varies depending on various factors such as their physicochemical properties, environmental conditions, and treatment methods. Studies have shown that BPs are not completely removed during conventional WWT, leading to their presence in both sewage sludge and liquid effluents (Huang et al., 2020). BPs may partition into the water phase, sorb onto solid fractions during WWT, and accumulate in sewage sludge (Xu et al., 2021a; Peng et al., 2020). During WWT and in soil, BPs undergo various transformations, influencing their environmental fate and transportation mechanisms (Sun et al., 2018; Zhu et al., 2019; Sánchez-Piñero et al., 2020). Xue and Kannan (2019) have noted that BPs are not completely removed during the activated sludge treatment process; a substantial portion becomes bound to solids within the sewage sludge during treatment, which subsequently facilitates their transportation to land when this sludge is utilized as fertilizer in agricultural practices (Zhu et al., 2019).

Considerable disparities exist in the removal efficiencies of BPs, especially BPA, across different WWT technologies. Wang et al. (2019) illustrate that primary treatment exhibits the lowest BPA removal rate, averaging approximately 25.4%, followed by lagoon treatment with a moderate removal efficiency averaging 58.3%. Biologically aerated filters demonstrate enhanced removal efficiency with an average rate of around 66.0%. Finally, the activated sludge process yields superior performance with an average removal rate of 74.7%. Additionally, the Oxidation Ditch (OD) process marginally outperforms modified activated sludge processes, such as Anaerobic-Anoxic-Oxic (AAO) and Anoxic-Oxic (AO) methods), which exhibit slightly lower removal efficiencies compared to standard activated sludge.

Upon application of sewage sludge as fertilizer or soil amendment, BPs can be released into the soil environment. Studies have identified BPs in sewage sludge (Peng et al., 2020; Sun et al., 2018; Zhu et al., 2019) and agricultural soils (Xu et al., 2021a; Sánchez-Piñero et al., 2020). The accumulation of BPs in soil from biosolids raises concerns about their persistence, estrogenic potency, and potential risks associated with land application. Several factors influence the fate of BPs in biosolids following their introduction into soil environments. Weaker binding may facilitate leaching into groundwater systems; conversely, strong adsorption can inhibit such occurrences. Exposure to environmental factors within soils can prompt further degradation of BPs through microbial action or photodegradation processes. Factors such as temperature, moisture content, and soil type can all influence degradation rates (Zhu et al., 2019; Sánchez-Piñero et al., 2020). Generally, BPs are chemically stable under certain conditions but may undergo hydrolysis or degradation under extreme pH, temperature, or oxidative conditions, impacting their persistence in the environment and potential biological effects (Zhu et al., 2019; Xu et al., 2021a). A study conducted in Scotland and the United Kingdom showed that 4% by mass of BPA in the soil was predicted to enter soil pore water, resulting in significant uptake by crops, primarily leafy vegetables, but also root crops, and much lower uptake into cereal grains (Zhang et al., 2015).

6.4.2 Phthalic acid esters

U.S. EPA's list of priority pollutants outlines six PAEs including the two widely used ones, DBP and DEHP (Wang

et al., 2022; USEPA, 2023b). The primary sources of PAEs in soils comprise the application of biosolids, sewage irrigation, and chemical fertilizers and additives (Wei et al., 2020; Becky Miriyam et al., 2022). The transport pathways and environmental destiny of PAEs within soil and air matrices are influenced by a confluence of factors, including the origins of pollution, their respective properties, the inherent properties of the soil, and its associated microbiological characteristics (Tuan Tran et al., 2022; Koniecki et al., 2011; Wang et al., 2022). Certain PAEs, particularly those of lower molecular weight, possess heightened volatility (e.g., DEHP Henry's law volatility constant value of 4.37×10^{-5} atm-m^3/mol) potentially resulting in their release into the atmosphere and subsequent exposure via inhalation or dermal contact (Koniecki et al., 2011; Howard, 2017).

Wang et al. (2022) studied the impact of soil-particle size on the migration of PAEs in soil medium. They found that total PAE concentrations in bulk soil decreased as samples were collected farther from point-source pollution, in this example, the landfill. Clay-sorbed PAEs (clay particle size < 2 μm) were found to be consistent with the Gaussian air pollution model, suggesting atmospheric transportation. Silt-sorbed (2 μm < particle size < 63 μm) PAEs showed a leveled-off concentration, suggesting shorter transport distance due to gravitational deposition, explained by the Boltzmann equation. Sand-sorbed PAEs (particle size > 63 μm) showed an unexpected increasing trend in surrounding soils, emphasizing the apparent accumulation of PAEs at downhill sites. When PAEs interact with trace elements in soils, their level of toxicity is raised (Zhang et al., 2021b). Berenstein et al. (2022) found that phthalic acid and monobutyl phthalate (MBP) are generated by the photodegradation of DBP in polyethylene mulches. The same research report that the migration of DBP and MBP from these plastic covers into the surrounding environment constitutes a notable concern. Another research by Zeng et al. (2008) pointed out that the PAEs' metabolites can exhibit toxicity levels similar to those of their parent compounds. It has been observed that there exists a positive correlation between the lipid content of vegetables and the bioconcentration factor (BCF) of PAEs (Wei et al., 2020). Furthermore, it is unlikely that complete mineralization of PAEs will occur under *in situ* conditions within biosolids-amended soils (Madsen et al., 1999). The airborne migration of PAEs significantly elevates human exposure levels. The dermal absorption rates are contingent upon the specific types of phthalates, with DEP demonstrating particularly high absorption potential. This underscores its likelihood of contributing to cutaneous exposure through cosmetic and personal care products. Extensive research has established a correlation between PAEs and various health concerns, especially regarding their capacity to disrupt endocrine functions as well as adversely affecting reproductive and developmental processes, and potentially pose carcinogenic risk (Wei et al., 2020; Koniecki et al., 2011; Kaewlaoyoong et al., 2018; Yoshida et al., 2020). Given their ubiquitous presence in the environment, PAEs raise critical concerns about chronic exposure within the general population, particularly among vulnerable demographics such as infants and toddlers (Koniecki et al., 2011).

7 Rubber and polymer antioxidants [Aromatic secondary amines (Ar-SAs)]

7.1 Overview

Two main categories of aromatic secondary amines (Ar-SAs) are commonly utilized as antioxidants in rubber and polymers: substituted diphenylamines (S-DPAs) and substituted p-phenylenediamines (S-PPDs) (Kassler et al., 2014; Lu et al., 2016; Liu et al., 2019).

7.1.1 Substituted diphenylamines

S-DPAs form a group of secondary amine antioxidants that are widely employed in petroleum-based products (fuels, lubricants), rubber materials, plastics, and other polymeric substances to prevent oxidative deterioration (Kassler et al., 2014). The diphenylamine (DPA) central structure is the basis and the lone functional group all S-DPAs share. The DPA's phenyl rings can have one to four side substitutions that are either phenyl or saturated alkyl, with the alkyl side chains containing four to nine carbons (OECD, 2016). In addition to the PREPOD (2-Propanone, reaction products with diphenylamine, CAS RN 68412-48-6) group investigated by Zhang et al. (2020c), three other remarkable classes of S-DPA antioxidants can be recognized as alkylated (mono-, di-, or poly-) DPAs [e.g., BNT (Benzenamine, N-phenyl-, reaction product with 2,4,4-trimethylpentene), CAS RN 68411-46-1], S-DPAs of variable numbers of phenyl/benzyl replacements on DPA rings [e.g., BNS (Benzenamine, N-phenyl-, styrenated), CAS RN 68442-68-2], and S-DPAs with changing numbers of alkyl/benzyl and phenyl substitutions [e.g., BNST (Benzenamine, N-phenyl-, reaction products with styrene and 2,4,4 trimethylpentene), CAS RN 68921-45-9] (Health Canada, 2010). These classes are mainly UVCB (Unknown or Variable Composition, Complex Reaction Products, and Biological Materials) chemical substances.

7.1.2 Substituted p-phenylenediamines

S-PPDs, on the other hand, are added to the rubber as antioxidants, anti-ozonolysis, and bending crack inhibitors (Huntink et al., 2004). Huang et al. (2021) mentions three forms of PPDs utilized in the rubber processing industry, which are the dialkylated PPD (N,N′-dialkyl-p-phenylenediamine), the mono-alkylated-mono-arylated PPD (N-alkyl-N′-aryl-p-phenylenediamine), and the diarylated PPD (N,N′-diaryl-p-phenylenediamine). The UVCB class of BENPAT (1,4-Benzenediamine, N,N′-mixed phenyl and tolyl derivatives, CAS RN 68953-84-4) studied by Zhang et al. (2020c) and captured on Canada's List of all Challenge chemical substances (Health Canada, 2010), alongside BENTAX (1,4-Benzenediamine, N,N′-mixed tolyl and xylyl derivatives, CAS RN 68478-45-5) exemplify S-PPDs.

7.2 Physical and chemical characteristics

7.2.1 Substituted diphenylamines

According to an evaluation report of S-DPAs released by the Canadian federal government in 2017, empirical and modeled determinations of the S-DPAs' physicochemical characteristics displayed high log K_{ow} values and poor water solubility (ECCC,

2017). For different S-DPAs, the anticipated log K_{ow} values range from 4.45 to 13.58; however, the experimental measurement of log K_{ow} values beyond 8.2 was uncertain because it is difficult to accurately determine the partitioning properties of superhydrophobic substances. The conjugate acid's acid dissociation constant (pK_aH) of S-DPAs that are of 0.8 ± 0.4 indicates that, in an aqueous environment, the neutral form will predominate (ECCC, 2017). Overall, the physicochemical attributes and kinetics of S-DPAs are reported to be influenced by the degree of substitution and the number of carbon atoms in the side chains (OECD, 2016). S-DPAs typically have higher molecular weights than S-PPDs, affecting their volatility and solubility (Liu et al., 2019).

7.2.2 Substituted p-phenylenediamines

The characteristics of S-PPDs, particularly in their roles as antioxidants and antiozonants, exhibit considerable variability based on the substitutions present on the nitrogen atoms. These diverse substitutions significantly impact their solubility, volatility, and reactivity profiles. S-PPDs demonstrate a range of solubility levels in aqueous environments. They are relatively stable under normal conditions but can undergo oxidation, especially when subjected to ozone or UV radiation, leading to the formation of secondary products like N-(1.3-dimethylbutyl)-N′-phenyl-p-phenylenediamine (6PPD) (Huang et al., 2021). The polarity of these compounds is influenced by the presence of functional groups such as hydroxyl and amine moieties or other polar substituents that enhance their polarity, thereby affecting their behaviors in the environment (Moschet et al., 2018; Huang et al., 2021). The chemical structure of S-DPPs determines their stability. Bulky substituents on the aromatic ring can increase stability and decrease volatility. The essential component of these antioxidants' activity is their reactivity with oxidizing agents like peroxy radicals, produced when rubber ages. They can scavenge radicals, thus effectively terminating oxidative chain reactions (Huntink et al., 2004). The widespread use of these antioxidants in tire formulations and their resultant environmental discharge highlight their persistence and potential accumulation (Tian et al., 2021; Zhang et al., 2021c).

7.3 Occurrence in sewage sludge, biosolids, and soils

7.3.1 Substituted diphenylamines

Fourteen S-DPAs, including six PREPODs and eight BNSTs, have been characterized in biosolids by Zhang et al. (2020c), Zhang et al. (2021c). The central tendency of the most occurring S-DPAs in biosolids is 86 and 66 ng/g for TO-DPA and DTOS DPA, respectively, while the central tendency for the least prevalent is 3.8 and 4.9 ng/g for IPDM-AD and IP-DPA, respectively. Overall, BNST compounds are the most recurring compared with the PREPODs, with a median concentration of 408 ng/g for the former and 43 ng/g for the latter. The maximum concentration of S-DPAs can achieve 3,133 ng/g of dry biosolids. Supplementary Table S8 gives information about the concentrations of S-DPAs quantified in indoor and playground dust. The most common among measured S-DPAs are di-n-octyl-DPA and 4,4′-bis(1,1-dimethylbenzyl)diphenylamine (diAMS), with median concentrations of 74 and 46 ng/g in dust. The cumulative median concentration of the six measured S-DPAs was 215 ng/g, while the maximum concentration reached 13,619 ng/g. Supplementary Tables S6, S7 present data on the occurrence of S-DPAs in biosolids and dusts.

7.3.2 Substituted p-phenylenediamines

This review investigates thirteen S-PPDs in biosolids or dust matrices, and respective occurrences are reported in Supplementary Tables S8, S9, respectively. N,N′-diphenyl-p-phenylenediamine (DPPD), N-phenyl-N′-(o-tolyl)-p-phenylenediamine (PTPD), and N,N′-di (o-tolyl)-p-phenylenediamine (DTPD) are three S-PPDs analyzed in biosolids totaling median concentration of 45 ng/g. Their concentration range varies from 4.5 to 366 ng/g. Twelve S-PPDs analyzed in dust showed a median concentration of 233 ng/g, with min-max spanning from 85 to 1,735 ng/g. The occurrence of S-PPDs in dust is dominated by N,N′-di (o-tolyl)-p-phenylenediamine-quinone (DTPD-Q), N-(1,3-dimethylbutyl)-N′-phenyl-p-phenylenediamine-quinone (6PPD-Q), and 6PPD, with respective concentrations of 60 ng/g, 42 ng/g, and 42 ng/g. The occurrence of DPPD in biosolids is to a certain extent similar as that found in dusts (15 ng/g in biosolids vs. 12 ng/g) and points out that there are several ways of exposure to this class of contaminants. Detailed statistical calculus is given in Supplementary Appendix B-Sheet 9-S-PPDs.

Research indicates that average concentrations of S-PPDs are significantly elevated in rubber playground dust in comparison to indoor dust, suggesting a heightened level of exposure in outdoor environments (Liu et al., 2019). Furthermore, another investigation revealed that S-PPDs, including 6PPD and its transformation product 6PPD-Q, are commonly found in dust samples collected from diverse settings, with the highest concentrations identified in road and vehicle dust (Huang et al., 2021). A study conducted in Tokyo, Japan, illustrated that the levels of 6PPD-Q were more pronounced in dust samples obtained from high-traffic areas than those from low-traffic regions, thereby demonstrating a significant influence of traffic patterns. Additionally, another study noted that concentrations of 6PPD-Q reached their zenith during months characterized by elevated atmospheric ozone levels (Hiki and Yamamoto, 2022).

7.4 Environmental fate and transport mechanisms

7.4.1 Substituted diphenylamines

S-DPAs have been found in biosolids at amounts ranging from hundreds to over 1,000 ng/g dw in biosolid samples (Zhang et al., 2020c). In their studies, Zhang et al. (2020c), Zhang et al. (2021) successfully reported seventeen S-DPA in various compartments, including biosolids. The prevalence of S-DPAs in WWTP influent was positively correlated with the sizes of the served populations, indicating that the amount of waste produced significantly influences the concentration of S-DPAs in the treatment system (Lu et al., 2017). Sorption to sludge was identified as the primary mechanism for S-DPAs removal within WWTPs, with longer retention times in treatment units facilitating more efficient

partitioning of S-DPAs to solid sludge. Despite relatively high removal efficiencies, notable concentrations of S-DPAs were still detected in effluents (Zhang et al., 2020b; Lu et al., 2017). The biological treatment phases within the anoxic/oxic (A/O) process and cyclic activated sludge technology (CAST) have exhibited a higher efficiency in removing most S-DPAs than the hydrolysis acidification tank. This initial chemical-assisted treatment showed lower efficacy, likely due to biological processes enabling longer retention times, enhanced interaction between bacteria and pollutants, and the superior ability of microorganisms to degrade S-DPAs compared to purely chemical approaches (Zhang et al., 2020b). The research has also determined that meteorological elements, such as temperature and sun radiation, can influence the microbial populations' behavior and biodegradation mechanisms, consequently impacting the adsorption of S-DPAs on the sludge fraction.

Higher loads in benthic invertebrates (crayfish, *Orcoescties* spp.) than in pelagic fish (hornyhead chub, *Nocomis biguttatus*) and common shiners (*Luxilus cornutus*) indicate that the partitioning of S-DPAs in sediments was higher than in water (Lu et al., 2016). Likewise, concentrations of S-DPAs in biosolids surpassed those found in effluents, confirming that these compounds potentially accumulate and persist in the solid wastes generated during WWT (Zhang et al., 2016; Lu et al., 2017). This raises concerns about their long-term environmental impacts if these solids are used for land application or disposal. Given the substantial levels of S-DPAs persisting in biosolids, there are apprehensions regarding their beneficial reuse, particularly concerning practices like composting or land application. The presence of S-DPAs may result in their integration into terrestrial ecosystems and subsequent amplification throughout the food chain. This was evidenced by the detection of S-DPAs in the livers and eggs of seabirds, as well as in the livers of seals, indicating pollution within the Arctic ecosystem by this particular group of organic contaminants (Lu et al., 2019). Lastly, compared to seal livers, seabird livers, more particularly, the livers of the northern fulmar and black-legged kittiwakes, showed noticeably higher levels of S-DPAs, suggesting that these seabirds may be more vulnerable to exposure due to their migratory nature. Also, compared to seabirds, which have established migratory patterns, ringed seals showed greater individual variability in their migration patterns, which may affect the quantities and compositions of contaminants in their tissues (Lu et al., 2019).

7.4.2 Substituted p-phenylenediamines

There is documented evidence that dust is an important sink for S-DPAs and S-PPDs in both indoor and outdoor settings (Hwang et al., 2016; Moschet et al., 2018), highlighting grime as a human exposure pathway through oral ingestion, skin exposure, and breathing (Mercier et al., 2011). S-PPD antioxidants are acknowledged as the most commonly encountered allergens among various rubber chemicals (Mahler, 2021). The recent revelation of the aquatic toxicity of 6PPD has been substantiated by its demonstrated harmful effects on fathead minnows (*Pimephales promelas*) (Prosser et al., 2017), along with the confirmed lethality of its oxidation byproducts, 6PPD-Q, towards coho salmon (*Oncorhynchus kisutch*) (Tian et al., 2021). Having said that, their environmental fate and transport mechanism are still unclear (Huang et al., 2021).

8 Pesticides

8.1 Overview

Here, literature pertaining to the presence of neonicotinoid insecticides (NEONICs) was reviewed. NEONICs are neuro-active and systemic insecticides that bear a structural resemblance to nicotine. Their widespread application in agriculture dates back to the early 1990s (Kollmeyer et al., 1999; Akter et al., 2023). NEONICs specifically target the sodium/potassium ionophore within an insect's brain-spinal cord axis, disrupting cholinergic neural signaling by acting as agonists on nicotinic acetylcholine receptors (nAChRs). Upon binding tightly to these receptors in insects' central nervous apparatus, NEONICs can result in receptor blockage, paralysis, or fatality at higher concentrations while inducing neurological activation at lower levels. Notably, NEONICs exhibit a higher affinity for binding with insects' nAChRs than mammalians, rendering them preferentially more toxic to insect populations (Tomizawa and Casida, 2005). Agricultural practitioners and horticulturists highly favor NEONICs for several key reasons. They exhibit a wide-ranging efficacy against various insect pests, can be conveniently administered through various methods (seed treatments, soil drenches, or foliar applications), possess minimal acute toxicity towards mammals, and demonstrate systemic properties that enable plant uptake and distribution throughout plant tissues. This systemic nature provides safeguarding capabilities against pests that target different components of plants (Akter et al., 2023).

8.2 Physicochemical and biological characteristics

NEONICs exhibit a low n-octanol-water partition coefficient (log K_{ow} < 1), an acid dissociation constant that lies considerably below or above the environmental pH ($pK_a \leq 2.2$ and $pK_a \geq 11$), and a half-life (DT_{50}) that spans from days to years (1–1,155 days). Supplementary Table S11 gives the log K_{ow}, pK_a, and half-life (DT_{50}) values of NEONICs and the associated phenyl-pyrazole fipronil. Several authors have employed these physicochemical properties to explain the environmental fate and transportation mechanisms of NEONICs in soil and water media (Simon-Delso et al., 2015; Hladik and Kolpin, 2015). The N-nitroguanidines (clothianidin, dinotefuran, imidacloprid, and thiamethoxam), nitromethylenes (nitenpyram), and N-cyanoamidines (acetamiprid and thiacloprid) are the three chemical categories into which they can be placed (Jeschke et al., 2011).

8.3 Occurrence in sewage sludge, biosolids, and soils

Using a business espresso maker and a pressurized hot water extraction (PHWE) inox steel column, Svahn and Björklund (2019) examined the extraction of imidacloprid (IMI) alongside various pharmaceutical products (PPs) from biosolids. The findings indicated that the two extraction methods resulted in concentrations of 1.4 ng/g d. w. and 3.3 ng/g d. w. for IMI,

respectively. The presence of NEONICs was documented in various soil types and uses in China by Zhang et al. (2020a), Ying et al. (2022), Zhou et al. (2018), Zhou et al. (2021), and in Belize and the Philippines by Bonmatin et al. (2019), Bonmatin et al. (2021). Supplementary Table S12 summarizes statistics on quantified NOEs in the soil. A comprehensive review of eight NOEs in the soil compartment reveals key statistical measures. IMI demonstrates a higher median concentration of 2.6 ng/g, followed by acetamiprid (ACE) and nitenpyram (NIT) with 2.6 ng/g and 2.5 ng/g, respectively. Imidaclothiz (IMT) was found to be below the LOD. Summing up all NEONICs, the total (Σ_8 NEONICs) exhibits a min-max interval of 0.372–273 ng/g, a median of 13 ng/g, and an average concentration measured at 25 ± 44 ng/g. Detailed statistics on quantified are found in Supplementary Appendix B-Sheet 10-Neonicotinoids.

8.4 Environmental fate and transport mechanisms

Because NEONICs dissolve in water, runoff and leaching are the usual ways through which they are transported. Plants may readily absorb them through their leaves or roots and then disperse them throughout their tissues (Borsuah et al., 2020; Pietrzak et al., 2020). Much of the applied NEONICs is lost to the surrounding soil, water, and air, with just about 5% being absorbed by the target plant (Wood and Goulson, 2017). In a leaching soil column study by Aseperi et al. (2020), the sorption potential of various NOEs in soil has been seen to differ. Specifically, thiacloprid (THA) exhibited soil adsorption up to 186 times higher than thiamethoxam (THI). Furthermore, it has been shown that the more organic matter in the soil, the more deeply NEONICs seep into the soil. It, therefore, seems obvious that NEONICs can accumulate in the soil after several applications and become persistent (Briceño et al., 2024).

9 Discussion

This review paper evaluated the presence, fate, and transportation mechanisms of various CECs grouped into pharmaceutical products (PPs), personal care products (PCPs), hormones, plastic-related compounds, rubber and polymer antioxidants, and neonicotinoid insecticides. A thorough methodology involving cross-referencing and thematic classification of existing literature provided a nuanced understanding of CECs in different environmental contexts. Figure 3 outlines the 50 most abundant CECs in biosolids, which are dominated by PAEs, followed by antifungals, chlorophenolic antimicrobials, fluoroquinolone antibiotics, and analgesics. Figure 4 provides a comprehensive overview of the occurrence of leading groups of CECs in sewage sludge or biosolids, with a predominant presence of PAEs, followed by PPs, PCPs, hormones, antioxidants, and bisphenols. Environmental transport and transformation processes are predominantly governed by factors such as solubility in water, volatility, biological and physicochemical degradation, sorption, and the potential for bioaccumulation, as described in Table 1. The findings aim to inform regulatory frameworks, illustrating how the cumulative presence of various groups of CECs in sewage sludge and soil could influence environmental health decisions. This review also underscores current knowledge gaps and identifies future research trends essential for addressing emerging environmental issues associated with CECs. By highlighting areas that require further investigation, this work lays the groundwork for enhanced monitoring and management strategies to mitigate the ecological and health impacts of CECs, thus contributing to more effective environmental governance.

In the context of CECs emanating from biosolids and similarly contaminated matrices like sewage sludge and soil, our review falls within salient research themes about CECs detected in complex media. In addition to the thematic of CECs classification, occurrence, and fate and transport mechanisms covered in this review, additional research thematic areas that have been recently investigated comprise analytical techniques, toxicity, uptake by living organisms, treatment technologies, and risk assessment. This review looked at the occurrence of various classes of CECs in biosolids, sewage sludge, and contaminated soils, providing insight into each class's cumulative occurrence. This work should help provide a framework for data needed to regulate the presence of such CECs for using biosolids for soil enhancement.

9.1 CECs' characteristics influencing their fate and transport

Sander et al. (2022) discussed the impact of Henry's Law Volatility Constant (H_v) on contaminants' environmental fate and migration pathways. The constant plays a critical role in quantitatively influencing the fate and transport of contaminants by determining their distribution between gas and liquid phases within environmental contexts. An elevated H_v value indicates that the contaminant exhibits an increased rate of evaporation, which is fundamentally significant for predicting the rate at which contaminants can escape water bodies, affecting both water quality and atmospheric pollution levels. This constant is mathematically defined as the ratio of the abundance of the contaminant in the gas phase (Q_g) to its abundance in the liquid phase (Q_l) at equilibrium under conditions of infinite dilution:

$$H_v = \frac{Q_g}{Q_l}$$

Moreover, the constant is a pivotal metric for evaluating the equilibrium distribution of contaminants between air and water, commonly referred to as air-water partitioning. Defined via partial pressure (p) of a species in the gas phase relative to its amount fraction (x) in the liquid phase under equilibrium conditions at infinite dilution, the Henry's Law volatility constant (H_v^{px}) can be expressed as follows:

$$H_v^{px} = \lim_{x \longrightarrow 0} \frac{p}{x}$$

The half-life degradation time (DT50) constitutes a significant parameter in elucidating contaminants' environmental destiny and mobility. DT50 represents the time required for the concentration of a specific contaminant to decrease by fifty percent in the environment due to biological and physicochemical degradation

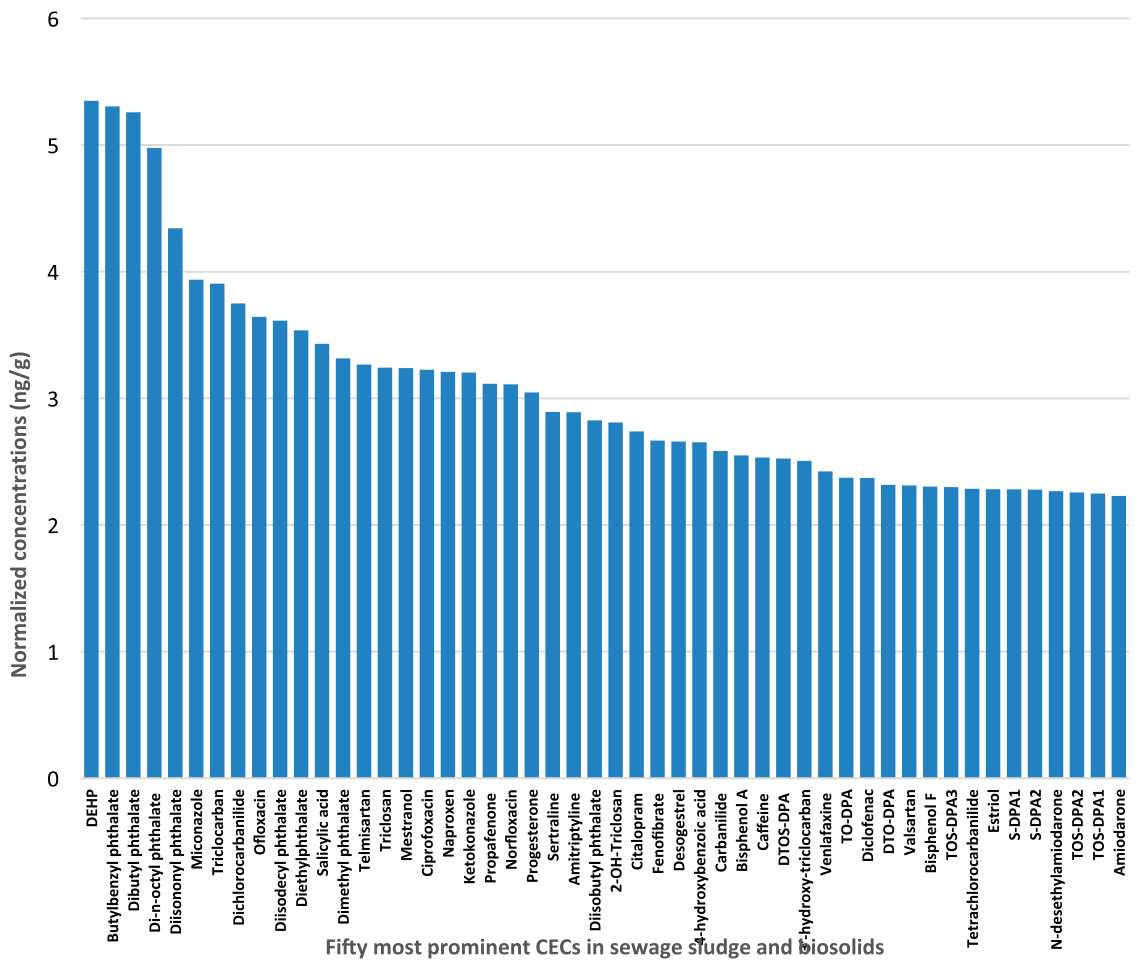

FIGURE 3
Normalized concentrations (ng/g) of the top 50 CECs in sewage sludge and biosolids.

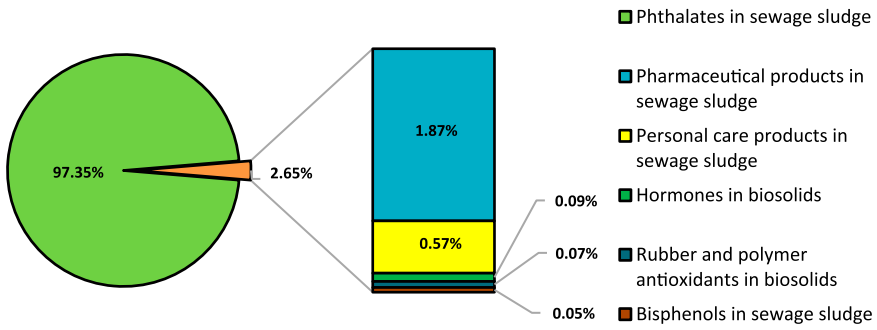

FIGURE 4
Weight percentage distribution of predominant classes of CECs found in biosolids or sewage sludge.

processes. A longer DT50 indicates a persistent contaminant, potentially resulting in extended exposure risks for aquatic ecosystems and human health. In contrast, a shorter DT50 indicates that a contaminant undergoes more rapid degradation, which may mitigate exposure risk. Environmental fate and transport models frequently incorporate DT50 as a key input parameter to simulate the movement of contaminants through diverse environmental matrices such as soil and water. By integrating DT50 values, researchers can better forecast pollutant concentrations over time in the environment, thereby aiding in assessing their potential impact and residence duration. Furthermore, understanding DT50 can guide remediation

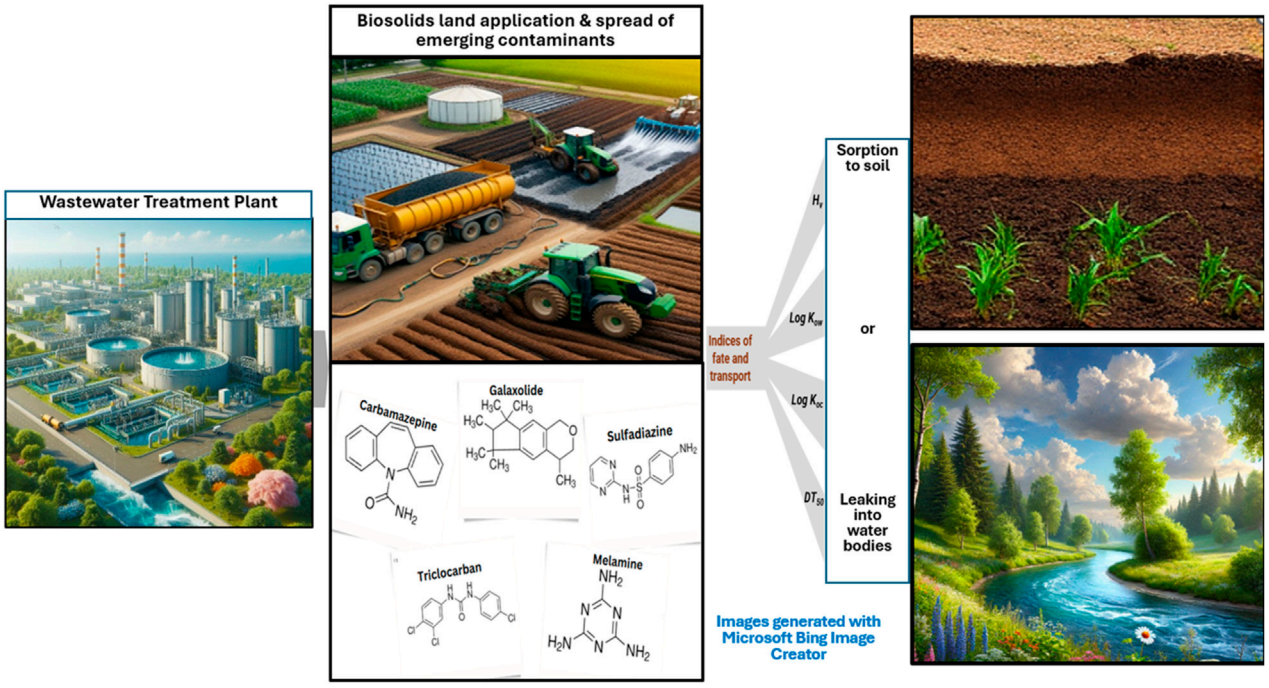

FIGURE 5
The characteristics of CECs that influence their fate and transport.

strategies by revealing whether natural degradation processes are adequate or if more proactive treatment measures are warranted (Li et al., 2015).

The organic carbon-water partitioning coefficient (log K_{oc}) and the octanol-water partitioning coefficient (log K_{ow}) are two additional characteristics that can affect the environmental fate and migratory mechanisms. The log K_{oc} value aids in forecasting the amount of a pollutant that will remain in the aqueous phase instead of being adsorbed onto soil organic matter. More significant adsorption to soils is correlated with higher log K_{oc} values. The log K_{ow} is a crucial metric for evaluating a contaminant's capacity for bioaccumulation. A chemical with a greater log K_{ow} is more hydrophobic and more likely to partition into organic tissues (USEPA, 1989). The Figure 5 summarizes the fate of CECs during WWT and land application of biosolids and highlights main characteristics that affect their migration in the environment.

9.2 Recent advances

Chemometric methodologies have been applied to optimize various parameters within analytical procedures, decipher complex datasets, and interpret results. Prominent examples include the studies conducted by Malvar et al. (2020a), Malvar et al. (2020b), who employed the Box-Behnken experimental design (BBD) to enhance extraction and cleanup techniques. The BBD represents a robust and efficient design that accommodates a quadratic model without necessitating the examination of all potential combinations; it is classified as a form of response surface methodology (RSM) aimed at optimizing processes involving multiple variables. This approach enables the meticulous adjustment of experimental conditions while minimizing the required experiments (Stalikas et al., 2009).

The grouping of contaminants is one of the key strategies proposed by regulatory entities and researchers to control the pollution of CECs (Chirsir et al., 2024; EC, 2006). Recently, advances have expanded beyond BPA to include structural analogs, providing a more holistic understanding of their occurrence in the environment (Huang et al., 2020; Xu et al., 2021a; Xue and Kannan, 2019). Future discussions should also emphasize the importance of permissible thresholds of groups of CECs. For example, bisphenol analogs exhibiting comparable or even greater toxic effects than BPA will compel regulatory bodies to reassess current standards. Specifically, Health Canada's proposed tolerable daily intake (TDI) of 25 μg/kg body weight/day applies exclusively to BPA found in food packaging materials, such as can coatings and lacquers (Health Canada, 2008). Although this proposed TDI is considered conservative, evaluating the risks associated with all bisphenol analogs across all potential exposure pathways is essential. With regards to the permissible concentration of parabens in cosmetics or food additives, the JECFA (Joint FAO/WHO Expert Committee on Food Additives) instituted an ADI (Acceptable Daily Intake) of 1–10 mg/kg body weight/day applicable to the sum of MeP and EtP (ECCC, 2020). Information on permitted concentrations in the environment and the possible harm they may cause to ecosystems is non-existent, and ADI thresholds do not consider either grouping similar substances or all possible ways of exposure. Thus, there is a need to review permissible intake limits in light of recently acquired information.

During sample preparation, progress was noticed in sample pretreatment procedures, where molecularly imprinted polymers (MIPs) were developed for the sensitive and selective extraction of

BPs from complex matrices. Illustratively, a highly class-selective dummy molecularly imprinted polymer for solid-phase extraction (DMIPSPE) and clean-up enabled to achieve LOQs as low as 0.003 ng/g and 0.0004 ng/g for BPS and BPAF, respectively, during the analysis of BP analogs in wastewater and sewage sludge samples (Sun et al., 2018). As for sample analysis, Matrix-Assisted Laser Desorption/Ionization Mass Spectrometry Imaging (MALDI-MSI) has emerged as an invaluable technique for detecting CECs in complex matrices. Particularly, the MALDI-MSI method facilitated multi-class analysis of 499 organic contaminants within biosolids (Villette et al., 2023). When contrasted with conventional GC-MS and LC-MS techniques, MALDI-MSI offers rapid and minimal sample preparation for analyses and possesses the capacity to identify a diverse range of substances, encompassing CECs. Additionally, Guironnet et al. (2022) implemented a hybrid triple quadrupole-linear ion trap (QqQ-LIT) spectrometer to perform triple-stage mass spectrometry acquisition. Within this QqQ-LIT framework, Multiple Reaction Monitoring cubed (MS/MS/MS or MRM^3) is executed through a series of sequential processes during triple-stage mass spectrometry acquisition. This methodology significantly enhances sensitivity, specificity, and quantification capabilities for complex analyses involving challenging environmental matrices.

9.3 Future trends

Antibiotics most frequently identified in sewage sludge or biosolids in this review differ from Canada's five commonly prescribed antibiotic classes. The Canadian Antimicrobial Resistance Surveillance System Report indicates that in 2019, the predominant antibiotic classes utilized in veterinary applications, quantified by weight, were tetracyclines (495,116 kg), macrolides (115,822 kg), penicillins (91,095 kg), sulfonamides (53,226 kg), and lincosamides (46,390 kg) (Public Health Agency of Canada, 2021). On the other hand, the same report lists the defined daily doses (DDD) per 1,000 inhabitants of the most prevalently administered antimicrobials for human use: tetracyclines (1,186.2), extended-spectrum penicillins (1,178.7), macrolides (781.9), penicillin combinations (489.4), first-generation cephalosporins (474.3), and fluoroquinolones (469.3) (Public Health Agency of Canada, 2021). Future research should prioritize the characterization of these antibiotic classes in biosolids and explore the correlation between their usage patterns and prevalence.

This review documents the presence of E2, NRT, NRG, and P in biosolids (Gewurtz et al., 2022). According to Costanian et al. (2018), these hormones are among the most employed in Canada in hormone replacement therapies (HRT). The Canadian HRT products primarily consist of estrogen-only formulations, with estradiol (E2) being a prevalent component (Viglino et al., 2008). These formulations are accessible in multiple forms, including oral tablets, transdermal patches, gels, and vaginal rings. Combined estrogen and progestogen therapies comprise E2 in conjunction with medroxyprogesterone Acetate (MPA), E2 combined with NRT, and E2 paired with NRG (often administered via a patch). Furthermore, formulations utilize combinations of E2 and P that are chemically identical to the endogenous hormones produced by the human body. T therapy is also available to address concerns related to diminished libido or other hormonal imbalances. Future research endeavors should cover this game of hormones and include additional ones such as MPA and others.

Advanced chemometric strategies to process and extract important information from large data sets will receive more attention due to the complexity of modern analytical procedures and instruments and the volume of data generated thereof (Malvar et al., 2020a; 2020b). Multivariate data analysis techniques will be increasingly integrated into chemometric methodologies to allow for the simultaneous exploration of several variables (Shayesteh et al., 2015). Developing chemometric techniques will also prioritize facilitating instantaneous analysis to expedite decision-making. This trend will be particularly significant for environmental monitoring and food safety sectors. Artificial intelligence and machine learning algorithms will play a major role in future data analysis and pattern recognition. Although MALDI-MSI has limitations when quantifying contaminant loads in biosolids, it can be useful for peripheral analysis that could inform traditional LC-MS or GC-MS quantitation assessments (Villette et al., 2023). These methods offer improved quantification, sensitivity, and selectivity for figuring out the quantities of particular substances in complex matrices like biosolids.

Another advancement is the increasing interest in analyzing the degradation byproducts of parent contaminants (Castro et al., 2018; Ma et al., 2018; Malvar et al., 2020b; Riva et al., 2021; Zhu et al., 2019). During the breakdown of contaminants, chemical transformations can lead to the formation of new and unregulated substances. These metabolites may pose equivalent or even higher risks than the parent compounds, as in the case of MeTCS, with a higher bioaccumulative potential than TCS (Coogan et al., 2007), and N-desethylamiodarone more prevalent in sewage sludge than amiodarone (Castro et al., 2018), leading to potentially significant ecological and human health impacts. Researchers have increasingly worked to determine the metabolites and transformation by-products of CECs.

About hormones, it is important to consider the conjugated/deconjugated process in developing analytical techniques to measure hormones to avoid underestimating their concentrations in the environment (Gewurtz et al., 2022; Naldi et al., 2016). These should have major implications since they underscore the importance of monitoring both conjugated and free forms of hormones in wastewater and soil environments. Future studies on the presence, mobility, and transformation of hormones during WWT and in the environment should consider assessing both conjugated and deconjugated forms of hormones.

While some researchers argue that the persistence of S-DPAs in the environment necessitates further investigations (Lu et al., 2019), others consider that current exposure and treatment technologies sufficiently mitigate risks posed by S-DPAs (ECCC, 2017). Further investigation and observation are necessary to better understand the occurrence and behavior of S-DPAs in the environment. With the recently documented toxicity of 6PPD-Q to coho salmon (Tian et al., 2021; Cao et al., 2022), debates regarding the toxicity levels and acceptable exposure limits to S-PPDs exist, raising questions about the permissible levels of such chemicals in the environment and whether current regulations adequately address these risks. Moreover, the recent determination of the presence of S-DPAs in Arctic seabirds and seals (Lu et al., 2019) highlights the urgency for

additional research on S-DPAs migration routes and fate in the environment, including the Arctic ecosystem. The persistent presence of S-DPAs has raised concerns regarding bioaccumulation and long-term ecological impacts; therefore, it is essential to better understand how they accumulate in food webs, including Arctic wildlife and Indigenous populations. Public awareness concerning their environmental implications will be indispensable to promote stronger advocacy for sustainable regulatory practices. Regarding S-PPDs, growth in initiatives for tracking their levels in various matrices is expected. More collaborative approaches among environmental scientists and toxicologists to tackle the complexities surrounding S-PPDs and other classes of CECs and their environmental fate will also increase. Klöckner et al. (2021) underscore the need to evaluate the implications of organic contaminants and metal constituents on toxicity and environmental health.

Author contributions

EH: Conceptualization, Data curation, Formal Analysis, Investigation, Methodology, Validation, Visualization, Writing–original draft, Writing–review and editing. SS: Conceptualization, Funding acquisition, Investigation, Project administration, Resources, Supervision, Validation, Writing–original draft, Writing–review and editing.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Generative AI statement

The author(s) declare that no Generative AI was used in the creation of this manuscript.

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Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvc.2025.1547596/full#supplementary-material

References

Abril, C., Santos, J. L., Malvar, J. L., Martín, J., Aparicio, I., and Alonso, E. (2018). Determination of perfluorinated compounds, bisphenol A, anionic surfactants and personal care products in digested sludge, compost and soil by liquid-chromatography-tandem mass spectrometry. *J. Chromatogr. A* 1576, 34–41. doi:10.1016/j.chroma.2018.09.028

Adeel, M., Song, X., Wang, Y., Francis, D., and Yang, Y. (2017). Environmental impact of estrogens on human, animal and plant life: a critical review. *Environ. Int.* 99, 107–119. doi:10.1016/j.envint.2016.12.010

Ahmed, A. A., Thiele-Bruhn, S., Aziz, S. G., Hilal, R. H., Elroby, S. A., Al-Youbi, A. O., et al. (2015). Interaction of polar and nonpolar organic pollutants with soil organic matter: sorption experiments and molecular dynamics simulation. *Sci. Total Environ.* 508, 276–287. doi:10.1016/j.scitotenv.2014.11.087

Ahn, K. C., Zhao, B., Chen, J., Cherednichenko, G., Sanmarti, E., Denison, M. S., et al. (2008). *In vitro* biologic activities of the antimicrobials triclocarban, its analogs, and triclosan in bioassay screens: receptor-based bioassay screens. *Environ. Health Perspect.* 116, 1203–1210. doi:10.1289/ehp.11200

Akter, S., Hulugalle, N. R., Jasonsmith, J., and Strong, C. L. (2023). Changes in soil microbial communities after exposure to neonicotinoids: a systematic review. *Environ. Microbiol. Rep.* 15, 431–444. doi:10.1111/1758-2229.13193

Almeida, F., and Nunes, B. (2019). Effects of acetaminophen in oxidative stress and neurotoxicity biomarkers of the gastropod Phorcus lineatus. *Environ. Sci. Pollut. Res.* 26, 9823–9831. doi:10.1007/s11356-019-04349-1

Anand, U., Adelodun, B., Cabreros, C., Kumar, P., Suresh, S., Dey, A., et al. (2022). Occurrence, transformation, bioaccumulation, risk and analysis of pharmaceutical and personal care products from wastewater: a review. *Environ. Chem. Lett.* 20, 3883–3904. doi:10.1007/s10311-022-01498-7

Arachchige Chamila Samarasinghe, S. V., Krishnan, K., Aitken, R. J., Naidu, R., and Megharaj, M. (2021). Persistence of the parabens in soil and their potential toxicity to earthworms. *Environ. Toxicol. Pharmacol.* 83, 103574. doi:10.1016/j.etap.2020.103574

Armstrong, D. L., Lozano, N., Rice, C. P., Ramirez, M., and Torrents, A. (2018). Degradation of triclosan and triclocarban and formation of transformation products in activated sludge using benchtop bioreactors. *Environ. Res.* 161, 17–25. doi:10.1016/j.envres.2017.10.048

Armstrong, D. L., Lozano, N., Rice, C. P., Ramirez, M., and Torrents, A. (2019). Fate of triclosan, triclocarban, and their transformation products in wastewater under nitrifying conditions. *J. Water Process Eng.* 28, 144–151. doi:10.1016/j.jwpe.2019.01.015

Arnnok, P., Singh, R. R., Burakham, R., Pérez-Fuentetaja, A., and Aga, D. S. (2017). Selective uptake and bioaccumulation of antidepressants in fish from effluent-impacted niagara river. *Environ. Sci. and Technol.* 51, 10652–10662. doi:10.1021/acs.est.7b02912

Artacho-Cordón, F., Fernández, M. F., Frederiksen, H., Iribarne-Durán, L. M., Jiménez-Díaz, I., Vela-Soria, F., et al. (2018). Environmental phenols and parabens in adipose tissue from hospitalized adults in Southern Spain. *Environ. Int.* 119, 203–211. doi:10.1016/j.envint.2018.05.052

Aseperi, A. K., Busquets, R., Hooda, P. S., Cheung, P. C. W., and Barker, J. (2020). Behaviour of neonicotinoids in contrasting soils. *J. Environ. Manag.* 276, 111329. doi:10.1016/j.jenvman.2020.111329

Ashraf, M., Ahammad, S. Z., and Chakma, S. (2023). Advancements in the dominion of fate and transport of pharmaceuticals and personal care products in the environment-a bibliometric study. *Environ. Sci. Pollut. Res.* 30, 64313–64341. doi:10.1007/s11356-023-26796-7

Assens, M., Frederiksen, H., Petersen, J. H., Larsen, T., Skakkebæk, N. E., Juul, A., et al. (2019). Variations in repeated serum concentrations of UV filters, phthalates, phenols and parabens during pregnancy. *Environ. Int.* 123, 318–324. doi:10.1016/j.envint.2018.11.047

Bai, X., and Acharya, K. (2019). Removal of seven endocrine disrupting chemicals (EDCs) from municipal wastewater effluents by a freshwater green alga. *Environ. Pollut.* 247, 534–540. doi:10.1016/j.envpol.2019.01.075

Bansal, O. (2022). Antibiotics in the environment: a review. *Int. J. Front. Biol. Pharm. Res.* 02 (01), 057–083. doi:10.53294/ijfbpr.2022.2.1.0030

Barroso, P. J., Santos, J. L., Martín, J., Aparicio, I., and Alonso, E. (2019). Emerging contaminants in the atmosphere: analysis, occurrence and future challenges. *Crit. Rev. Environ. Sci. Technol.* 49, 104–171. doi:10.1080/10643389.2018.1540761

Bartelt-Hunt, S. L., Snow, D. D., Kranz, W. L., Mader, T. L., Shapiro, C. A., Donk, S. J., et al. (2012). Effect of growth promotants on the occurrence of endogenous and synthetic steroid hormones on feedlot soils and in runoff from beef cattle feeding operations. *Environ. Sci. and Technol.* 46, 1352–1360. doi:10.1021/es202680q

Becky Miriyam, I., Anbalagan, K., and Magesh Kumar, M. (2022). Phthalates removal from wastewater by different methods – a review. *Water Sci. Technol.* 85, 2581–2600. doi:10.2166/wst.2022.133

Berenstein, G., Hughes, E. A., Zalts, A., Basack, S., Bonesi, S. M., and Montserrat, J. M. (2022). Environmental fate of dibutylphthalate in agricultural plastics: photodegradation, migration and ecotoxicological impact on soil. *Chemosphere* 290, 133221. doi:10.1016/j.chemosphere.2021.133221

Boix, C., Ibáñez, M., Fabregat-Safont, D., Morales, E., Pastor, L., Sancho, J. V., et al. (2016). Behaviour of emerging contaminants in sewage sludge after anaerobic digestion. *Chemosphere* 163, 296–304. doi:10.1016/j.chemosphere.2016.07.098

Bond, P., Smit, D. L., and De Ronde, W. (2022). Anabolic-androgenic steroids: how do they work and what are the risks? *Front. Endocrinol.* 13, 1059473. doi:10.3389/fendo.2022.1059473

Bonmatin, J.-M., Mitchell, E. A. D., Glauser, G., Lumawig-Heitzman, E., Claveria, F., Bijleveld Van Lexmond, M., et al. (2021). Residues of neonicotinoids in soil, water and people's hair: a case study from three agricultural regions of the Philippines. *Sci. Total Environ.* 757, 143822. doi:10.1016/j.scitotenv.2020.143822

Bonmatin, J.-M., Noome, D. A., Moreno, H., Mitchell, E. A. D., Glauser, G., Soumana, O. S., et al. (2019). A survey and risk assessment of neonicotinoids in water, soil and sediments of Belize. *Environ. Pollut.* 249, 949–958. doi:10.1016/j.envpol.2019.03.099

Borsuah, J. F., Messer, T. L., Snow, D. D., Comfort, S. D., and Mittelstet, A. R. (2020). Literature review: global neonicotinoid insecticide occurrence in aquatic environments. *Water* 12, 3388. doi:10.3390/w12123388

Briceño, G., Diez, M. C., Palma, G., Jorquera, M., Schalchli, H., Saez, J. M., et al. (2024). Neonicotinoid effects on soil microorganisms: responses and mitigation strategies. *Sustainability* 16, 3769. doi:10.3390/su16093769

Camotti Bastos, M., Soubrand, M., Le Guet, T., Le Floch, É., Joussein, E., Baudu, M., et al. (2020). Occurrence, fate and environmental risk assessment of pharmaceutical compounds in soils amended with organic wastes. *Geoderma* 375, 114498. doi:10.1016/j.geoderma.2020.114498

Cao, G., Wang, W., Zhang, J., Wu, P., Zhao, X., Yang, Z., et al. (2022). New evidence of rubber-derived quinones in water, air, and soil. *Environ. Sci. and Technol.* 56, 4142–4150. doi:10.1021/acs.est.1c07376

Carda-Broch, S., and Berthod, A. (2004). Countercurrent chromatography for the measurement of the hydrophobicity of sulfonamide amphoteric compounds. *Chromatographia* 59, 79–87. doi:10.1365/s10337-003-0140-5

Carey, D. E., and Mcnamara, P. J. (2014). The impact of triclosan on the spread of antibiotic resistance in the environment. *Front. Microbiol.* 5, 780. doi:10.3389/fmicb.2014.00780

Carey, D. E., Zitomer, D. H., Hristova, K. R., Kappell, A. D., and Mcnamara, P. J. (2016). Triclocarban influences antibiotic resistance and alters anaerobic digester microbial community structure. *Environ. Sci. and Technol.* 50, 126–134. doi:10.1021/acs.est.5b03080

Carter, L. J., Ryan, J. J., and Boxall, A. B. (2016). Does uptake of pharmaceuticals vary across earthworm species? *Bull. Environ. Contam. Toxicol.* 97, 316–322. doi:10.1007/s00128-016-1875-7

Castro, G., Carpinteiro, I., Rodríguez, I., and Cela, R. (2018). Determination of cardiovascular drugs in sewage sludge by matrix solid-phase dispersion and ultra-performance liquid chromatography tandem mass spectrometry. *Anal. Bioanal. Chem.* 410, 6807–6817. doi:10.1007/s00216-018-1268-3

Cavanagh, J.-A. E., Trought, K., Mitchell, C., Northcott, G., and Tremblay, L. A. (2018). Assessment of endocrine disruption and oxidative potential of bisphenol-A, triclosan, nonylphenol, diethylhexyl phthalate, galaxolide, and carbamazepine, common contaminants of municipal biosolids. *Toxicol. Vitro* 48, 342–349. doi:10.1016/j.tiv.2018.02.003

Chabilan, A., Landwehr, N., Horn, H., and Borowska, E. (2022). Impact of log(kow) value on the extraction of antibiotics from river sediments with pressurized liquid extraction. *Water* 14, 2534. doi:10.3390/w14162534

Chae, E., Jung, U., and Choi, S.-S. (2024). Types and concentrations of tire wear particles (TWPs) in road dust generated in slow lanes. *Environ. Pollut.* 346, 123670. doi:10.1016/j.envpol.2024.123670

Chen, M.-H., Yu, B., Zhang, Z.-F., and Ma, W.-L. (2021). Occurrence of parabens in outdoor environments: implications for human exposure assessment. *Environ. Pollut.* 282, 117058. doi:10.1016/j.envpol.2021.117058

Chen, J., Ahn, K. C., Gee, N. A., Ahmed, M. I., Duleba, A. J., Zhao, L., et al. (2008). Triclocarban enhances testosterone action: a new type of endocrine disruptor? *Endocrinology* 149, 1173–1179. doi:10.1210/en.2007-1057

Chen, J., Meng, X.-Z., Bergman, A., and Halden, R. U. (2019). Nationwide reconnaissance of five parabens, triclosan, triclocarban and its transformation products in sewage sludge from China. *J. Hazard. Mater.* 365, 502–510. doi:10.1016/j.jhazmat.2018.11.021

Chirsir, P., Palm, E. H., Baskaran, S., Schymanski, E. L., Wang, Z., Wolf, R., et al. (2024). Grouping strategies for assessing and managing persistent and mobile substances. *Environ. Sci. Eur.* 36, 102. doi:10.1186/s12302-024-00919-4

Conde-Cid, M., Fernández-Calviño, D., Fernández-Sanjurjo, M. J., Núñez-Delgado, A., Álvarez-Rodríguez, E., and Arias-Estévez, M. (2020). Effects of pine bark amendment on the transport of sulfonamide antibiotics in soils. *Chemosphere* 248, 126041. doi:10.1016/j.chemosphere.2020.126041

Coogan, M. A., Edziyie, R. E., La Point, T. W., and Venables, B. J. (2007). Algal bioaccumulation of triclocarban, triclosan, and methyl-triclosan in a North Texas wastewater treatment plant receiving stream. *Chemosphere* 67, 1911–1918. doi:10.1016/j.chemosphere.2006.12.027

Costa Junior, I. L., Machado, C. S., Pletsch, A. L., and Torres, Y. R. (2020). Simultaneous HPLC-PDA determination of commonly prescribed antidepressants and caffeine in sludge from sewage treatment plants and river sediments in the Itaipu reservoir region, Paraná, Brazil. *Int. J. Environ. Anal. Chem.* 100, 1004–1020. doi:10.1080/03067319.2019.1646738

Costanian, C., Edgell, H., Ardern, C. I., and Tamim, H. (2018). Hormone therapy use in the Canadian Longitudinal Study on Aging: a cross-sectional analysis. *Menopause* 25, 46–53. doi:10.1097/gme.0000000000000954

Czajka, C. P., and Londry, K. L. (2006). Anaerobic biotransformation of estrogens. *Sci. Total Environ.* 367, 932–941. doi:10.1016/j.scitotenv.2006.01.021

Daughton, C. G., and Ternes, T. A. (1999). Pharmaceuticals and personal care products in the environment: agents of subtle change? *Environ. Health Perspect.* 107 (Suppl. 6), 907–938. doi:10.1289/ehp.99107s6907

Diamanti-Kandarakis, E., Bourguignon, J. P., Giudice, L. C., Hauser, R., Prins, G. S., Soto, A. M., et al. (2009). Endocrine-disrupting chemicals: an Endocrine Society scientific statement. *Endocr. Rev.* 30, 293–342. doi:10.1210/er.2009-0002

EC (2006). EUR-Lex-02006R1907-20140410-EN-EURLex consolidated text: regulation (EC) No 1907/2006 of the European parliament and of the council of 18 december 2006 concerning the registration, evaluation, authorisation and restriction of chemicals (REACH), establishing a European chemicals agency, amending directive 1999/45/EC and repealing council regulation (EEC) No 793/93 and commission regulation (EC) No 1488/94 as well as council directive 76/769/EEC and commission directives 91/155/EEC, 93/67/EEC, 93/105/EC and 2000/21/EC. Available at: https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:02006R1907-20140410 (Accessed July 07, 2024).

EC (2011a). Bisphenol A: EU ban on baby bottles to enter into force tomorrow. Available at: https://ec.europa.eu/commission/presscorner/detail/en/IP_11_664 (Accessed June 13, 2023).

EC (2011b). "*Commission Directive 2011/8/EU of 28 January 2011 amending Directive 2002/72/EC as regards the restriction of use of Bisphenol A in plastic infant feeding bottles Text with EEA relevance*,". Luxembourg: Publications Office of the European Union. Available at: https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32011L0008 (Accessed July 08, 2024).

EC (2016). Commission Implementing Decision (EU) 2016/110 of 27 January 2016 not approving triclosan as an existing active substance for use in biocidal products for product-type 1. Available at: https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32016D0110 (Accessed July 09, 2024).

ECCC (2017). *Screening assessment for substituted diphenylamines*. Ottawa, Canada: Environment and Climate Change Canada. Available at: https://www.canada.ca/content/dam/eccc/documents/pdf/pded/sdpas/English%20Screening%20Assessment%20for%20Substituted%20Diphenylamines1.pdf (Accessed May 03, 2023).

ECCC (2020). Risk management scope for Parabens group. *Environ. Clim. Change Can. and Health Can.* Available at: https://www.canada.ca/content/dam/eccc/documents/pdf/pded/parabens/Risk-management-scope-parabens-group.pdf (Accessed July 10, 2024).

ECHA (2022). *Phthalates*. Helsinki, Finland: European Chemicals Agency Available at:. Available at: https://echa.europa.eu/hot-topics/phthalates (Accessed August 17, 2024).

EFSA (2023). *Bisphenol A*. Parma: EFSA. Available at: https://www.efsa.europa.eu/en/topics/topic/bisphenol (Accessed August 15, 2024).

Eike, D., Marcus, R., and Dirk, J. (2019). The database "Pharmaceuticals in the Environment" - update and new analysis. Available at: https://www.umweltbundesamt.de/en/publikationen/the-database-pharmaceuticals-in-the-environment (Accessed July 15, 2024).

Environment Canada (2020). "Canada gazette, Part I," in *Umber 41: notice requiring the preparation and implementation of pollution prevention plans with respect to triclosan in certain products*. Available at: https://gazette.gc.ca/rp-pr/p1/2020/2020-10-10/html/sup1-eng.html (Accessed October 10, 2024).

Environment Canada and Health Canada (2023). Screening assessment: urea, N-(4-chlorophenyl)-N'-(3,4-dichlorophenyl)- (triclocarban): chemical abstracts service registry number 101-20-2. Available at: https://publications.gc.ca/collections/collection_2023/eccc/En84-317-2022-eng.pdf (Accessed September 30, 2024).

EPA (2023). *Basic Information about biosolids*. Washington, DC: USEPA. Available at: https://www.epa.gov/biosolids/basic-information-about-biosolids (Accessed August 08, 2023).

European Communities (1986). Council Directive 86/278/EEC of 12 June 1986 on the protection of the environment, and in particular of the soil, when sewage sludge is used in. Available at: http://data.europa.eu/eli/dir/1986/278/oj (Accessed July 09, 2024).

FAO (2000). *FAO PESTICIDE DISPOSAL SERIES 8. Assessing soil contamination - a reference manual*. Rome: FAO Information Division. Available at: https://www.fao.org/fileadmin/templates/agphome/documents/Pests_Pesticides/Obsolete/Assessing_contamination_-_A_reference_manual.pdf (Accessed July 15, 2024).

FDA (2023). *Bisphenol A (BPA): use in food contact application*. Washington, DC: FDA. Available at: https://www.fda.gov/food/food-packaging-other-substances-come-contact-food-information-consumers/bisphenol-bpa-use-food-contact-application (Accessed August 17, 2024).

Feng, L., Hullebusch, E. D. V., Rodrigo, M. A., Esposito, G., and Oturan, M. A. (2013). Removal of residual anti-inflammatory and analgesic pharmaceuticals from aqueous systems by electrochemical advanced oxidation processes. A review. *Chem. Eng. J.* 228, 944–964. doi:10.1016/j.cej.2013.05.061

Fenni, F., Sunyer-Caldú, A., Ben Mansour, H., and Diaz-Cruz, M. S. (2022). Contaminants of emerging concern in marine areas: first evidence of UV filters and paraben preservatives in seawater and sediment on the eastern coast of Tunisia. *Environ. Pollut.* 309, 119749. doi:10.1016/j.envpol.2022.119749

Fowler, S., Roush, R., and Wise, J. (2018). *Concepts of biology*. Tallahassee, FL: 12th Media Services.

Gao, D., Li, Z., Wang, H., and Liang, H. (2018). An overview of phthalate acid ester pollution in China over the last decade: environmental occurrence and human exposure. *Sci. Total Environ.* 645, 1400–1409. doi:10.1016/j.scitotenv.2018.07.093

Gao, X., Xu, Y., Ma, M., Rao, K., and Wang, Z. (2019). Simultaneous passive sampling of hydrophilic and hydrophobic emerging organic contaminants in water. *Ecotoxicol. Environ. Saf.* 178, 25–32. doi:10.1016/j.ecoenv.2019.04.014

Gewurtz, S. B., Teslic, S., Hamilton, M. C., and Smyth, S. A. (2022). Influence of conjugation on the fate of pharmaceuticals and hormones in Canadian wastewater treatment plants. *ACS ES&T Water* 2, 329–338. doi:10.1021/acsestwater.1c00376

Girovich, M. J. (1996). *Biosolids treatment and management: processes for beneficial use*. New York: Taylor and Francis, 453.

Goeury, K., Vo Duy, S., Munoz, G., Prévost, M., and Sauvé, S. (2019). Analysis of Environmental Protection Agency priority endocrine disruptor hormones and bisphenol A in tap, surface and wastewater by online concentration liquid chromatography tandem mass spectrometry. *J. Chromatogr. A* 1591, 87–98. doi:10.1016/j.chroma.2019.01.016

Graumnitz, S., and Jungmann, D. (2021). The database "pharmaceuticals in the environment" - update for the period 2017-2020. Available at: https://www.umweltbundesamt.de/publikationen/the-database-pharmaceuticals-in-the-environment-0 (Accessed July 15, 2024).

Grgić, I., Čižmek, A.-M., Babić, S., Ljubas, D., and Rožman, M. (2021). UV filters as a driver of the antibiotic pollution in different water matrices. *J. Environ. Manag.* 289, 112389. doi:10.1016/j.jenvman.2021.112389

Guironnet, A., Wiest, L., and Vulliet, E. (2022). Advantages of MS/MS/MS (MRM3) vs classic MRM quantification for complex environmental matrices: analysis of beta-lactams in WWTP sludge. *Anal. Chim. Acta* 1205, 339773. doi:10.1016/j.aca.2022.339773

Hanselman, T. A., Graetz, D. A., and Wilkie, A. C. (2003). Manure-borne estrogens as potential environmental contaminants: a review. *Sci. Total Environ.* 37, 5471–5478. doi:10.1021/es034410+

Health Canada (2008). *Health risk assessment of bisphenol A from food packaging applications*. Ottawa, ON: Health Canada. Available at: https://www.canada.ca/content/dam/hc-sc/migration/hc-sc/fn-an/alt_formats/hpfb-dgpsa/pdf/securit/bpa_hra-ers-eng.pdf (Accessed August 14, 2024).

Health Canada (2010). List of all Challenge chemical substances. Available at: https://www.canada.ca/en/health-canada/services/chemical-substances/challenge/list.html (Accessed July 13, 2024).

Health Canada (2024). Natural health products ingredients database. Available at: https://webprod.hc-sc.gc.ca/nhpid-bdipsn/ingredReq?id=4704 (Accessed September 30, 2024).

He, L., Gielen, G., Bolan, N. S., Zhang, X., Qin, H., Huang, H., et al. (2015). Contamination and remediation of phthalic acid esters in agricultural soils in China: a review. *Agron. Sustain. Dev.* 35, 519–534. doi:10.1007/s13593-014-0270-1

Hiki, K., and Yamamoto, H. (2022). Concentration and leachability of N-(1,3-dimethylbutyl)-N′-phenyl-p-phenylenediamine (6PPD) and its quinone transformation product (6PPD-Q) in road dust collected in Tokyo, Japan. *Environ. Pollut.* 302, 119082. doi:10.1016/j.envpol.2022.119082

Hladik, M., and Kolpin, D. W. (2015). First national-scale reconnaissance of neonicotinoid insecticides in streams across the USA. *Environ. Chem.* 13, 12–20. doi:10.1071/en15061

Hofmann, T., Ghoshal, S., Tufenkji, N., Adamowski, J. F., Bayen, S., Chen, Q., et al. (2023). Plastics can be used more sustainably in agriculture. *Commun. Earth and Environ.* 4, 332. doi:10.1038/s43247-023-00982-4

Howard, P. H. (2017). *Handbook of environmental fate and exposure data: for organic chemicals, volume III pesticides*. Boca Raton, FL: CRC Press.

Huang, B., Li, X., Sun, W., Ren, D., Li, X., Li, X., et al. (2014). Occurrence, removal, and fate of progestogens, androgens, estrogens, and phenols in six sewage treatment plants around Dianchi Lake in China. *Environ. Sci. Pollut. Res.* 21, 12898–12908. doi:10.1007/s11356-014-3236-6

Huang, J., He, P., Duan, H., Yang, Z., Zhang, H., and Lü, F. (2023). Leaching risk of antibiotic resistance contamination from organic waste compost in rural areas. *Environ. Pollut.* 320, 121108. doi:10.1016/j.envpol.2023.121108

Huang, W., Shi, Y., Huang, J., Deng, C., Tang, S., Liu, X., et al. (2021). Occurrence of substituted p-phenylenediamine antioxidants in dusts. *Environ. Sci. and Technol. Lett.* 8, 381–385. doi:10.1021/acs.estlett.1c00148

Huang, Z., Zhao, J.-L., Yang, Y.-Y., Jia, Y.-W., Zhang, Q.-Q., Chen, C.-E., et al. (2020). Occurrence, mass loads and risks of bisphenol analogues in the Pearl River Delta region, South China: urban rainfall runoff as a potential source for receiving rivers. *Environ. Pollut.* 263, 114361. doi:10.1016/j.envpol.2020.114361

Huntink, N. M., Datta, R. N., and Noordermeer, J. W. M. (2004). Addressing durability of rubber compounds. *Rubber Chem. Technol.* 77, 476–511. doi:10.5254/1.3547833

Huntink, N. M. D., R, N., and Noordermeer, J. W. M. (2004). Addressing durability of rubber compounds *rubber Chemistry and technology,* 77, 476–511.

Hwang, H.-M., Fiala, M., Park, D., and Wade, T. L. (2016). Review of pollutants in urban road dust and stormwater runoff: part 1. Heavy metals released from vehicles. *Int. J. Urban Sci.* 20, 334–360. doi:10.1080/12265934.2016.1193041

Ikehata, K., Jodeiri Naghashkar, N., and Gamal El-Din, M. (2006). Degradation of aqueous pharmaceuticals by ozonation and advanced oxidation processes: a review. *Ozone Sci. and Eng.* 28, 353–414. doi:10.1080/01919510600985937

Jacobson, M. H., Woodward, M., Bao, W., Liu, B., and Trasande, L. (2019). Urinary bisphenols and obesity prevalence among U.S. Children and adolescents. *J. Endocr. Soc.* 3, 1715–1726. doi:10.1210/js.2019-00201

Jeschke, P., Nauen, R., Schindler, M., and Elbert, A. (2011). Overview of the status and global strategy for neonicotinoids. *J. Agric. Food Chem.* 59, 2897–2908. doi:10.1021/jf101303g

Jobling, S., Williams, R., Johnson, A., Taylor, A., Gross-Sorokin, M., Nolan, M., et al. (2006). Predicted exposures to steroid estrogens in U.K. rivers correlate with widespread sexual disruption in wild fish populations. *Environ. Health Perspect.* 114 (Suppl. 1), 32–39. doi:10.1289/ehp.8050

Johnston, S. J., and Cheung, K. L. (2018). Endocrine therapy for breast cancer: a model of hormonal manipulation. *Oncol. Ther.* 6, 141–156. doi:10.1007/s40487-018-0062-x

Kaewlaoyoong, A., Vu, C. T., Lin, C., Liao, C. S., and Chen, J.-R. (2018). Occurrence of phthalate esters around the major plastic industrial area in southern Taiwan. *Environ. Earth Sci.* 77, 457. doi:10.1007/s12665-018-7655-4

Kalavrouziotis, I. K., Kyritsis, S. S., and Koukoulakis, P. H. (2023). "Chapter 12 - benefits from reclaimed wastewater and biosolid reuse in agriculture and in the environment," in *Water management and circular economy*. Editors M. G. ZAMPARAS and G. L. KYRIAKOPOULOS (Amsterdam: Elsevier), 273–302.

Kassler, A., Pittenauer, E., Doerr, N., and Allmaier, G. (2014). Ultrahigh-performance liquid chromatography/electrospray ionization linear ion trap Orbitrap mass spectrometry of antioxidants (amines and phenols) applied in lubricant engineering. *Rapid Commun. Mass Spectrom.* 28, 63–76. doi:10.1002/rcm.6756

Kim, K.-R., and Owens, G. (2010). Potential for enhanced phytoremediation of landfills using biosolids – a review. *J. Environ. Manag.* 91, 791–797. doi:10.1016/j.jenvman.2009.10.017

Kinnunen, M., Piirainen, P., Kokki, H., Lammi, P., and Kokki, M. (2019). Updated clinical pharmacokinetics and pharmacodynamics of oxycodone. *Clin. Pharmacokinet.* 58, 705–725. doi:10.1007/s40262-018-00731-3

Klement, A., Kodešová, R., Bauerová, M., Golovko, O., Kočárek, M., Fér, M., et al. (2018). Sorption of citalopram, irbesartan and fexofenadine in soils: estimation of sorption coefficients from soil properties. *Chemosphere* 195, 615–623. doi:10.1016/j.chemosphere.2017.12.098

Klöckner, P., Seiwert, B., Weyrauch, S., Escher, B. I., Reemtsma, T., and Wagner, S. (2021). Comprehensive characterization of tire and road wear particles in highway tunnel road dust by use of size and density fractionation. *Chemosphere* 279, 130530. doi:10.1016/j.chemosphere.2021.130530

Kollmeyer, W. D., Flattum, R. F., Foster, J. P., Powell, J. E., Schroeder, M. E., and Soloway, S. B. (1999). "Discovery of the nitromethylene heterocycle insecticides," in *Nicotinoid insecticides and the nicotinic acetylcholine receptor*. Editors I. YAMAMOTO and J. E. CASIDA (Tokyo: Springer Japan), 71–89.

Koniecki, D., Wang, R., Moody, R. P., and Zhu, J. (2011). Phthalates in cosmetic and personal care products: concentrations and possible dermal exposure. *Environ. Res.* 111, 329–336. doi:10.1016/j.envres.2011.01.013

Koubovec, D., Ronacher, K., Stubsrud, E., Louw, A., and Hapgood, J. P. (2005). Synthetic progestins used in HRT have different glucocorticoid agonist properties. *Mol. Cell. Endocrinol.* 242, 23–32. doi:10.1016/j.mce.2005.07.001

Kubwabo, C., Kosarac, I., Stewart, B., Gauthier, B. R., Lalonde, K., and Lalonde, P. J. (2009). Migration of bisphenol A from plastic baby bottles, baby bottle liners and reusable polycarbonate drinking bottles. *Food Addit. and Contam. Part A* 26, 928–937. doi:10.1080/02652030802706725

Kumar, R., Vuppaladadiyam, A. K., Antunes, E., Whelan, A., Fearon, R., Sheehan, M., et al. (2022). Emerging contaminants in biosolids: presence, fate and analytical techniques. *Emerg. Contam.* 8, 162–194. doi:10.1016/j.emcon.2022.03.004

Kumirska, J., Łukaszewicz, P., Caban, M., Migowska, N., Plenis, A., Białk-Bielińska, A., et al. (2019). Determination of twenty pharmaceutical contaminants in soil using ultrasound-assisted extraction with gas chromatography-mass spectrometric detection. *Chemosphere* 232, 232–242. doi:10.1016/j.chemosphere.2019.05.164

Kwon, J.-W., and Xia, K. (2012). Fate of triclosan and triclocarban in soil columns with and without biosolids surface application. *Environ. Toxicol. Chem.* 31, 262–269. doi:10.1002/etc.1703

Lang, I. A., Galloway, T. S., Scarlett, A., Henley, W. E., Depledge, M., Wallace, R. B., et al. (2008). Association of urinary bisphenol A concentration with medical disorders and laboratory abnormalities in adults. *J. Am. Med. Assoc.* 300, 1303–1310. doi:10.1001/jama.300.11.1303

Larivière, A., Lissalde, S., Soubrand, M., and Casellas-Français, M. (2017). Overview of multiresidues analytical methods for the quantitation of pharmaceuticals in environmental solid matrixes: comparison of analytical development strategy for sewage sludge, manure, soil, and sediment samples. *Anal. Chem.* 89, 453–465. doi:10.1021/acs.analchem.6b04382

Larousse, D. (2024). Médicament, dictionnaire Larousse. Available at: https://www.larousse.fr/dictionnaires/francais/m%C3%A9dicament/50118 (Accessed August 04, 2024).

Lee, Y.-S., Lee, S., Lim, J.-E., and Moon, H.-B. (2019). Occurrence and emission of phthalates and non-phthalate plasticizers in sludge from wastewater treatment plants in Korea. *Sci. Total Environ.* 692, 354–360. doi:10.1016/j.scitotenv.2019.07.301

Li, A., Zhuang, T., Zhu, Q., Song, M., Liao, C., and Jiang, G. (2020). Concentration and distribution of parabens, triclosan, and triclocarban in pregnant woman serum in China. *Sci. Total Environ.* 710, 136390. doi:10.1016/j.scitotenv.2019.136390

Liang, J., Liu, Q. S., Ren, Z., Min, K., Yang, X., Hao, F., et al. (2023). Studying paraben-induced estrogen receptor- and steroid hormone-related endocrine disruption effects via multi-level approaches. *Sci. Total Environ.* 869, 161793. doi:10.1016/j.scitotenv.2023.161793

Li, H., Dong, Z., Weng, Q., Chang, C.-C., and Liu, B. (2015). Emerging pollutants – Part I: occurrence, fate and transport. *Water Environ. Res.* 87, 1849–1872. doi:10.2175/106143015x14338845156425

Li, J., Sabourin, L., Renaud, J., Halloran, S., Singh, A., Sumarah, M., et al. (2021). Simultaneous quantification of five pharmaceuticals and personal care products in biosolids and their fate in thermo-alkaline treatment. *J. Environ. Manag.* 278, 111404. doi:10.1016/j.jenvman.2020.111404

Limpiyakorn, T., Homklin, S., and Ong, S. K. (2009). "Chap 6: hormones," in *Contaminants of emerging environmental concern*. Editors A. BHANDARI, R. Y. SURAMPALLI, C. D. ADAMS, P. CHAMPAGNE, S. K. ONG, and R. D. T. T. C. ZHANG (Reston, Virginia: American Society of Civil Engineers), 186–234.

Li, R., Liang, J., Gong, Z., Zhang, N., and Duan, H. (2017). Occurrence, spatial distribution, historical trend and ecological risk of phthalate esters in the Jiulong River, Southeast China. *Sci. Total Environ.* 580, 388–397. doi:10.1016/j.scitotenv.2016.11.190

Liu, J.-D., and Wu, Y.-Q. (2019). Anabolic-androgenic steroids and cardiovascular risk. *Chin. Med. J.* 132, 2229–2236. doi:10.1097/cm9.0000000000000407

Liu, R., Li, Y., Lin, Y., Ruan, T., and Jiang, G. (2019). Emerging aromatic secondary amine contaminants and related derivatives in various dust matrices in China. *Ecotoxicol. Environ. Saf.* 170, 657–663. doi:10.1016/j.ecoenv.2018.12.036

Lorenzen, A., Hendel, J. G., Conn, K. L., Bittman, S., Kwabiah, A. B., Lazarovitz, G., et al. (2004). Survey of hormone activities in municipal biosolids and animal manures. *Environ. Toxicol.* 19, 216–225. doi:10.1002/tox.20014

Lozano, N., Rice, C. P., Ramirez, M., and Torrents, A. (2010). Fate of triclosan in agricultural soils after biosolid applications. *Chemosphere* 78, 760–766. doi:10.1016/j.chemosphere.2009.10.043

Lu, J., Li, H., Tu, Y., and Yang, Z. (2018). Biodegradation of four selected parabens with aerobic activated sludge and their transesterification product. *Ecotoxicol. Environ. Saf.* 156, 48–55. doi:10.1016/j.ecoenv.2018.02.078

Lu, Z., De Silva, A. O., Peart, T. E., Cook, C. J., Tetreault, G. R., Servos, M. R., et al. (2016). Distribution, partitioning and bioaccumulation of substituted diphenylamine antioxidants and benzotriazole UV stabilizers in an urban creek in Canada. *Environ. Sci. and Technol.* 50, 9089–9097. doi:10.1021/acs.est.6b01796

Lu, Z., De Silva, A. O., Provencher, J. F., Mallory, M. L., Kirk, J. L., Houde, M., et al. (2019). Occurrence of substituted diphenylamine antioxidants and benzotriazole UV stabilizers in Arctic seabirds and seals. *Sci. Total Environ.* 663, 950–957. doi:10.1016/j.scitotenv.2019.01.354

Lu, Z., Smyth, S. A., Peart, T. E., and De Silva, A. O. (2017). Occurrence and fate of substituted diphenylamine antioxidants and benzotriazole UV stabilizers in various Canadian wastewater treatment processes. *Water Res.* 124, 158–166. doi:10.1016/j.watres.2017.07.055

Ma, W.-L., Zhao, X., Zhang, Z.-F., Xu, T.-F., Zhu, F.-J., and Li, Y.-F. (2018). Concentrations and fate of parabens and their metabolites in two typical wastewater treatment plants in northeastern China. *Sci. Total Environ.* 644, 754–761. doi:10.1016/j.scitotenv.2018.06.358

Madsen, P. L., Thyme, J. B., Henriksen, K., Moeldrup, P., and Roslev, P. (1999). Kinetics of di-(2-ethylhexyl)phthalate mineralization in sludge-amended soil. *Environ. Sci. and Technol.* 33, 2601–2606. doi:10.1021/es981015o

Mahler, V. (2021). "Allergic reactions to rubber components," in *Contact dermatitis*. Editors J. D. JOHANSEN, V. MAHLER, J.-P. LEPOITTEVIN, and P. J. FROSCH (New York: Springer International Publishing), 1–22.

Malvar, J. L., Santos, J. L., Martín, J., Aparicio, I., and Alonso, E. (2020a). Comparison of ultrasound-assisted extraction, QuEChERS and selective pressurized liquid extraction for the determination of metabolites of parabens and pharmaceuticals in sludge. *Microchem. J.* 157, 104987. doi:10.1016/j.microc.2020.104987

Malvar, J. L., Santos, J. L., Martín, J., Aparicio, I., and Alonso, E. (2020b). Simultaneous pressurized liquid extraction and clean-up for the determination of metabolites in complex environmental solid matrices. *Microchem. J.* 152, 104370. doi:10.1016/j.microc.2019.104370

Mannina, G., Barbara, L., Cosenza, A., and Wang, Z. (2023). "Chapter 2 - treatment and disposal of sewage sludge from wastewater in a circular economy perspective," in *Current developments in biotechnology and bioengineering smart solutions for wastewater: road-mapping the transition to circular economy*. Editors G. MANNINA, A. PANDEY, and R. SIROHI (Amsterdam: Elsevier), 11–30.

Martín-Pozo, L., Gómez-Regalado, M. D. C., Moscoso-Ruiz, I., and Zafra-Gómez, A. (2021). Analytical methods for the determination of endocrine disrupting chemicals in cosmetics and personal care products: a review. *Talanta* 234, 122642. doi:10.1016/j.talanta.2021.122642

Mcbride, M. B. (1995). Toxic metal accumulation from agricultural use of sludge: are USEPA regulations protective? *J. Environ. Qual.* 24, 5–18. doi:10.2134/jeq1995.00472425002400010002x

Mercier, F., Glorennec, P., Thomas, O., and Le Bot, B. (2011). Organic contamination of settled house dust, a review for exposure assessment purposes. *Environ. Sci. and Technol.* 45, 6716–6727. doi:10.1021/es200925h

Mercl, F., Košnář, Z., Maršík, P., Vojtíšek, M., Dušek, J., Száková, J., et al. (2021). Pyrolysis of biosolids as an effective tool to reduce the uptake of pharmaceuticals by plants. *J. Hazard. Mater.* 405, 124278. doi:10.1016/j.jhazmat.2020.124278

Mesnage, R., Phedonos, A., Arno, M., Balu, S., Corton, J. C., and Antoniou, M. N. (2017). Editor's highlight: transcriptome profiling reveals bisphenol A alternatives activate estrogen receptor alpha in human breast cancer cells. *Toxicol. Sci.* 158, 431–443. doi:10.1093/toxsci/kfx101

Moschet, C., Anumol, T., Lew, B. M., Bennett, D. H., and Young, T. M. (2018). Household dust as a repository of chemical accumulation: new insights from a comprehensive high-resolution mass spectrometric study. *Environ. Sci. and Technol.* 52, 2878–2887. doi:10.1021/acs.est.7b05767

Moško, J., Pohořelý, M., Cajthaml, T., Jeremiáš, M., Robles-Aguilar, A. A., Skoblia, S., et al. (2021). Effect of pyrolysis temperature on removal of organic pollutants present in anaerobically stabilized sewage sludge. *Chemosphere* 265, 129082. doi:10.1016/j.chemosphere.2020.129082

Munoz, G., Michaud, A. M., Liu, M., Vo Duy, S., Montenach, D., Resseguier, C., et al. (2022). Target and nontarget screening of PFAS in biosolids, composts, and other organic waste products for land application in France. *Environ. Sci. and Technol.* 56, 6056–6068. doi:10.1021/acs.est.1c03697

Naldi, A. C., Fayad, P. B., Prévost, M., and Sauvé, S. (2016). Analysis of steroid hormones and their conjugated forms in water and urine by on-line solid-phase extraction coupled to liquid chromatography tandem mass spectrometry. *Chem. Central J.* 10, 30. doi:10.1186/s13065-016-0174-z

Net, S., Sempéré, R., Delmont, A., Paluselli, A., and Ouddane, B. (2015). Occurrence, fate, behavior and ecotoxicological state of phthalates in different environmental matrices. *Environ. Sci. and Technol.* 49, 4019–4035. doi:10.1021/es505233b

NIEHS (2023). *Bisphenol A (BPA). Natl. Inst. Environ. Health Sci.* Available at: https://www.niehs.nih.gov/health/topics/agents/sya-bpa (Accessed August 17, 2024).

OECD (2016). *Case Study on the Use of integrated Approaches for Testing and Assessment for repeat dose Toxicity of substituted diphenylamines (SDPA)*. Paris, France: Organisation for Economic Co-operation and Development. Available at: https://one.oecd.org/document/ENV/JM/MONO(2016)50/en/pdf (Accessed December 08, 2024).

Olsen, P., Bach, K., Barlebo, H. C., Ingerslev, F., Hansen, M., Sørensen, B. H., et al. (2007). Leaching of estrogenic hormones from manure-treated structured soils. *Environ. Sci. and Technol.* 41, 3911–3917. doi:10.1021/es0627747

Park, S.-R., Park, S.-J., Jeong, M.-J., Choi, J. C., and Kim, M. (2018). Fast and simple determination and exposure assessment of bisphenol A, phenol, p-tert-butylphenol, and diphenylcarbonate transferred from polycarbonate food-contact materials to food simulants. *Chemosphere* 203, 300–306. doi:10.1016/j.chemosphere.2018.03.185

Peng, G., Lu, Y., You, W., Yin, Z., Li, Y., and Gao, Y. (2020). Analysis of five bisphenol compounds in sewage sludge by dispersive solid-phase extraction with magnetic montmorillonite. *Microchem. J.* 157, 105040. doi:10.1016/j.microc.2020.105040

Pérez, D. J., Lombardero, L. R., and Doucette, W. J. (2023). Influence of exposure time, physicochemical properties, and plant transpiration on the uptake dynamics and translocation of pharmaceutical and personal care products in the aquatic macrophyte Typha latifolia. *Sci. Total Environ.* 896, 165107. doi:10.1016/j.scitotenv.2023.165107

Pérez-Lemus, N., López-Serna, R., Pérez-Elvira, S. I., and Barrado, E. (2019). Analytical methodologies for the determination of pharmaceuticals and personal care products (PPCPs) in sewage sludge: a critical review. *Anal. Chim. Acta* 1083, 19–40. doi:10.1016/j.aca.2019.06.044

Pérez-Lemus, N., López-Serna, R., Pérez-Elvira, S. I., and Barrado, E. (2020). Sample pre-treatment and analytical methodology for the simultaneous determination of pharmaceuticals and personal care products in sewage sludge. *Chemosphere* 258, 127273. doi:10.1016/j.chemosphere.2020.127273

Pietrzak, D., Kania, J., Kmiecik, E., Malina, G., and Wątor, K. (2020). Fate of selected neonicotinoid insecticides in soil-water systems: current state of the art and knowledge gaps. *Chemosphere* 255, 126981. doi:10.1016/j.chemosphere.2020.126981

Prosser, R. S., Parrott, J. L., Galicia, M., Shires, K., Sullivan, C., Toito, J., et al. (2017). Toxicity of sediment-associated substituted phenylamine antioxidants on the early life stages of *Pimephales promelas* and a characterization of effects on freshwater organisms. *Environ. Toxicol. Chem.* 36, 2730–2738. doi:10.1002/etc.3828

Public Health Agency of Canada (2021). Canadian antimicrobial resistance surveillance system report. Available at: https://www.canada.ca/content/dam/phac-aspc/documents/services/publications/drugs-health-products/canadian-antimicrobial-resistance-surveillance-system-report-2021/canadian-antimicrobial-resistance-surveillance-system-report-2021.pdf (Accessed December 02, 2024).

Pullagurala, V. L. R., Rawat, S., Adisa, I. O., Hernandez-Viezcas, J. A., Peralta-Videa, J. R., and Gardea-Torresdey, J. L. (2018). Plant uptake and translocation of contaminants of emerging concern in soil. *Sci. Total Environ.* 636, 1585–1596. doi:10.1016/j.scitotenv.2018.04.375

Rabii, F. W., Segura, P. A., Fayad, P. B., and Sauvé, S. (2014). Determination of six chemotherapeutic agents in municipal wastewater using online solid-phase extraction coupled to liquid chromatography-tandem mass spectrometry. *Sci. Total Environ.* 487, 792–800. doi:10.1016/j.scitotenv.2013.12.050

Riva, F., Zuccato, E., Pacciani, C., Colombo, A., and Castiglioni, S. (2021). A multi-residue analytical method for extraction and analysis of pharmaceuticals and other selected emerging contaminants in sewage sludge. *Anal. Methods* 13, 526–535. doi:10.1039/d0ay02027c

Rochester, J. R., and Bolden, A. L. (2015). Bisphenol S and F: a systematic review and comparison of the hormonal activity of bisphenol A substitutes. *Environ. Health Perspect.* 123, 643–650. doi:10.1289/ehp.1408989

Rødland, E. S., Heier, L. S., Lind, O. C., and Meland, S. (2023). High levels of tire wear particles in soils along low traffic roads. *Sci. Total Environ.* 903, 166470. doi:10.1016/j.scitotenv.2023.166470

Rout, P. R., Zhang, T. C., Bhunia, P., and Surampalli, R. Y. (2021). Treatment technologies for emerging contaminants in wastewater treatment plants: a review. *Sci. Total Environ.* 753, 141990. doi:10.1016/j.scitotenv.2020.141990

Saliu, T. D., and Sauvé, S. (2024). A review of per- and polyfluoroalkyl substances in biosolids: geographical distribution and regulations. *Front. Environ. Chem.* 5. doi:10.3389/fenvc.2024.1383185

Sánchez-Piñero, J., Bowerbank, S. L., Moreda-Piñeiro, J., López-Mahía, P., and Dean, J. R. (2020). The occurrence and distribution of polycyclic aromatic hydrocarbons, bisphenol A and organophosphate flame retardants in indoor dust and soils from public open spaces: implications for human exposure. *Environ. Pollut.* 266, 115372. doi:10.1016/j.envpol.2020.115372

Sander, R., Acree, W. E., Visscher, A. D., Schwartz, S. E., and Wallington, T. J. (2022). Henry's law constants (IUPAC Recommendations 2021). *Pure Appl. Chem.* 94, 71–85. doi:10.1515/pac-2020-0302

Sartor, O., and Gillessen, S. (2014). Treatment sequencing in metastatic castrate-resistant prostate cancer. *Asian J. Androl.* 16, 426–431. doi:10.4103/1008-682x.126378

Sauvé, S., and Desrosiers, M. (2014). A review of what is an emerging contaminant. *Chem. Central J.* 8, 15. doi:10.1186/1752-153x-8-15

Schaffer, M., Boxberger, N., Börnick, H., Licha, T., and Worch, E. (2012). Sorption influenced transport of ionizable pharmaceuticals onto a natural sandy aquifer sediment at different pH. *Chemosphere* 87, 513–520. doi:10.1016/j.chemosphere.2011.12.053

Shaheen, J. F., Sizirici, B., and Yildiz, I. (2022). Fate, transport, and risk assessment of widely prescribed pharmaceuticals in terrestrial and aquatic systems: a review. *Emerg. Contam.* 8, 216–228. doi:10.1016/j.emcon.2022.04.001

Shahid, M. K., Kashif, A., Fuwad, A., and Choi, Y. (2021). Current advances in treatment technologies for removal of emerging contaminants from water – a critical review. *Coord. Chem. Rev.* 442, 213993. doi:10.1016/j.ccr.2021.213993

Shang, M., Dong, J., Xie, H., Wang, Y., and Du, Y. (2023). Source, transport, and fate of perfluoroalkyl acids (PFAAs) in turbid bay environments: significant roles of suspended sediment and water column stratification. *Water Res.* 243, 120384. doi:10.1016/j.watres.2023.120384

Shayesteh, T. H., Radmehr, M., Khajavi, F., and Mahjub, R. (2015). Application of chemometrics in determination of the acid dissociation constants (pKa) of several benzodiazepine derivatives as poorly soluble drugs in the presence of ionic surfactants. *Eur. J. Pharm. Sci.* 69, 44–50. doi:10.1016/j.ejps.2014.12.013

Shin, M.-Y., Shin, C., Choi, J. W., Lee, J., Lee, S., and Kim, S. (2019). Pharmacokinetic profile of propyl paraben in humans after oral administration. *Environ. Int.* 130, 104917. doi:10.1016/j.envint.2019.104917

Shi, W., Wang, L., Rousseau, D. P. L., and Lens, P. N. L. (2010). Removal of estrone, 17α-ethinylestradiol, and 17ß-estradiol in algae and duckweed-based wastewater treatment systems. *Environ. Sci. Pollut. Res.* 17, 824–833. doi:10.1007/s11356-010-0301-7

Sidhu, J. P. S., Ahmed, W., Gernjak, W., Aryal, R., Mccarthy, D., Palmer, A., et al. (2013). Sewage pollution in urban stormwater runoff as evident from the widespread presence of multiple microbial and chemical source tracking markers. *Sci. Total Environ.* 463-464, 488–496. doi:10.1016/j.scitotenv.2013.06.020

Silva, S., Rodrigues, J. A., Coelho, M. R., Martins, A., Cardoso, E., Cardoso, V. V., et al. (2021). Occurrence of pharmaceutical active compounds in sewage sludge from two urban wastewater treatment plants and their potential behaviour in agricultural soils. *Environ. Sci. Water Res. and Technol.* 7, 969–982. doi:10.1039/d1ew00132a

Simon-Delso, N., Amaral-Rogers, V., Belzunces, L. P., Bonmatin, J. M., Chagnon, M., Downs, C., et al. (2015). Systemic insecticides (neonicotinoids and fipronil): trends, uses, mode of action and metabolites. *Environ. Sci. Pollut. Res.* 22, 5–34. doi:10.1007/s11356-014-3470-y

Smith, S. R. (2009). Organic contaminants in sewage sludge (biosolids) and their significance for agricultural recycling. *Philosophical Trans. R. Soc.* A, 3674005–3674041. doi:10.1098/rsta.2009.0154

Solliec, M., Roy-Lachapelle, A., Gasser, M.-O., Coté, C., Généreux, M., and Sauvé, S. (2016). Fractionation and analysis of veterinary antibiotics and their related degradation products in agricultural soils and drainage waters following swine manure amendment. *Sci. Total Environ.* 543, 524–535. doi:10.1016/j.scitotenv.2015.11.061

Soni, M. G., Carabin, I. G., and Burdock, G. A. (2005). Safety assessment of esters of p-hydroxybenzoic acid (parabens). *Food Chem. Toxicol.* 43, 985–1015. doi:10.1016/j.fct.2005.01.020

Srinivasan, P., Sarmah, A. K., and Manley-Harris, M. (2013). Co-contaminants and factors affecting the sorption behaviour of two sulfonamides in pasture soils. *Environ. Pollut.* 180, 165–172. doi:10.1016/j.envpol.2013.05.022

Stalikas, C., Fiamegos, Y., Sakkas, V., and Albanis, T. (2009). Developments on chemometric approaches to optimize and evaluate microextraction. *J. Chromatogr. A* 1216, 175–189. doi:10.1016/j.chroma.2008.11.060

Stefanakis, A., and Becker, J. (2015). "Chapter 3: a review of emerging contaminants in water: classification, sources, and potential risks," in *Impact of water pollution on human health and environmental sustainability*. Editors A. E. MCKEOWN and G. BUGYI (Hershey PA: IGI Global), 57–82.

Stoiber, T. (2019). *What are parabens, and why don't they belong in cosmetics?* Washington DC: Environmental Working Group. Available at: https://www.ewg.org/what-are-parabens (Accessed August 10, 2024).

Sun, X., Peng, J., Wang, M., Wang, J., Tang, C., Yang, L., et al. (2018). Determination of nine bisphenols in sewage and sludge using dummy molecularly imprinted solid-phase extraction coupled with liquid chromatography tandem mass spectrometry. *J. Chromatogr. A* 1552, 10–16. doi:10.1016/j.chroma.2018.04.004

Svahn, O., and Björklund, E. (2019). Extraction efficiency of a commercial espresso machine compared to a stainless-steel column pressurized hot water extraction (PHWE) system for the determination of 23 pharmaceuticals, antibiotics and hormones in sewage sludge. *Appl. Sci.* 9, 1509. doi:10.3390/app9071509

Swan Shanna, H., Main Katharina, M., Liu, F., Stewart Sara, L., Kruse Robin, L., Calafat Antonia, M., et al. (2005). Decrease in anogenital distance among male infants with prenatal phthalate exposure. *Environ. Health Perspect.* 113, 1056–1061. doi:10.1289/ehp.8100

Tao, Y., Cui, Y., Zhu, G., Zhong, G., and Zhang, Y. (2023). Fate, ecotoxicity, and remediation of phthalic acid ester in soils. *Curr. Opin. Environ. Sci. and Health* 32, 100440. doi:10.1016/j.coesh.2022.100440

Teymoorian, T., Munoz, G., Vo Duy, S., Liu, J., and Sauvé, S. (2023). Tracking PFAS in drinking water: a review of analytical methods and worldwide occurrence trends in tap water and bottled water. *ACS ES&T Water* 3, 246–261. doi:10.1021/acsestwater.2c00387

Tian, Z., Zhao, H., Peter, K. T., Gonzalez, M., Wetzel, J., Wu, C., et al. (2021). A ubiquitous tire rubber–derived chemical induces acute mortality in coho salmon. *Science* 371, 185–189. doi:10.1126/science.abd6951

Tisler, S., Liang, C., Carvalho, P. N., and Bester, K. (2021). Identification of more than 100 new compounds in the wastewater: fate of polyethylene/polypropylene oxide copolymers and their metabolites in the aquatic environment. *Sci. Total Environ.* 761, 143228. doi:10.1016/j.scitotenv.2020.143228

Tomizawa, M., and Casida, J. E. (2005). Neonicotinoid insecticide toxicology: mechanisms of selective action. *Annu. Rev. Pharmacol. Toxicol.* 45, 247–268. doi:10.1146/annurev.pharmtox.45.120403.095930

Tran, H.-T., Lin, C., Lam, S. S., Le, T. H., Hoang, H.-G., Bui, X.-T., et al. (2023). Biodegradation of high di-(2-Ethylhexyl) phthalate (DEHP) concentration by food waste composting and its toxicity assessment using seed germination test. *Environ. Pollut.* 316, 120640. doi:10.1016/j.envpol.2022.120640

Tuan Tran, H., Lin, C., Bui, X.-T., Ky Nguyen, M., Dan Thanh Cao, N., Mukhtar, H., et al. (2022). Phthalates in the environment: characteristics, fate and transport, and advanced wastewater treatment technologies. *Bioresour. Technol.* 344, 126249. doi:10.1016/j.biortech.2021.126249

UNEP (2019). *Stockholm Convention on persistent organic pollutants (POPs)-Text and annexes Revised in 2019*. Stockholm: Secretariat of the Stockholm Convention. Available at: https://www.pops.int/Portals/0/download.aspx?d=UNEP-POPS-COP-CONVTEXT-2021.English.pdf (Accessed May 18, 2023).

USEPA (1989). *Appendix K: soil organic carbon (Koc)/water (Kow) partition coefficients*. Washington, DC: Office of Emergency and Remedial Response, USEPA. Available at: https://semspub.epa.gov/work/HQ/175223.pdf.

USEPA (1994). *A plain English Guide to the EPA Part 503 biosolids rule*. Washington, DC: U.S. Environmental Protection Agency. Available at: https://www.epa.gov/sites/default/files/2018-12/documents/plain-english-guide-part503-biosolids-rule.pdf (Accessed March 23, 2023).

USEPA (2023a). *Risk Management for phthalates*. Washington, DC: USEPA. Available at: https://www.epa.gov/assessing-and-managing-chemicals-under-tsca/risk-management-phthalates (Accessed August 17, 2024).

USEPA (2023b). *Toxic and priority pollutants Under the clean water act*. Available at: https://www.epa.gov/sites/default/files/2015-09/documents/priority-pollutant-list-epa.pdf (Accessed July 24, 2024).

USFDA (2016). Safety and effectiveness of consumer antiseptics; topical antimicrobial drug products for over-the-counter human use. Available at: https://www.federalregister.gov/documents/2016/09/06/2016-21337/safety-and-effectiveness-of-consumer-antiseptics-topical-antimicrobial-drug-products-for (Accessed May 16, 2024).

Vaudreuil, M.-A., Vo Duy, S., Munoz, G., and Sauvé, S. (2022). Pharmaceutical pollution of hospital effluents and municipal wastewaters of Eastern Canada. *Sci. Total Environ.* 846, 157353. doi:10.1016/j.scitotenv.2022.157353

Viglino, L., Aboulfadl, K., Prévost, M., and Sauvé, S. (2008). Analysis of natural and synthetic estrogenic endocrine disruptors in environmental waters using online preconcentration coupled with LC-APPI-MS/MS. *Talanta* 76, 1088–1096. doi:10.1016/j.talanta.2008.05.008

Villette, C., Maurer, L., Zumsteg, J., Mutterer, J., Wanko, A., and Heintz, D. (2023). Mass spectrometry imaging for biosolids characterization to assess ecological or health risks before reuse. *Nat. Commun.* 14, 4244. doi:10.1038/s41467-023-40051-0

Viñas, R., and Watson, C. S. (2013). Bisphenol S disrupts estradiol-induced nongenomic signaling in a rat pituitary cell line: effects on cell functions. *Environ. Health Perspect.* 121, 352–358. doi:10.1289/ehp.1205826

Vom Saal, F. S., Akingbemi, B. T., Belcher, S. M., Birnbaum, L. S., Crain, D. A., Eriksen, M., et al. (2007). Chapel Hill bisphenol A expert panel consensus statement: integration of mechanisms, effects in animals and potential to impact human health at current levels of exposure. *Reprod. Toxicol.* 24, 131–138. doi:10.1016/j.reprotox.2007.07.005

Wang, L. K., Shammas, N. K., and Hung, Y. T. (2009). *Biosolids engineering and management*. Totowa, NJ: Humana Press, 802.

Wang, H., Liu, Z.-H., Zhang, J., Huang, R.-P., Yin, H., Dang, Z., et al. (2019). Insights into removal mechanisms of bisphenol A and its analogues in municipal wastewater treatment plants. *Sci. Total Environ.* 692, 107–116. doi:10.1016/j.scitotenv.2019.07.134

Wang, Q., Lv, K.-N., Wang, A.-T., Liu, X., Yin, G., Wang, J., et al. (2022). Release of phthalate esters from a local landfill in the Tibetan Plateau: importance of soil particle-size specific association. *Sci. Total Environ.* 806, 151281. doi:10.1016/j.scitotenv.2021.151281

Wang, Y., and Qian, H. (2021). Phthalates and their impacts on human health. *Healthc. (Basel)* 9 (5), 603. doi:10.3390/healthcare9050603

Wei, L., Li, Z., Sun, J., and Zhu, L. (2020). Pollution characteristics and health risk assessment of phthalate esters in agricultural soil and vegetables in the Yangtze River Delta of China. *Sci. Total Environ.* 726, 137978. doi:10.1016/j.scitotenv.2020.137978

WHO (2024). Pharmaceutical products. Available at: https://www.emro.who.int/health-topics/pharmaceutical-products/index.html#:~:text=Photo%20credit%3A%20WHOPharmaceutical%20products,in%20modern%20and%20traditional%20medicine (Accessed August 03, 2024).

Wijesekara, H., Bolan, N. S., Kumarathilaka, P., Geekiyanage, N., Kunhikrishnan, A., Seshadri, B., et al. (2016). Chapter 3 - biosolids enhance mine site rehabilitation and revegetation. *In:* M. N. V. PRASAD and K. SHIH (eds.) *Environmental materials and waste resource recovery and pollution prevention.* London: Academic Press, 45–71.

Wood, T. J., and Goulson, D. (2017). The environmental risks of neonicotinoid pesticides: a review of the evidence post 2013. *Environ. Sci. Pollut. Res.* 24, 17285–17325. doi:10.1007/s11356-017-9240-x

Wu, X., Ren, J., Xu, Q., Xiao, Y., Li, X., and Peng, Y. (2023). Priority screening of contaminant of emerging concern (CECs) in surface water from drinking water sources in the lower reaches of the Yangtze River based on exposure-activity ratios (EARs). *Sci. Total Environ.* 856, 159016. doi:10.1016/j.scitotenv.2022.159016

Xuan, R., Blassengale, A. A., and Wang, Q. (2008). Degradation of estrogenic hormones in a silt loam soil. *J. Agric. Food Chem.* 56, 9152–9158. doi:10.1021/jf8016942

Xue, J., and Kannan, K. (2019). Mass flows and removal of eight bisphenol analogs, bisphenol A diglycidyl ether and its derivatives in two wastewater treatment plants in New York State, USA. *Sci. Total Environ.* 648, 442–449. doi:10.1016/j.scitotenv.2018.08.047

Xu, Y., Hu, A., Li, Y., He, Y., Xu, J., and Lu, Z. (2021a). Determination and occurrence of bisphenol A and thirteen structural analogs in soil. *Chemosphere* 277, 130232. doi:10.1016/j.chemosphere.2021.130232

Xu, Y., Yu, X., Xu, B., Peng, D., and Guo, X. (2021b). Sorption of pharmaceuticals and personal care products on soil and soil components: influencing factors and mechanisms. *Sci. Total Environ.* 753, 141891. doi:10.1016/j.scitotenv.2020.141891

Yang, Y., Ok, Y. S., Kim, K.-H., Kwon, E. E., and Tsang, Y. F. (2017). Occurrences and removal of pharmaceuticals and personal care products (PPCPs) in drinking water and water/sewage treatment plants: a review. *Sci. Total Environ.* 596-597, 303–320. doi:10.1016/j.scitotenv.2017.04.102

Ye, X., Bishop Amber, M., Reidy John, A., Needham Larry, L., and Calafat Antonia, M. (2006). Parabens as urinary biomarkers of exposure in humans. *Environ. Health Perspect.* 114, 1843–1846. doi:10.1289/ehp.9413

Ying, G. G., and Kookana, R. S. (2005). Sorption and degradation of estrogen-like-endocrine disrupting chemicals in soil. *Environ. Toxicol. Chem.* 24, 2640–2645. doi:10.1897/05-074r.1

Ying, Z., Guo, B., Zhang, G., Sun, L., Yang, X., and Zhang, Q. (2022). The characteristics and potential risks of neonicotinoid residues in soils of different types of land use in hangzhou. *Ecotoxicol. Environ. Saf.* 245, 114091. doi:10.1016/j.ecoenv.2022.114091

Yoshida, T., Mimura, M., and Sakon, N. (2020). Intakes of phthalates by Japanese children and the contribution of indoor air quality in their residences. *Environ. Sci. Pollut. Res.* 27, 19577–19591. doi:10.1007/s11356-020-08397-w

Zahedi, S., Gros, M., Petrović, M., Balcazar, J. L., and Pijuan, M. (2022). Anaerobic treatment of swine manure under mesophilic and thermophilic temperatures: fate of veterinary drugs and resistance genes. *Sci. Total Environ.* 818, 151697. doi:10.1016/j.scitotenv.2021.151697

Zeng, F., Cui, K., Xie, Z., Wu, L., Liu, M., Sun, G., et al. (2008). Phthalate esters (PAEs): emerging organic contaminants in agricultural soils in peri-urban areas around Guangzhou, China. *Environ. Pollut.* 156, 425–434. doi:10.1016/j.envpol.2008.01.045

Zhang, Y.-L., Lin, S.-S., Dai, C.-M., Shi, L., and Zhou, X.-F. (2014). Sorption–desorption and transport of trimethoprim and sulfonamide antibiotics in agricultural soil: effect of soil type, dissolved organic matter, and pH. *Environ. Sci. Pollut. Res.* 21, 5827–5835. doi:10.1007/s11356-014-2493-8

Zhang, Z.-F., Zhang, X., Sun, M.-Y., Meng, B., Liu, L.-Y., Song, W.-W., et al. (2020b). Substituted diphenylamine antioxidants (SDPAs) in typical domestic wastewater treatment plants and Songhua River in the northeast of China. *Chemosphere* 260, 127519. doi:10.1016/j.chemosphere.2020.127519

Zhang, Z.-F., Zhang, X., Sverko, E., Marvin, C. H., Jobst, K. J., Smyth, S. A., et al. (2020c). Determination of diphenylamine antioxidants in wastewater/biosolids and sediment. *Environ. Sci. and Technol. Lett.* 7, 102–110. doi:10.1021/acs.estlett.9b00796

Zhang, Z.-F., Zhang, X., Zhang, X., Sverko, E., Smyth, S. A., and Li, Y.-F. (2021c). Diphenylamine Antioxidants in wastewater influent, effluent, biosolids and landfill leachate: contribution to environmental releases. *Water Res.* 189, 116602. doi:10.1016/j.watres.2020.116602

Zhang, C., Yi, X., Chen, C., Tian, D., Liu, H., Xie, L., et al. (2020a). Contamination of neonicotinoid insecticides in soil-water-sediment systems of the urban and rural areas in a rapidly developing region: guangzhou, South China. *Environ. Int.* 139, 105719. doi:10.1016/j.envint.2020.105719

Zhang, H. (2017). Using biosolids as a plant nutrient source. Available at: https://extension.okstate.edu/fact-sheets/using-biosolids-as-a-plant-nutrient-source.html (Accessed August 08, 2017).

Zhang, M., Deng, Y.-L., Liu, C., Chen, P.-P., Luo, Q., Miao, Y., et al. (2021a). Urinary phthalate metabolite concentrations, oxidative stress and thyroid function biomarkers among patients with thyroid nodules. *Environ. Pollut.* 272, 116416. doi:10.1016/j.envpol.2020.116416

Zhang, Y., Duy, S. V., Whalen, J. K., Munoz, G., Gao, X., and Sauvé, S. (2023). Cyanotoxins dissipation in soil: evidence from microcosm assays. *J. Hazard. Mater.* 454, 131534. doi:10.1016/j.jhazmat.2023.131534

Zhang, Y., Huang, F., Wang, L., Guan, R., Yu, H., Wang, L., et al. (2021b). Facilitating effect of heavy metals on di(2-ethylhexyl) phthalate adsorption in soil: New evidence from adsorption experiment data and quantum chemical simulation. *Sci. Total Environ.* 772, 144980. doi:10.1016/j.scitotenv.2021.144980

Zhang, Z., Le Velly, M., Rhind, S. M., Kyle, C. E., Hough, R. L., Duff, E. I., et al. (2015). A study on temporal trends and estimates of fate of Bisphenol A in agricultural soils after sewage sludge amendment. *Sci. Total Environ.* 515-516, 1–11. doi:10.1016/j.scitotenv.2015.01.053

Zhang, Z., Sverko, E., Smyth, S. A., and Marvin, C. H. (2016). Determination of substituted diphenylamines in environmental samples. *Anal. Bioanal. Chem.* 408, 7945–7954. doi:10.1007/s00216-016-9881-5

Zhi, D., Yang, D., Zheng, Y., Yang, Y., He, Y., Luo, L., et al. (2019). Current progress in the adsorption, transport and biodegradation of antibiotics in soil. *J. Environ. Manag.* 251, 109598. doi:10.1016/j.jenvman.2019.109598

Zhong, M., Wang, T., Zhao, W., Huang, J., Wang, B., Blaney, L., et al. (2022). Emerging organic contaminants in Chinese surface water: identification of priority pollutants. *Engineering* 11, 111–125. doi:10.1016/j.eng.2020.12.023

Zhou, L., Chen, H., Xu, Q., Han, X., Zhao, Y., Song, X., et al. (2019). The effect of di-2-ethylhexyl phthalate on inflammation and lipid metabolic disorder in rats. *Ecotoxicol. Environ. Saf.* 170, 391–398. doi:10.1016/j.ecoenv.2018.12.009

Zhou, Y., Lu, X., Fu, X., Yu, B., Wang, D., Zhao, C., et al. (2018). Development of a fast and sensitive method for measuring multiple neonicotinoid insecticide residues in soil and the application in parks and residential areas. *Anal. Chim. Acta* 1016, 19–28. doi:10.1016/j.aca.2018.02.047

Zhou, Y., Lu, X., Yu, B., Wang, D., Zhao, C., Yang, Q., et al. (2021). Comparison of neonicotinoid residues in soils of different land use types. *Sci. Total Environ.* 782, 146803. doi:10.1016/j.scitotenv.2021.146803

Zhou, Y., Zha, J., and Wang, Z. (2012). Occurrence and fate of steroid estrogens in the largest wastewater treatment plant in Beijing, China. *Environ. Monit. Assess.* 184, 6799–6813. doi:10.1007/s10661-011-2459-y

Zhu, Q., Jia, J., Wang, Y., Zhang, K., Zhang, H., Liao, C., et al. (2019). Spatial distribution of parabens, triclocarban, triclosan, bisphenols, and tetrabromobisphenol A and its alternatives in municipal sewage sludges in China. *Sci. Total Environ.* 679, 61–69. doi:10.1016/j.scitotenv.2019.05.059

Zorpas, A., and Inglezakis, V. (2012). *Sewage sludge management: from the past to our century*. New York: Nova Science Publishers, Inc., 511.