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Understanding the extent of emerging contaminants in english soils: Environmental implications of differing organic waste applications

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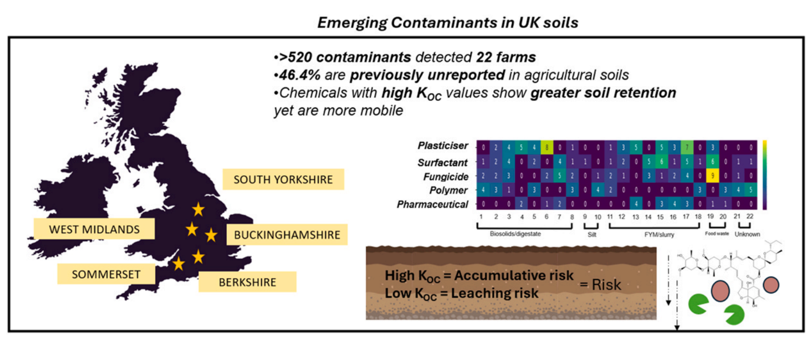
HIGHLIGHTS

- Over 520 contaminants detected in English soils; including > 190 unique entities.
- Plasticisers, pharmaceuticals, polymers, fungicides, surfactants frequently detected.
- 46.4 % of pharmaceuticals had not been identified in previous monitoring campaigns.
- Atrazine and atraton presence confirms persistence in soils despite restricted use.
- Organic content of soils positively correlated to contaminant presence.

GRAPHICAL ABSTRACT



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ABSTRACT

The application of differing organic fertilisers to agricultural land is a long-standing practice that supports sustainable nutrient recycling. Despite the widespread use of organic amendments, the occurrence, distribution and fate of Emerging Contaminants (ECs), within agricultural soils remains poorly understood. To address this knowledge-gap, this study presents a comprehensive assessment of ECs across 22 English farms with diverse amendment histories and soil types. We evaluated and developed both a harmonised in-field sampling strategy alongside targeted and non-targeted mass spectrometry approaches, to reveal the presence of a wide range of ECs in soils. The antiparasitic ivermectin had the highest reported concentrations (21.8 ± 7.3–105.9 ± 86.7 ng/g

Abbreviations: POPs, Persistent Organic Pollutants; PAHs, Poly Aromatic Hydrocarbons; PCBs, Polychlorinated Biphenyl's; ECs, Emerging Contaminants; LOD, Limit of Detection; LOQ, Limit of Quantification; QC, Quality Control; OC, Organic Carbon; CEC, Cation Exchange Capacity; PPP, Plant Protection Products; NSAID, Non-Steroidal Anti-Inflammatory; PCA, Principal Component Analysis; NMDS, Non-metric Multidimensional Scaling; DT_{50}, Half-life; MAE, Mean Absolute Error; NTS, Non Target Screening.

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(dw)), followed by the antibiotics oxytetracycline, ofloxacin, enrofloxacin, and plant protection products atrazine, and diazinon. Non-target screening identified 524 chemical entities, 194 were singular occurrences. Prevalent contaminant classes included pharmaceuticals, plasticisers, polymers, fungicides and surfactants; > 40 % of these had not been previously detected in English soils. Dominant pharmaceuticals included antibiotics (n = 9), steroids (n = 4), anticancer (n = 3), and antipsychotic metabolites (n = 3). Here we present a feasible, and accurate approach to soil sampling – analyses which reflects accurate concentrations in field, in addition to the wide-spread occurrence of ECs in English agricultural soils receiving an array of organic fertilisers.

1. Introduction

Soil quality underpins agricultural productivity, ecosystem resilience, and ultimately human wellbeing [120,7]. Historically, agricultural soils have been monitored for legacy contaminants such as heavy metals, nutrients, and persistent organic pollutants (POPs). These substances have relatively well-established analytical methods, known persistence in the environment and well characterised toxicology profiles [100,137]. As such traditional pollutants, including lead, cadmium, arsenic, mercury, as well as poly aromatic hydrocarbons (PAHs), and polychlorinated biphenyl (PCBs), have been the focus of regulatory frameworks and environmental risk assessments [154]. However, the intensification of agricultural practices and evolving environmental pressures (e.g., the need to reduce synthetic fertiliser production and use [66,75]), has led to an increase in the use of organic waste amendments (e.g., farm yard manures, biosolids, digestates), that have the potential to introduce a new suite of chemical stressors, emerging contaminants (ECs), into the soil environment [19]. These amendments, while beneficial for soil fertility and structure, can introduce a wide range of chemical residues, including pharmaceuticals, personal care products, pesticides, and industrial chemicals. ECs are not routinely monitored meaning there is a marked deficiency in data on the presence and distribution of ECs in agricultural soils [73,90], despite the significant environmental and human health risk they may pose [19].

Where monitoring data does exist, this has largely focused on the presence of ECs in the United States and Australia [23,71]. Occurrence and concentrations of ECs in UK soils remain poorly characterised despite the fact that a diverse spectrum of organic waste amendments are routinely applied across the UK's agricultural landscape, ranging from livestock manures, treated sewage sludge (biosolids), wastewater treatment residuals and green waste composts [5,137,138,146]. This knowledge gap is concerning given that farming systems, amendment types, amendment treatments/practices and environmental conditions differ between regions, which can influence contaminant fate and bioavailability [104,13,15]. [32] the UK's Circular Economy Package set the aim to achieve a 65 % reduction in municipal waste by 2035 [28]. As part of this initiative, wastewater treatment companies are incentivised to reuse biosolids, aligning with the principles of a circular economy. This lack of knowledge is critical, as persistent contaminants can accumulate over time, potentially affecting soil microbial communities, crop health, and food safety, while also posing risks to groundwater and wider ecosystems [44,65,112,135]. Whilst our understanding of chemical presence and fate is evolving; our evaluations of risk are consistently behind that of chemical production and use. Furthermore, there is a clear need to understand how ECs, in particular, can persist in natural systems following repeated applications of organic waste amendment [151].

The importance of addressing this knowledge gap is underscored by technological advances in analytical chemistry, particularly mass spectrometry (MS), techniques. Modern MS methods, including high-resolution and tandem mass spectrometry, enable the detection and quantification of a much broader array of chemical compounds at trace levels than was previously possible [86]. These advances facilitate comprehensive chemical profiling of soils and amendments, providing new insights into the chemical landscape shaped by contemporary agricultural practices. Currently, regulation surrounding ECs in agricultural soils is non-existent in the UK. The introduction of regulatory frameworks to safeguard soil quality and food safety will require robust, validated analytical methods and monitoring strategies. Without baseline data and standardised approaches, risk evaluations will be obsolete thus hindering policymaking. Leading to the potential risks associated with ECs to go unrecognised or unmanaged. Moreover, monitoring studies consistently overlook key analytical components such as sampling strategy and effects of sample transport which ultimately result in a poor understanding of absolute concentrations in the field.

A comprehensive evaluation of the persistence of ECs with known ecological and human health concerns under real crop production conditions is urgently needed. This study aimed to; 1) critically assess and devise a harmonised approach of soil monitoring through to analysis (targeted and untargeted), in soils; and 2) determine the presence of ECs in English agricultural soils with diverse organic amendment histories. To achieve this a multi-residue extraction method was developed targeting 18 analytes that consisted of a wide range of physicochemical properties and optimised for their extraction from soils with varying properties. Liquid chromatography-tandem mass spectrometry (LC-MS/MS), and liquid chromatography-tandem high-resolution mass spectrometry (HRMS), techniques were employed to capture a broad spectrum of contaminants, enabling a comprehensive assessment of EC presence in soils. To the best of our knowledge, this is the first study to report a wide analysis of ECs in English agricultural soils.

Ascertaining representative soils across farming landscapes remains a challenge for environmental scientists; controversial goals between production/farming and environmental research (i.e., farming, academia, and regulatory bodies), continuously stunts our abilities to assess chemical risks towards soils in the environment. To address this, the study adopted a citizen science approach, engaging directly with farmers to raise awareness of the issue associated with chemical contamination in soils. This approach underscores the importance of such studies for improving soil health and advancing our current evaluation of chemical risks. Our approach fosters stakeholder engagement, raising awareness of soil contamination issues and promoting a shared responsibility for environmental stewardship.

2. Materials and methods

2.1. Soil sampling and site descriptions and chemicals

Twenty-three farms with contrasting farming practices and application histories were sampled across England; SI Table 1 contains the application histories for the selected farms. In brief, the farm application histories were comprised of, FYM/slurry (n = 9), biosolids (n = 8), food digestate (n = 3), Wastewater Treatment Residuals (n = 2), and unknown (n = 2). Fourteen farms were sampled using a W - transect sampling method following site visits (as described below), and samples from eight farms were shipped to the University of Leeds (1 sample per field). Soil samples were sampled to 0–10 cm depth using a trowel and stored in zip-lock bags and cool boxes containing ice packs (≤ 8 h), before storage at −20 °C. Participating farmers were given sampling instructions detailing how to collect soil samples and minimise cross-contamination. For farms sampled using the W - transect method, triplicate samples (S1–3), were taken at each sampling point which were combined to make a composite sample (P1), each field therefore consisted of five composite samples (P1–5; Fig. 1).

To evaluate in-field variability within the CECs concentration profile, the W – transect sampling methodology was assessed in one field at farm 17. In Method 1, three samples were obtained at each point of the W - transect (n = 5), and were compared to Method 2, where soil samples at each point of the W were homogenised and combined (n = 15). At each field, hedgerow samples were obtained to compare to soil samples not in receipt of direct amendment. Shipment samples were received in mailing packages which contained a thermal lined envelope and ice pack, farmers were requested to obtain a hedgerow sample and field sample from the middle of the field. In addition, nine control samples were obtained from differing sites with no known history of previous contamination, these contained a Site of Special Scientific Interest and a Sustainability Garden with no known use of soil amendments, and hedgerow samples from a farm adhering to good veterinary and husbandry farming practices which only applies manure from their livestock and has reduced use of veterinary medicines (lower chemical usage).

A priority list of ECs was identified for analysis using targeted methods owing to their known persistence in soils, previous detection in global monitoring campaigns and high usage in agricultural settings. A detailed list of chemicals showcasing the wide variety of physicochemical properties is provided in Table1. All chemicals purchased were of the highest available purity (≥ 98 %), (Table1; SI Section A Text 1.0).

2.2. *Emerging contaminant soil extraction and solid phase extraction clean up*

For each sample 2.5 ± 0.2 g of freeze-dried (sieved < 2 mm), soil was extracted using a three-stage solid-liquid sequential extraction, comprising of; *1.* acetone *2.* McIlvaine buffer –methanol (50:50 v/v), *3.* 5 mM phosphoric acid in acetonitrile [10]. Prior to extraction Internal standards (atrazine-d5, clotrimazole-d5, cyclophosphamide-d4, diazinon-d10, diclofenac-d4, enrofloxacin-d5, lincomycin-d4, ofloxacin-d8, robenidine hydrochloride-d8, sulphamethoxazole-d4, and oxytetracycline-d6), were spiked into soils at 30 ng/g to correct for losses during the extraction and analyses. All stages included shaking at 250 rpm (16 h for acetone and 30 min for extraction buffer/solvent 2, and 3), vortexing (30 s), sonication – ten minutes (ambient temperature) followed by centrifugation at 2500 rpm prior to decanting and collecting supernatants (Fig.1). Between each extraction step supernatants were refrigerated at 4 °C prior to combining and diluting to 5 % solvent using deionised water. Extracts underwent Solid Phase Extraction (SPE), using OASIS HLB (500 mg, 6cc Waters Elstree), please see SI Section A text 1.1 for further details of the SPE procedure. When ≥ 1 day of storage time was required, samples were stored at –20 °C with freshly prepared calibration standards prior to analysis.

2.3. *Targeted EC analyses and data processing*

Targeted quantification was achieved using a Thermo Scientific Quantiva Mass Spectrometer in tandem with a Thermo Scientific Vanquish Liquid Chromatography system (LC-MS/MS). In brief a reversed-phase chromatographic methodology was employed that utilised a Waters Acquity (HST3, 100 Å, C18 -silica, 1.8 μm particle size, 2.1 ×100 mm), column with a Vanguard pre column filter (100 Å, 1.8 μm, 2.1 mm×5 mm), which utilised positive – negative polarity switching, mobile phases of 0.1 % formic acid, and 5 mM ammonium formate in water and acetonitrile – methanol (50:50), with a flowrate of 0.3 mL/min. Data processing was achieved using Xcalibur V4.3; 10-point calibration curve, internal standards, QC recovery and matrix suppression, were included for precisions and accuracy in quantification calculations. Matrix suppression was found to be variable and compound specific, suppression ranged between 6.4 – 69.7 % for the assessed ECs. Please see Section A Text 1.2, and SI Text 1.2.1, for full methodological details and optimisation, and SI Table 2, and SI Table 3 for the mass spectrometer parameters for targeted analytes and their associated performance (Limit of Detection/Quantification (LOD (0.000047 – 4.7 μg/kg dw)/LOQ (0.000047 – 4.9 μg/kg dw))), as well as chemical matrix suppression percentages for analyte response in soil

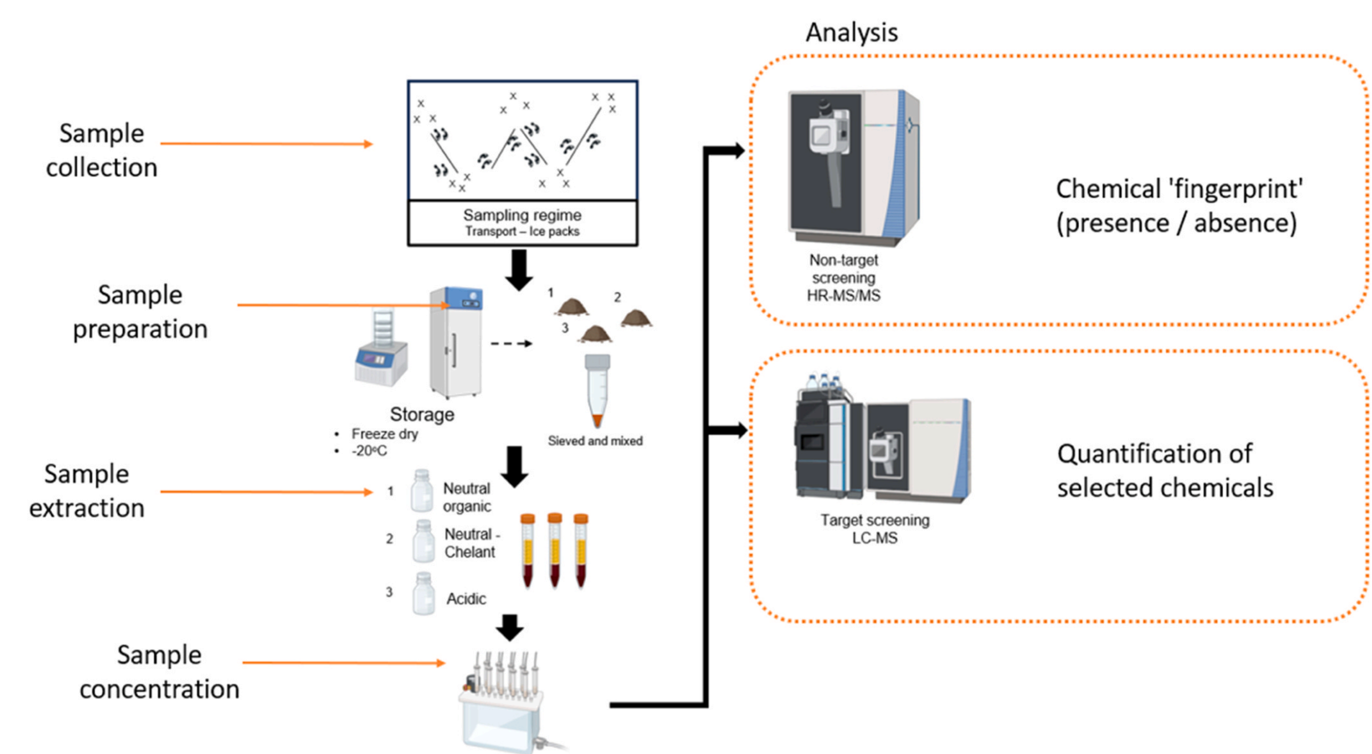

Fig. 1. A schematic diagram illustrating field and laboratory methodologies employed to evaluate EC presence in arable soils.

Table 1
Physicochemical properties and fate parameters of selected target analytes for monitoring and quantification.

Chemical	Pharmaceutical Class	pKa_a	pKa_b	log K_{OW}	K_{OC} (L/kg)	Soil DT_{50} (days)	Biodegradation rate (1/h)
Atrazine	Herbicide	$1.6_{(A)}$		$2.61_{(A)}$	$37–121_{(C)}$	$101_{(D)}$	NA
Carbamazepine	Pharmaceutical – anticonvulsant	$13.9_{(A)}$		$2.77_{(A)}$	$158.48_{(E)}$	$533.2_{(F)}$	$0.26_{(F)}$
Clotrimazole	Antifungal	$4.1_{(A)}$		$6.1_{(A)}$	$57544_{(G)}$	$68_{(H)}$	$0.0213_{(I)}$
Cyclophosphamide	Pharmaceutical – Immunosuppressant	$0.02_{(B)}$		$0.63_{(A)}$	$38_{(J)}$		$0.0372_{(K)}$
Diazinon	Insecticide – Organophosphate	$2.6_{(A)}$		$3.81_{(A)}$	$1493–1589_{(L)}$	$26.6–78.1_{(M)}$	$0–0.056_{(N)}$
Diclofenac	Pharmaceutical – NSAID	$3.99_{(A)}$		$4.51_{(A)}$	$479–956_{(O)}$	$< 5_{(P)}$	$11.79_{(F)}$
Enrofloxacin	Pharmaceutical – Antibiotic	$6.09_{(A)}$	$8.74_{(A)}$	$-0.2_{(A)}$	$987.12_{(Q)}$	$280_{(R)}$	$0.012_{(S)}$
Ivermectin	Veterinary medicine – Anthelmintic				$25800_{(T)}$	$16–67_{(U)}$	
Lamotrigine	Pharmaceutical – Anticonvulsant	$5.7_{(A)}$		$2.5_{(A)}$	$93.13–702.88_{(V)}$	$129–264_{(W)}$	$0.048_{(F)}$
Lincomycin	Pharmaceutical – Antibiotic	$7.6_{(A)}$		$0.2_{(A)}$	$288.3_{(X)}$	$31.35_{(Y)}$	NA
Metformin	Pharmaceutical – Diuretic	$12.4_{(A)}$		$-1.3_{(A)}$	$12–19_{(Z)}$	$5_{(Z)}$	$7.23_{(F)}$
Ofloxacin	Pharmaceutical – Antibiotic	$5.97_{(A)}$	$9.28_{(A)}$	$0.39_{(A)}$	$1657.8_{(AA)}$	$1.1–2.01\ \text{years}_{(AB)}$	$0_{(AC)}$
Oxytetracycline	Pharmaceutical – Antibiotic	$3.27_{(A)}$	$9.5_{(A)}$	$-0.9_{(A)}$	$4102.59_{(AD)}$	$18–28_{(R)}$	$0.00463_{(AE)}$
Robenidine HCl	Veterinary medicine – Coccodiciostat	$3.3_{(AF)}$		$3.8_{(AF)}$	$> 426,580_{(AG)}$		
Sulfamethoxazole	Pharmaceutical – Antibiotic	$1.6_{(A)}$	$5.7_{(A)}$	$0.89_{(A)}$	$94.9_{(AH)}$	$10.81–33.24_{(AI)}$	$0.0289_{(AJ)}$
Triclosan	Personal care product	$7.9_{(A)}$		$4.76_{(A)}$	$12981.75_{(AK)}$	$\sim 9\ \text{months}_{(AL)}$	$0.0578_{(H)}$
Trimethoprim	Pharmaceutical – Antibiotic	$7.12_{(A)}$		$0.91_{(A)}$	$623.8_{(AM)}$	$64.6_{(D)}$	$0.02_{(F)}$
Tylosin	Pharmaceutical – Antibiotic	$7.73_{(A)}$		$1.63_{(A)}$	$623.80_{(AN)}$	$4.5_{(AO)}$	$0.00214_{(AP)}$

Footnote citation key: *A* = *PubChem* [116]*; B* = *DrugBank (NA); C* = Martins et al., [91]*; D* = Blume et al., [11]*;E*=Shao et al., [127]*; F* = Lautz et al., [74]*; G* = Chen et al., [22]*; H* = Sabourin et al., [121]*; I* = Kahle et al., [58]*; J* = Mansouri et al., [89]*; (Opera prediction); K* = Česen et al., [20]*; L* = Nemeth-Konda et al., [101]*; M* = Aggarwal et al., [2]*;N*=Campo et al., [16]*; O* = Yu and Bi, [152]*;P*=Al-Rajab et al., [3]*; Q* = Wu et al., [148]*; R* = Li et al., [80]*; S* = Frade et al., [40]*;T*=Krogh et al., [68]*; U* = *Krogh et al.,* [67]*; V* = Li et al., [77]*; W* = Menacherry et al., [93]*;X*=Wang et al., [141]*; Y* = [84]*;Z*=Mrozik and Stefańska, [98]*; AA* = Straub et al., [134]*; AB* = Yang et al., [151]*; AC* = Kümmerer et al., [72]*; AD* = Jones et al., [57]*; AE* = Li et al., [81]*; AF* = Hansen et al., [51]*; AG* = *EFSA,* [32]*; AH* = Stoob et al., [133]*; AI* = Srinivasan and Sarmah, [132]*; AJ* = Gao et al., [43]*; AK* = Karnjanapiboonwong et al., [61]*; AL* = Wu et al., [149]*; AM* = Zhang et al., [155]*; AN* = Rabølle and Spliid, [118]*; AO* = Carlson and Mabury, [17]*; AP* = Prado et al., [111].

extracts

2.4. Non-Target EC analysis

Non-Target Screening (NTS), was employed on twenty-two farms; Farm 11b was excluded from this assessment as these samples were used to evaluate the sampling techniques. NTS was achieved using a High-Resolution Tandem Mass Spectrometer, Thermo Exploris in tandem with a Vanquish Liquid Chromatography system. Positive ionisation (ESI probe), was employed on a reversed phase chromatographic method that was 15 min in length and had a flowrate of 0.3 mL/min. Gradient conditions differed slightly to that of the targeted; 0.1 % formic acid in acetonitrile was 1 % at 0–0.5 min, 80 % at 0.5–1 min, 99 % at 1–9.5 min, held at 99 % from 9.5 to 11.5 min with a ramp in flow rate to 0.35 mL/minutes, and 1 % at 12, and 15 min. The Vanquish was set to a column pre-heater temperature and temperature of 30 °C, samples were maintained at 3 °C, and injection volume was set to 5 μL. Mass spectrometer temperature settings were 320 °C, and 350 °C for the ion transfer tube and vaporiser temperature respectively. Inert gasses were static with the following arbitrary units, 50, 10, and 1 for sheath, auxiliary, and sweep respectively. Scan range was set to 200 – 1000 *m/z*, RF Lense percentage was 70 %, and the scans/second was set to twenty. In brief the workflow contained MIPS (Multidimensional Ionisation and Partitioning), intensity (10^{E4}), dynamic exclusion, and $ddMS^2$.

2.4.1. Confidence intervals and spectra processing

Spectra processing was achieved using Compound Discoverer (V 3.3), libraries included mzCloud (± 5 ppm *m/z*, RT ± 0.3 min, MS/MS (± 10 ppm), > 70 % confidence, isotopic assessments/calculations, and peak ratings ≥ 5 (Level 2a). For accuracy and precision, the present study reports only those chemical compounds that met the Level 2 identification criteria [124]. This criterion was selected to establish a robust baseline for understanding the presence of ECs in English soils. However, it should be acknowledged that compounds identified at Levels 3–4 also provide indicative chemical fingerprints, and future research should focus on assessing their reliability and occurrence within terrestrial systems. In an additional step to ensure accurate reporting, refined processing was achieved via investigating similarity scores and chemical structure appropriateness. Presence in a sample was recorded following the steps outlined above; for both target and NTS the threshold criteria of presence in 66.7 % of samples and peak areas ≥ 20 % of those within controls. To eliminate the occurrence of false positives controls were processed, peaks were visually inspected. Generally, the criteria were that if a chemical was present in 33.3 % ($\frac{1}{3}$) of the controls then it failed to meet the presence criteria.

2.5. Validation of extraction methods

Validation of the extraction and analysis methods for eighteen prioritised ECs in arable soils was achieved using Lufa Speyer standard soils 5 M, and 6S. Soil properties comprised of 1.18 %, and 1.66 % Organic Carbon (OC), 0.14 %, and 0.18 % nitrogen, pH 7.45, and pH 7.31, CEC 9.86 meq/100 g, and 18.67 meq/100 g respectively. Soil classifications were a sandy loam (5 M) and silty clay (6S), [87]. Three concentrations were assessed in the contrasting soil types (0.3, 3, 30 ng/g (dw)). Soils were spiked and equilibrated for one hour prior to extraction to facilitate an accurate representation of extractability and to equilibrate sorption isotherms [95].

2.6. Validation of sampling methodology and sample shipment

Contaminant concentration was evaluated to assess how representative composite samples are considering the variability within a field. Samples were collected using the W - transect approach with triplicate samples taken at each of the five sampling points for all the fields. However, for one of the fields the concertation of each of the triplicate samples was assessed separately to the composite of each point, and to a whole-field composite sample.

Furthermore, we assessed the impact of soil storage and transport on the stability of ECs. Soil monitoring studies generally include the collection of soils and storage prior to freeze-drying or freezing before analysis takes place. These processes expose ECs to differing temperatures (in transit), and thus stability-dissipation which likely results in an undervaluation of true concentrations in the field. Moreover, in some instances samples may undergo longer shipment times, for example via the post or internationally which likely exposes samples to greater temperature extremes. To evaluate these effects a shorter timeframe (24 h), and a longer timeframe (72 h), were assessed for stability during storage, samples were spiked at 3 ng/g and stored in a thermal mailer (envelope), containing two ice packs and left at room temperature, to mimic that of sample shipments.

2.7. Statistical analysis

Statistical analyses were conducted using Python to investigate associations between chemical properties and contaminant concentrations. Both univariate (Pearson's correlation), and multivariate regression approaches were employed to assess the relationship between chemical properties (e.g., log K_{OW} log K_{OC}, biodegradation rate (1/h – *k*), DT_{50}), and summed quantified concentrations of chemicals detected in soils. To explore broader patterns in contaminant profiles across sites with differing management histories, Principal Component Analysis (PCA), and Non-metric Multidimensional Scaling (NMDS), were applied using chemical class frequency data grouped by application type (e.g., biosolids, manures/slurries, digestates, and Wastewater Treatment Residuals (silt)). Spearman's rank correlation (Spearman's ρ) was used to assess the monotonic relationships between application history categories and chemical class presence. In addition, Kruskal–Wallis tests followed by Tukey-type post hoc comparisons were performed to identify statistically significant differences in contaminant class abundance across different farm types and application regimes. All non-parametric statistical tests were selected due to the non-normal distribution of the data, and significance thresholds were set at $p < 0.05$.

3. Results & discussion

3.1. Method performance and evaluations

3.1.1. Validation of extraction methods

Eighteen ECs were validated across two standardised soil types, and three concentration ranges (0.3 – 30 ng/g (dw)). On average 53.6 ± 21.9 % was recovered from the two soil types at all of the assessed concentration ranges. Recoveries were identified to be compound specific and concentration dependent. For example, at 30 ng/g (dw) recoveries ranged between 1.6 ± 92.9 % –111.2 ± 4.7 % (triclosan–atrazine), whilst at 3 ng/g (dw) and 0.3 ng/g (dw) recoveries were identified to range between 9.6 ± 39.1 % - 125.7 ± 21.6 % (ivermectin – atrazine), and 21.3 ± 49 % - 111.7 ± 17.4 % (lincomycin - lamotrigine), respectively. In some instances, EC recovery was improved at lower concentration ranges (i.e., clotrimazole, cyclophosphamide, lamotrigine, oxytetracycline, trimethoprim, and tylosin), indicating poor retention to the HLB material at the higher concentration ranges (increased competition). As expected soil properties were identified to have an influence on the percentage recovered. As a whole, recoveries from 6S were identified to be lower than that of 5 M; a consequence of the higher OC (1.18 % < 1.66 %). For example, the summed recovery at 30 ng/g (dw) for the assessed ECs was 599.4 ng in the sandy loam, and 572.6 ng in the silty clay loam. Irrespective of this trend a compound specific effect was observed where differences for chemicals with a higher affinity for carbon were recovered at a greater rate in 5 M (sandy loam), over that of 6S (clay loam), including oxytetracycline (72.6 % vs 31.9 %), lamotrigine (84.6 % vs 65.2 %), and ivermectin (19.6 % vs 4.8 %). Conversely higher recoveries in the 5 M soil were observed for less polar chemicals such as diclofenac (78.6 % vs 56.13 %), sulphamethoxazole (74.5 % vs 45.3 %), trimethoprim (98.6 % vs 41.8 %), and carbamazepine (80.62 % vs 61.7 %), (SI Figure 1). Some analytes were unaffected by soil properties demonstrating consistent recoveries across the two soil types (atrazine, metformin, cyclophosphamide, and clotrimazole). Compound specific relationships are to be expected in any extraction methodology or environmental matrix and reiterate the requirement to assess recoveries over a range of soils.

Our results also revealed the extent to which concentration affects analyte recovery with competition for binding sites and analyte suppression hypothesised to result in the lower recoveries observed for lower soil concentrations [96], (SI Figure 1). For example, diclofenac, robenidine hydrochloride, triclosan, oxytetracycline, enrofloxacin, atrazine, and ivermectin were below the limits of quantification or detection at 0.3 ng/g (dw) for both soil types. Atrazine, carbamazepine, diclofenac, lamotrigine, and trimethoprim were all easily extracted from the contrasting soil types and analytical recoveries for these analytes met the criteria set out in SANCO 3029 [36]. While other analytes did not meet the specified ≤ 70 % threshold it is important to acknowledge the method developed was a multi-residue extraction method assessing true recovery (spiked samples were aged and not extracted immediately), for a range of chemicals with diverse physio-chemical properties (e.g. log p −1.3–6.1, pKa 0.02–13.9). Care should be taken when interpreting the reported concentrations of lincomycin, ofloxacin, and ivermectin; these CECs were associated with poor recoveries and therefore quantified concentrations could be lower than that of reality. Approximately, 40 % of recoveries were > 60 % with additional method development results are provided in SI Figure 2, and SI Table.4. For quantification linearity ranges were identified to range between R^2 0.96 and 0.99 for all targeted chemicals.

3.1.2. Evaluation of infield sampling strategy

Spatial chemical presence was assessed across a manure amended field (Farm 11b), to investigate variability within the W-sampling methodology and the suitability of composite samples for ECs in soils (Fig. 2, SI Table. 5). Sample points (P1–5), referred to the composition of triplicate samples (S1–3, Method 2), taken at each point of the transect. Results confirmed the presence of a range of ECs including veterinary medicines and agricultural chemicals (clotrimazole, diazinon, diclofenac, enrofloxacin, oxytetracycline, and tylosin). Chemical presence and concentration were unevenly distributed throughout the field. For example, oxytetracycline, and tylosin were identified at only 40 % of the sampling points (P1–5), while clotrimazole, diazinon, diclofenac and enrofloxacin were detected at 60 %, 60 %, 80 %, and 100 % of sampling points (P1–5), respectively. Results also revealed variability in the detection of select ECs among the triplicate samples (S 1–3), collected at each point. For example, at P1 (S1–3), clotrimazole and diazinon were only identified in 1/3 of samples. However, this was not the case for the majority of analytes and thus differences were not considered to be of significance ($p = 0.65$–1 Wilcoxon), (SI Table. 6). The Mean Absolute Error (MAE), ranged between 0.003 and 6.9 when quantified concentrations in mixed samples (S1 P1–15) were compared to the averaged replicate concentrations. Larger errors were observed for some analytes such as diclofenac and oxytetracycline. Diclofenac showed MAEs of 11.5, and 6.9 at S1–3 (P1), and S1–15 (P1–5), respectively, while oxytetracycline recorded MAEs of 1.18, and 0.9 at S10–12, and S13–15 when comparing mixed triplicates and quantified averages. Composite sampling provides a pragmatic approach to assessing environmental contamination (cost and time effective). However, it may overlook true in-field concentrations and result in reported concentrations below that of reality, a prefect of intra-field variability. The presented evaluation indicates the potential for the under or overestimation of absolute concentrations in – field, as well it demonstrates that our current procedures for estimating ECs in soils is unrealistic (uniform vs non-uniform), [34]. Such a finding is unsurprising and has been identified for other pollutants such as micro plastics [140]. Thus, care should be taken when considering in-field quantification data when evaluating environmental risks, to address this future research/monitoring efforts should encompass a comprehensive sampling technique and evaluate the cost – time vs data reliability. Therefore, the most feasible approach is to adopt a W – transect approach and homogenise repeat samples at specific sampling points.

3.1.3. Validation of sample shipment

In all variants of sample shipment evaluated, chemical recovery was affected. After 24 hrs all ECs were recovered from soil 5 M (sandy loam), whereas at 72 hrs, diclofenac, and ivermectin were below detectable

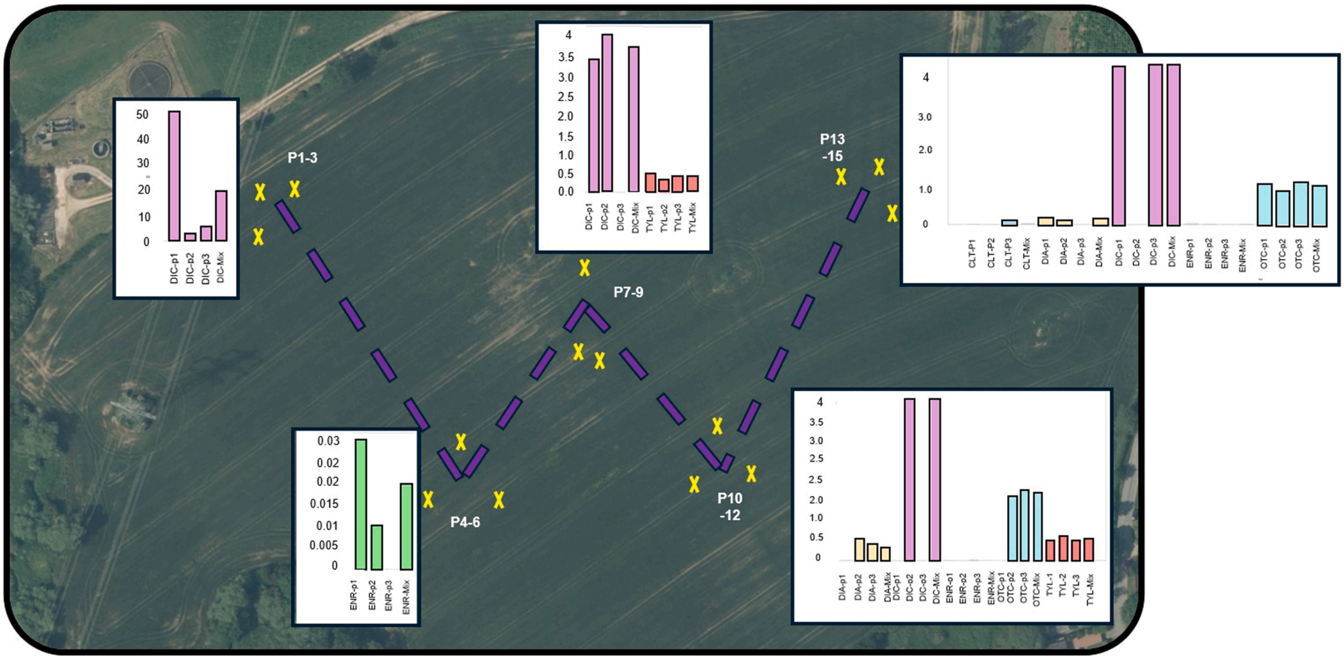

Fig. 2. Results from an assessment of variability in EC concentrations across samples collected as part of a W-sampling strategy in a manure amended field. The table refers to the concentration of contaminants detected at each sampling point in mixtures with a comparison between a combined composite sample (S1–5, P1–15), and the average of quantified composites for S1, S2, S3, S4 S5. CLT, DIA, DIC, ENR, OTC, and TYL refer to clotrimazole, diazinon, diclofenac, enrofloxaicn, oxytetracycline, and tylosin respectively. Abbreviated forms were required for visual purposes.

limits. At 24 hrs recoveries were identified to be on average 33.6 % lower than calculated after 1 hr indicating a substantial loss of parent analyte during the shipment of samples. At 48 hrs the effect was exacerbated with recoveries being on average 71.7 % lower than that of the 1 hr assessment (Figs. 3), 55.7% of the decline in recovery was associated between 24 hrs, and 72 hrs. However, for cyclophosphamide, metformin, and to a lesser extent clotrimazole, recoveries were found to be comparable between the 24 hr and 72 hr time points demonstrating a compound specific relationship which is suspected to be driven via biphasic degradation patterns (Fig. 3).

3.1.4. *Evaluating soil sampling methodologies*

Method performance was critically assessed and included variability within in-field samples, recoveries over 1-hour timeframes (true

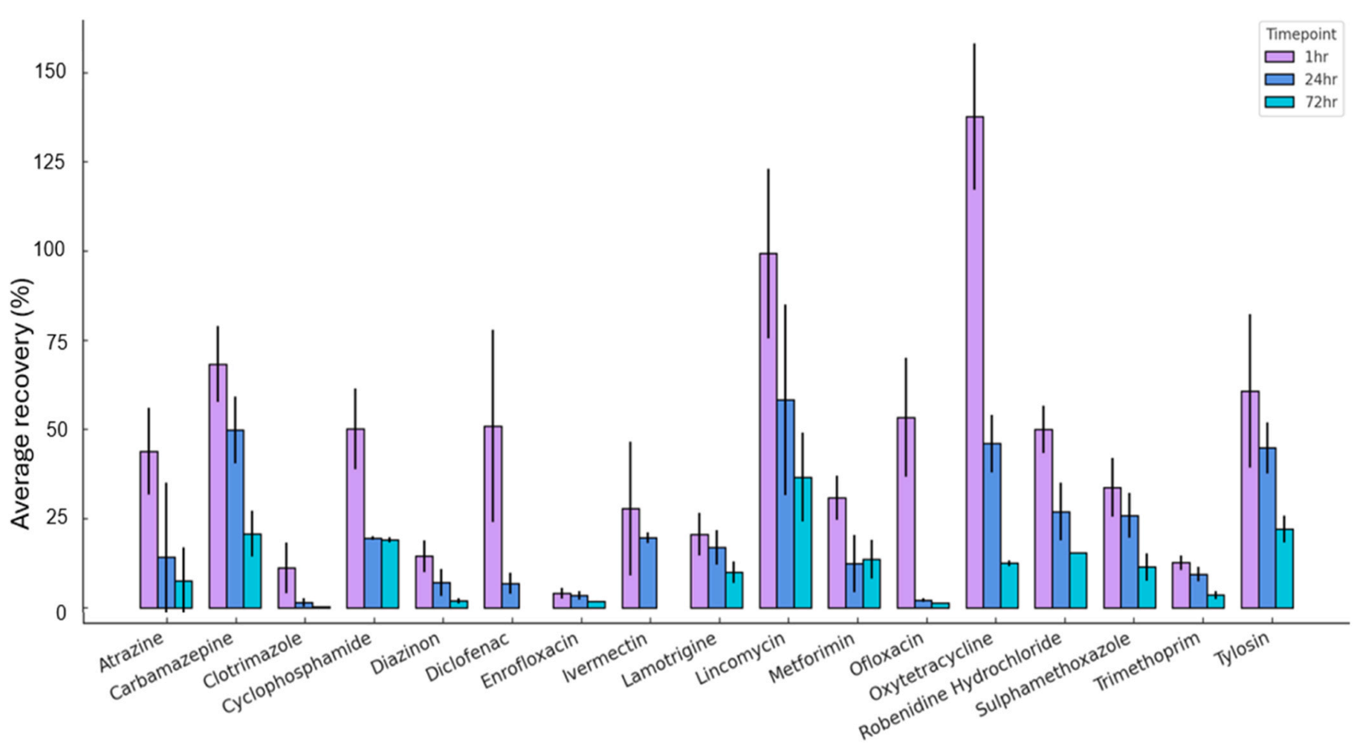

Fig. 3. Average recoveries for target chemicals during the simulated shipment sampling assessment (1-hr to 72-hr).

recovery), shipment timeframes (24 hr, and 48 hr), as well as variability within field samples. The data demonstrates a pragmatic approach to sampling and extracting samples with a comprehensive evaluation of concentration, therefore demonstrating a means to accurately assess in-field concentrations. Future monitoring campaigns should combine three replicates at each point of the W - transect and steer away from field composition, should it be economically and practically feasible. Moreover, stability during shipment should be addressed and corrected. Our results suggest that by overlooking the effect of sample ageing, the reported presence of ECs in soil could be misleading and underestimate the true concentrations present in the environment; within just a 72-hr time window of shipping samples an error of 33.6 % was identified. The data demonstrates poorer analyte recovery with longer shipment timeframes, however further assessments with accurate temperature recordings are required. As well future research should investigate losses during shipping samples in varying regions. It is essential that future monitoring efforts should incorporate field quality controls or laboratory assessments to address potential losses during transit.

3.2. Chemical presence in arable soils

3.2.1. Targeted analyses

Fig. 4, and SI Table 7 present the concentration ranges of Level-1-identified and quantified contaminants detected in English agricultural soils [124]. Of the eighteen targeted chemicals, 83 % were detected across the 22 farms. Pharmaceuticals–primarily antibiotics ($n = 6$)-comprised the largest chemical class ($n = 13$), originating from both human and veterinary use. Measured concentrations ranged from 0.017 ± 0.007–61.32 ± 65.72 ng/g (dw), consistent with previous findings in agricultural soils [139,4]. Only three contaminants—oxytetracycline, ofloxacin, and ivermectin—were found at concentrations exceeding 40 ng/g (dry weight). This trend likely reflects their strong affinity for organic carbon/matter (OC/OM; high K_d/K_f values), and environmental persistence, suggesting limited mobility in soil and potential accumulation from repeated manure or biosolid applications [129,64]. Oxytetracycline was found at farms using animal manure and farms which had applied biosolids (i.e., Farm 1, 3, and 6 (biosolids/digestate), and Farms 13, and 16 were farmyard manure or slurry).

Despite differences in sampling timeframes following application/use and the lower affinity of some compounds for OC/OM, our results clearly show that soils retained sulphamethoxazole, carbamazepine, and lamotrigine, compounds that are typically considered mobile due to their potential for leaching, runoff, or plant uptake [97]. Carbamazepine and lamotrigine, both anticonvulsants, are known for their high persistence in soils, with reported DT_{50} values of 533.2 days, and 129–264 days, respectively [74,93]. However, their relatively low concentrations (< 3 ng/g dry weight), in this study may be explained by additional removal processes ([109,93,143]). These compounds are frequently monitored in studies examining soil and plant accumulation following wastewater irrigation or biosolid application [128,97], but until now, have not been reported in English agricultural soils (Table 2, and SI Table 8).

In general, reported concentrations were comparable to those reported in previously published literature in different regions (Table 2, and SI Table 8), some variation is expected (i.e., carbamazepine and sulphamethoxazole), owing to differences in agricultural practices, soil types, exposure/use rates, as well as differences between sampling and timing of application of the organic amendment. For example, quantified concentration ranges in English soils were identified to range from 0.004 to 425.5 ng/g (dw), whereas the concentrations that were reported in the literature ranged between 0.01 and 7900 ng/g (dw), (Table 2). Of the available, 80 % of the chemicals were within a factor of ten with at least one other region from the literature. Moreover, concentrations typically range between 1.0 and 10,000 ng/g for biosolid amended soils, and 0.1-< 30 ng/g for manure amended soils [108,113, 50,59]. As shown in Fig. 4, the presented minimum and maximum data falls within these windows and demonstrates a compound specific relationship between application type and concentration. However, clearly more research is required to assess the concentrations of human and veterinary pharmaceuticals in arable soils receiving organic fertilisers [114].

Ivermectin was identified at elevated concentrations in 18 % of farms receiving manure amendment, concentrations ranged between 21.8 and 105.9 ng/g dw (on average); a precursor of its high excretion rate 80–98 %, immobility-accumulative nature, use and persistence (DT_{90} >396 d in dung-soil), ([1,145,82,18]). However, despite these factors previous monitoring campaigns have largely failed to quantify its presence in soils [1,145], a problem associated with poor extraction recoveries and ionisation via MS techniques. For example, ivermectin was identified in 65/105 soil samples but was not quantified (Carillo *et al.,* 2024). Nevertheless, where data is available concentrations reported in this study are comparable (e.g. 18.4–86.6 ng/g (dw) in field soils receiving 300 ng/g of dosed faeces [56]).

Multivariate statistical analyses were carried out to identify whether there was a relationship between physicochemical properties and summed concentration (of all assessed fields - ng/g (dw)). When all parameters were considered, this resulted in a R^2 value of 0.949, however the model was not significant ($p = 0.099$), due to a small sample size ($n = 7$), and low degrees of freedom ($df = 2$), statistical power was constrained. Despite K_{OC} and log p being indicators for mobility-bioavailability in soils, research has seldom assessed the relationship between these parameters and concentration. Results from our targeted analysis reveal a significant positive correlation between K_{OC}, log p, and the summed concentration for the target analytes ($p = 0.013$ (R^2 – 0.4, confidence interval – 0.17, 1.13, effect size – 0.39), p=0.072 (R^2 – 0.4, confidence interval – 0.09, 2.75, effect size – 0.36), respectively), with a higher K_{OC} resulting in a higher EC concentration (SI Figure 3, SI Table 9). This finding suggests that chemical concentration is a factor of immobility and poor removal mechanisms when bound to OM/OC residues. Clotrimazole was however an outlier, with a known high affinity for soils (K_{OC} of 57544 L/kg), and relatively long half-life in biosolid amended soils (DT_{50} 365 days), [22], but the summed concentration across all farms was only 37 ng/g (dw), (Fig. 4). As clotrimazole has been previously detected in UK biosolids (1.32 mg/kg (dw)), [90], its presence in English soils is not surprising. However, the lower observed concentration range is likely related towards lower application histories in comparison to veterinary medicines as well as the formation of non-extractable residues [33].

Nevertheless, the identification of this correlation supports the notion that K_{OC} can be used as a reasonably good indicator of chemical fate in the context of chemical regulation, prioritisation, and risk assessment for soils however this should be used with caution, given potential outliers as evidenced by the example of clotrimazole. Additional analysis of three farms which underwent comprehensive soil property analysis (Farm 8, 9, and 11), further confirmed the influence of soil properties on EC concentration. Results revealed that out of pH, OC, cation exchange capacity, and texture, pH had the strongest correlation with the concentration of targeted analytes ($p = 0.32$). Specific contaminants contributing towards this correlation were enrofloxacin (Spearman's $\rho = -0.79$, $p = 0.002$), clotrimazole ($\rho = -0.59$, $p = 0.073$), and diclofenac ($\rho = -0.58$, $p = 0.064$). These contaminants were identified to have higher concentrations in soils with a lower pH; enrofloxacin has pKa values of 6.09, and 8.74 for pKa_a and pKa_b respectively and thus will behave as a cation at pH values < 6.5, which results in a greater attenuation and accumulation in soils with a lower pH. Clotrimazole and diclofenac deviate from this, their higher accumulation could be driven via application histories ($p = 0.035$, $H = 8.63$; Kruskal-Wallis, $n = 3$ farms), or through altered soil microbial population dynamics (Xiaqiang *et al.,* 2008, [119]).

Table 2
Reported concentration ranges of target ECs in 23 English soils, compared with values reported in the global literature.

Chemical	English Soils - MEC (µg/kg dw)	Location	Concentration (µg/kg dw)	Amendment	Citation
Atrazine		Mexico	360–11640	Wastewater irrigated & herbicide application	Salazar-Ledesma et al., [122]
		Germany	0.01–0.2	Organic fertiliser (21 years following the ban)	Vonberg et al., [138]
	24.5 ± 0.3	USA	74.62–5447.8	Organic fertiliser (1–180 days after application	Selim, [126]
		USA	66–81	Biosolids	Wu et al., [146]
	0.001–12,8 ± 0.38	Canada	30	Biosolids	Gottschall et al., [47]
		USA	93.1	Biosolids	Holling et al., [53]
Carbamazepine		Spain	47–493	Biosolids	Malvar et al., [88]
		Denmark	3.64–61.8	Biosolids (six years)	Holling et al., [53]
Clotrimazole	0.03–4.84 ± 2.33	China	2.2–41	Biosolids	Chen et al., [22]
Cyclophosphamde	0.84 ± 0.1–34.82 ± 58.89				
Diazinon	0.08–13.89 ± 0.51	Costa Rica	20–7900	Insecticide	Natal-da-Luz et al., [99]
		Japan	13–49	Rice Paddy Soil - Sprayed with insecticide	Ghassempour et al., [45]
Diclofenac		Spain	72–363	Biosolids	Malvar et al., [88]
	0.74	Africa	0.6–0.9	Soils	Otoo et al., [107]
Enrofloxacin		Czech Republic	3.04–36.3	Poultry manure	Fučík et al., [42]
		Malaysia	36–647	Broiler manure	Ho et al., [52]
			bdl	Biosolids (dewatered)	Sabourin et al., [123]
		France	14–518	Poultry Manure	Pourcher et al., [110]
		Catalonia	151	Pig slurry	Gros et al., [49]
		USA	19.6	Soil	Kim et al., [63]
		Brazil	22.93	Soil	Leal et al., [76]
	0.02–13.13 ± 6.99	Turkey	50	Soil	[60]
		Brazil	0.39–30.97	Soil	Leal et al., [76]
Ivermectin	21.8 ± 7.3–105.9 ± 86.7	Argentina	1.58 ± 0.9–17.1 ± 13.9	Sediment	Mesa et al., [94]
Lamotrigine	0.03 ± 0.005–33.9 ± 36.14				
Lincomycin	< 0.047–0.21	China	0.97		Li et al., [79]
		Canada	46.3–117	Manure	Kuchta et al., [69]
		China	7.3–900	Compost	Wang et al., [142]
Ofloxacin	38.56 ± 14.9				
Oxytetracycline		Italy	4–6	Cow manure	De Liguoro et al., [26]
		Catalonia	75	Pig slurry	Gros et al., [49]
			2500–50000		Loke et al., [85]
		China	33–33.5	Pig manure	Wei et al.,. [144]
		China	32.1–3676	Cow manure	Wei et al.,. [144]
		China	44.8–3511	Poultry manure	Wei et al.,. [144]
	0.17 ± 0.01–112.2 ± 94.37	China	797.97	Beef, swine, broiler	Hou et al., [54]
		USA	67.4	Biosolids	Holling et al., [53]
		China	0.5–2.6	Pig manure	Wei et al.,. [144]
		China	1163	Cow manure	Wei et al.,. [144]
			500	Animal manure	Awad et al., [6]
Sulphamethoxazole		China	13.7–1316	Poultry manure	Wei et al.,. [144]
	0.06 ± 0.07–0.1 ± 0.07	China	90.45	Beef, swine, broiler	Hou et al., [54]
Trimethoprim		USA	24.70	Biosolids	Holling et al., [53]
		Malaysia	3–4	Broiler manure	Ho et al., [52]
	0.4 ± 0.04–1.47 ± 2	China	107.42	Beef, swine, broiler	Hou et al., [54]
Tylosin	0.4 ± 0.3–14.1 ± 29.2	USA	bdl	Pig manure	[70]
		USA	bdl	Turkey manure	Kang et al., [59]
		USA	19.6		Kim et al., [63]
		Denmark	24.7		Holling et al., [53]
		Malaysia	6 ± 1–679 ± 492	Broiler manure	Ho et al., [52]
Robenidine Hydrochloride	0.66 ± 0.37–1.05 ± 0.81				

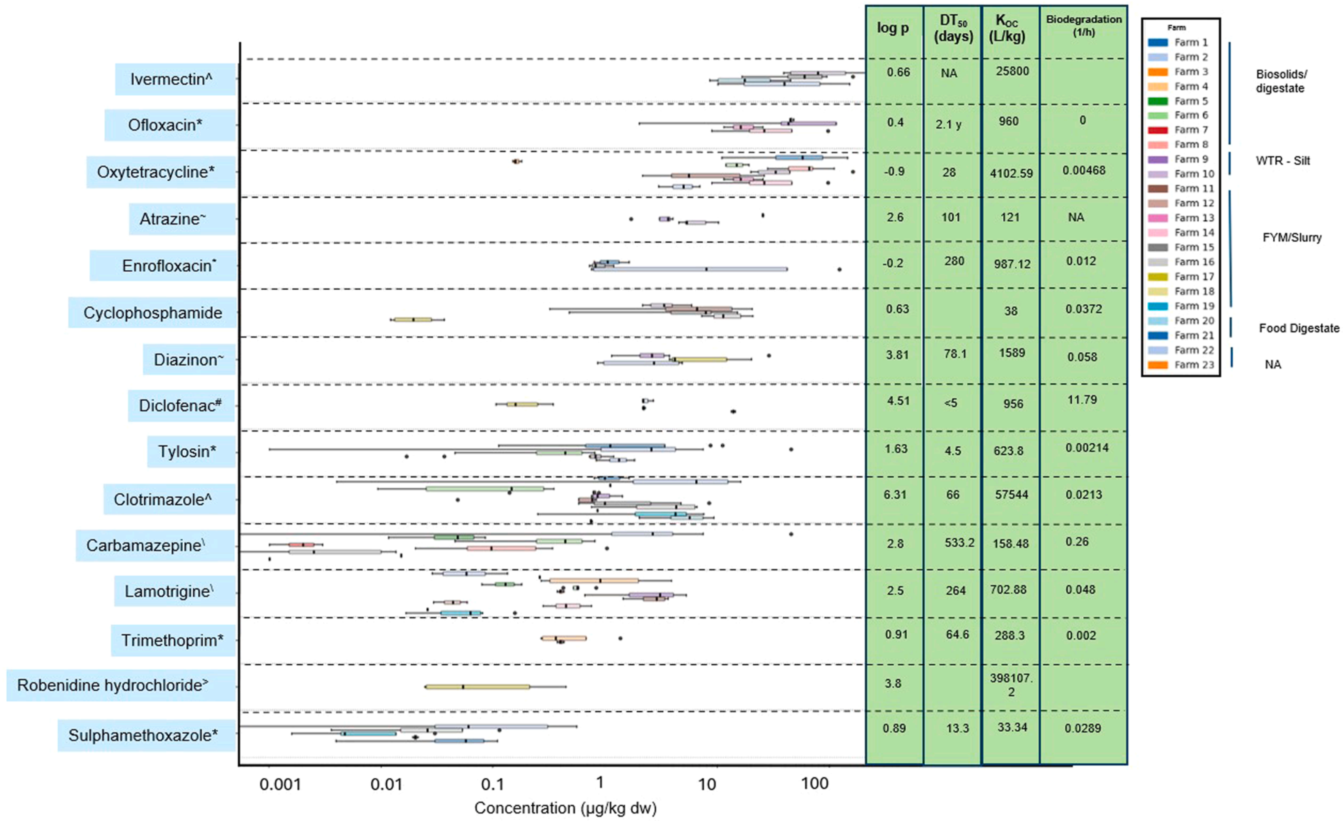

Fig. 4. Quantified concentrations of pharmaceuticals in soil samples collected from farms with differing application histories. Selected physicochemical and fate parameters can be found to the right of this figure, the citations/source can be found in Table 1. Footnote - ^ - Antiparasitic/antifungals, *Antibiotics, ~Plant Protection Products, *Non-Steroidal Anti-inflammatory, \ - Anticonvulsant, > Coccidiostat.

3.3. Non-targeted screening (NTS) for emerging contaminants in assessed soil samples

A total of 523 chemical contaminants were identified in the assessed arable soils, 194 of these were individual entities identified to Level 2 according to the Schmyanski *et al.,* (2014), criteria. Dominant contaminant classes included human pharmaceuticals (n = 52, Rank 1) > plasticisers (n = 51, Rank 2) > surfactants (n = 44, Rank 3) > fungicides (n = 40, Rank 4) > polymers (n = 35, Rank 5) > veterinary medicines (n = 23, Rank 6) > herbicides (n = 20, Rank 7) > human or veterinary use pharmaceuticals (n = 16, Rank 8) > disinfectant (n = 14, Rank 9) > pharmaceutical derivative/research chemical (n = 11, Rank 10). NTS studies have seldom been used to assess EC presence in arable soils; typically, they are associated with water matrices, sediments, or focussed on other chemical classes (PAHs, pesticides) [136]. NTS demonstrated human use chemicals to overarch that of agricultural chemicals in English arable soils, thus demonstrating that future efforts should be tailored towards specifically evaluating their risk towards soil and human health (Figs. 5–6). Where data does exist results include a range of chemical classes and number of detected chemical entities including Gravert et al., [48], who reported 2306 ECs in Danish soils (confidence Level 2–3) and Huang et al., [55], and Qiu et al., [117] where 81 and 405 contaminants were detected in arable soils, respectively. Differences in detections are likely attributed towards varying extractive recoveries, analytical challenges, and regional differences [55].(Fig. 7)

The widespread presence of plasticisers across UK arable soils, results from the fact these contaminants are routinely found in biosolids and animal manures, where they are generally considered persistent with associated half-lives ranging from weeks to decades [9]. Dominant classes observed in this study included phthalates, citrate esters and terephthalate plasticisers (Fig. 5, SI Tables 10–31). Interestingly only one plasticiser was detected on farms that had a history of applying wastewater treatment residuals (Fig. 5). As wastewater treatment residuals (silts) are the by-products of filtration and coagulation/flocculation processes used in drinking water treatment, these processes will likely remove the majority of these ECs, thereby preventing their release into the wider environment following the application of silts to soil [137]. Conversely, silts along with biosolids appear to be a significant source of polyethylene glycol (PEG) which is a synthetic polymer derived from petroleum and ranked third in terms of its abundance of all ECs identified following non-target screening (Fig. 5). This widespread presence across all amendment types was to be expected given the high use of polymers across a range of sectors [130]. However, the low ranking of Plant Protection Products (PPPs), such as herbicides and pesticides was not expected given their known widespread use in agricultural settings (Fig. 5). The highest frequently occurring PPPs were comprised of, flusilazole (n = 6) > flutiafol – epiconazole – isoproturon – fenamiphos – tidemorph (n = 5). Despite the direct application of agricultural chemicals, thereby minimising any potential loss during storage or treatment, our results revealed that there are more chemical entities associated with other EC classes such as plasticisers, pharmaceuticals, and PEGs than that of PPPs.

Despite their widespread occurrence/frequency in the environment [131], PPPs were not ranked the highest in our analyses in terms of detection frequency. Such a finding is likely driven via differences in physicochemical properties and structurally related fate parameters (i.e., sorption – degradation). As PPPs are under regulatory scrutiny, they are inherently synthesised with rapid decay mechanisms in mind which may also be a contributing factor. In the NTS of arable soils, the top seven most frequently detected contaminants across sampled sites were: 1) the sulphonamide sulphamethazine, a commonly used antibiotic in veterinary practices which has been previously monitored in UK arable soils [30], (n = 4); 2) the antidepressant metabolite desmethydoxepin (n = 3); 3) cordycepin (anabolic steroid veterinary n = 5); 4) trenbolone (veterinary medicine, steroid), (n = 3); 5) tiamulin (veterinary medicine, n = 2); 6) testosterone benzoate (n = 2); and 7) minoxidil (n = 2).

To the best of our knowledge the presented monitoring data is the first detection for desmethydoxepin, tiamulin, testosterone benzoate, minodoxidil, and alminoprofen in agricultural soils. Their higher frequency of detections meeting Level 2 criteria indicates a requirement to investigate their persistence and presence further. The occurrence of trenbolone is particularly surprising and requires further investigation as it was identified to have a half-life in arable soils of 0.5–3 days at 25 °C suggesting environmental presence is unlikely [62]. The widespread presence of antibiotics (i.e., tetranactin, tiamulin, roxithromycin, spiramaycin), also warrants further investigation as these biologically active chemicals have been shown to influence soil bacterial community structure and function [78], with implications for nutrient cycling and the degradation of other contaminants as well as being linked to the proliferation of antimicrobial resistance (AMR), [147]. Nordenholt et al., [105], for example identified a 25–50 % reduction in atrazine degradation in the presence of the veterinary antibiotics oxyetracycline and sulphamethoxazole. Literature-reported quantified concentration data for chemicals identified via non-targeted screening (NTS) are provided in SI Table 8 .

3.4. Trends in emerging contaminant presence across different amendment types

Combining identified presence following targeted analysis with NTS results, the data clearly demonstrates the widespread occurrence of multiple classes of ECs from different sources in English arable soils (Figs. 5–6, SI Tables 10–31). The total number of chemical entities detected per farm ranged from 1 to 52. The farms with higher co-contaminant presence per organic application were Farm 19 (n = 37; Food bio digestate), Farm 17 (n = 33), (FYM/slurry), Farm 3 (n = 31), (biosolids or digestate), and Farm 16 (n = 31), (FYM/slurry). Pharmaceuticals were identified as a frequently occurring class of ECs. A total of 93 pharmaceutical detections, originating from human and veterinary sources, were identified in agricultural soils across the sampled farms following NTS and targeted analysis (Fig. 6). Pharmaceuticals comprised of numerous sub classifications, the top-ranking classes at level 1–2a included antibiotics (n = 30) > nervous system medication (n = 15) > antifungals (n = 10) > anticancer agents (n = 6) > antiparasitic (n = 5), (Fig. 6). Interestingly, pharmaceuticals considered as high use by OECD country specific data [106], were not always the pharmaceuticals identified as frequently occurring in this sampling campaign which demonstrates that consumption/administration is not a sole indicator of presence in the environment when considering organic fertiliser applications.

Application history was found to not have a significant effect on chemical profiling in soils (p=0.228). Although trends were observed (as discussed below), a lack of statistical significance was likely hindered by the fact multiple farms had complex application histories consisting of different amendment types. For example, veterinary medicines such as diaveridine, and desmethylclozapine (Farm 6) were identified in fields that had received biosolids/digestate recently. The presence of veterinary medicines can be explained by previous applications of animal slurries or manures, or their presence in wastewater influent originating from the washing of domesticated animals and rearing facilities, [27]. Moreover, Farms 17, and 18 were recorded as FYM/slurry, yet their contaminant profile clearly indicates contaminants from human origins (Figs. 5, and 6). Other contributing factors to the lack of significance could include the complexity and abundance of chemical entities identified within the dataset, as well as the difficulties associated in identifying chemicals to Level 2 confidence. Strict reporting protocols were implemented to facilitate future research on the occurrence of ECs in arable systems. However, as Fig. 2 demonstrates their distribution is not uniform across fields. Consequently, certain compounds of interest may have fallen outside the methodological specifications; these are listed in SI Table 32 and warrant inclusion in future monitoring programmes. Notably, the most frequently detected compounds included amitriptyline, flusilazole, fenpropimorph, prosulfocarb, and atomoxetine.

For fields receiving biosolids and food bio digestate, weak to moderate correlations were observed for food additives and dietary

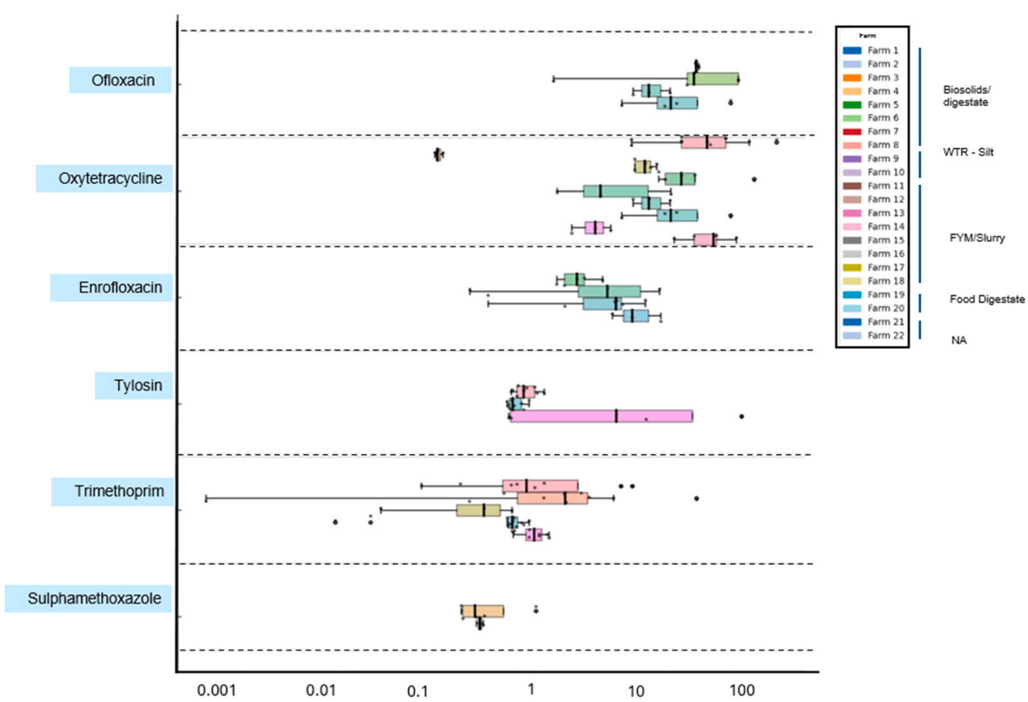

Fig. 5. Quantified concentrations of antibiotics in soil samples collected from farms with differing application histories.

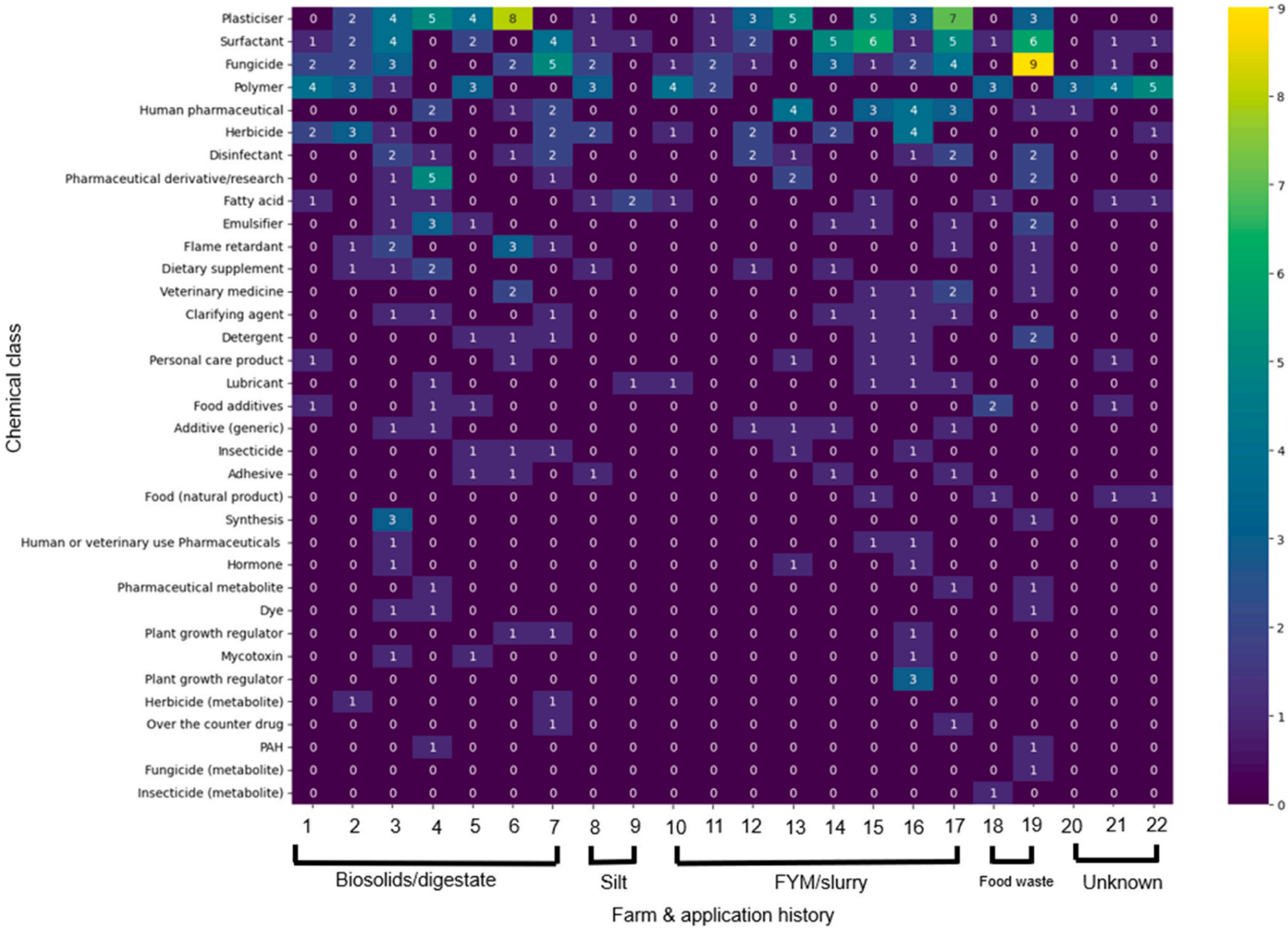

Fig. 6. A heatmap detailing the highest frequency ECs classes in UK arable soils following NTS (Level 2a).

supplements (ρ = 0.23 and 0.13 for biosolids, and ρ = 0.27 and 0.11 for food bio digestate), (SI Figure. 4). Such trends indicate the potential risks arising from everyday consumption of food containing preservatives and dietary supplements (L-carnitine, niacinamide, butylated hydroxytoluene, and sodium benzoate). Whilst these chemicals are contained within food produce, they do pose a wider environmental hazard. For example, the antioxidant butylated hydroxytoluene is persistent, bioaccumulative and available toxicological studies demonstrate it to be toxic to *Danio rerio* [123], with unknown implications for soil dwelling organisms. Moreover, flame retardants were identified to be most prevalent in fields receiving biosolids, a moderate correlation demonstrates persistence and frequency of this chemical classification within wastewater treatment coupled with their higher accumulative nature in arable soils (ρ = 0.49), [21,25]. The most frequently detected flame retardants were Tributyl phosphate, Tri-o-Cresyl phosphate, and Tris(2-ethylhexyl) phosphate in fields receiving biosolid/digestate (SI Tables. 10–31), raising concerns about their potential effects. Available research suggests Tri-o-Cresyl phosphate and its metabolites are cytotoxic to cells mimicking those of a human lung [153], and Tris (2-ethylhexyl) phosphate is bioaccumulative in aquatic organisms [150].

When compared to other application histories, elevated and significant correlations were found for insecticide and fungicide metabolites in food waste digestate - amended soils (ρ = 0.69, $p = 0.05$), supports previous research which has shown that PPPs can either be taken up or transformed in crops where they accumulate [38]. Our results show that PPPs including desnitro-imidacloprid, and prothioconazole-desthio were persistent in soils and these chemicals are considered to present human health risks. For example, prothioconazole-desthio is a metabolite of prothioconazole and has a hazard quotient of 1.3–5.95 % for human consumption [83]. Another significant rank correlation observed in food digestate-amended fields involved PAHs (ρ = 0.45, $p = 0.036$). Their elevated presence in these fields is consistent with their persistence during anaerobic digestion, resistance to degradation mechanisms, and formation of non-extractable residues in soils, as well as their prevalence in organic food wastes [14,29]. Silt-amended soils were generally less contaminated than biosolid and digestate amended soils (Fig. 5) but revealed weak to moderate correlations with adhesives (ρ = 0.21), and lubricants (ρ = 0.16). The moderate positive correlations for these chemical classes in silt-amended fields likely arises from their inherent persistence during wastewater and drinking water treatment processes [125].

In fields treated with animal manures or slurry, weak to moderate correlations were identified for veterinary medicines (ρ = 0.34), human/veterinary use pharmaceuticals (ρ = 0.2). Over-the-counter drugs exhibited weak correlations with fields receiving biosolids (ρ = 0.1), as well as FYM/slurry (ρ = 0.1). Their presence varied across farm types, suggesting that application history, particularly over multiple years also influences EC levels in agricultural soils. Notably, veterinary medicines were identified to be ubiquitous, and their presence in fields which had not received animal manures within 1 year indicates persistence and long-term accumulation. Examples of this include azaperol, diaveridine, and sulphamethazine, which were detected in fields which had received biosolids or food digestate most recently (Farms 6, 19, and 20). These findings indicate that such compounds either persist in soils following previous amendment with animal manures or are also

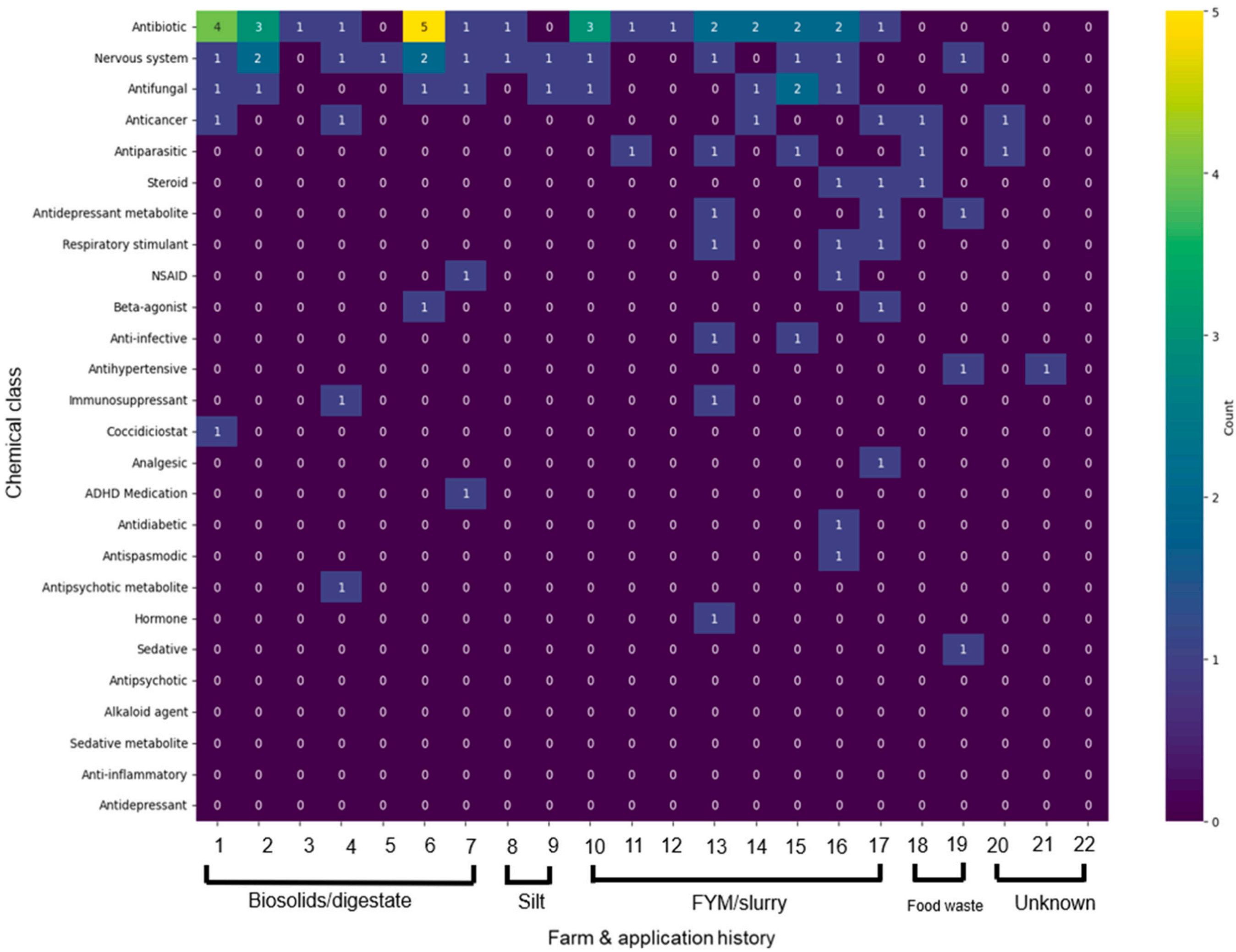

Fig. 7. A heatmap detailing the highest frequency human/veterinary pharmaceuticals in emerging contaminant classes in UK arable soils (Level 1–2a).

introduced through biosolid applications (SI Tables 10–31), [39]. The presented data clearly demonstrates organic fertilisers of varying origins results in the exposure of specific ECs (i.e., flame retardants – biosolids/digestates and silts), however, only moderate correlations were identified. To expand the evaluations that were undertaken here, as well to ascertain a better understanding of ECs presence in fields receiving differing organic fertilisers, future studies should encompass longer-term monitoring assessments to capture EC presence over multiple years of organic waste application.

3.5. Environmental significance

To date, previous soil monitoring efforts have largely focused on identifying the presence of ECs following wastewater irrigation [12,92,97]. Comparatively speaking chemical fate in fields receiving biosolid amendments is more complex than that of wastewater irrigation; biosolid management practices differ regionally (storage timeframes, and heterogenic properties), as well as being influenced via environmental processes more than wastewater. Some of these processes include input timeframes (input vs loss - singular vs daily), as well as biosolids being a heterogenic matrix that changes with time (storage). Thus, both monitoring and modelling efforts are still within their infancy [115,24].

Our targeted analysis and NTS revealed not only the presence of ECs commonly reported in global studies, but also lesser-known ECs. Specifically for pharmaceuticals (human and veterinary origin), further analysis revealed that 32.1 % had not been previously monitored or quantified in agricultural soils, 18.2 % were absent from biosolid/sludge datasets, 21.4 % were missing from slurries/manures datasets, and 45.5 % have not been reported in any of the assessed matrices, including surface waters. These findings highlight the expanding chemical burden in agricultural environments and underscore the need for broader monitoring frameworks. Pharmaceuticals with no environmental monitoring data which require urgent attention include, (11beta,16-beta)-9-fluoro-11-hydroxy-16-methyl-21-[(methylsulfonyl)oxy]-3,20-dioxopregna-1,4-dien-17-yl propionate, alminoprofen, alphaprodine, atomoxetine, azaperol, cordacepin, ethamivan, levalbuterol miglitol, minoxidil, oxprenolol, perindopril, pipenzolate, sulforidazine, tetranactin, and tretinoin (SI Table 33). We strongly suspect that if additional environmental chemical classes had been considered in this analysis, the lack of literature coverage would be even more pronounced. Although plasticisers, surfactants, PEGs, and personal care products are occasionally investigated, their detection in soils is seldom investigated despite their ease in analyses/extraction, [102,8], (SI Tables 10–31). Accompanying the lack of known presence data is the scarcity of comprehensive terrestrial effects data. Together, these gaps demonstrate that our current understanding of environmental risk in agricultural environments remains incomplete. While some literature exists [125], much of it lacks standardisation, thereby hindering meaningful chemical-to-chemical comparisons and clear priority rankings.

It is important to highlight that ECs present in soils pose a risk to other receiving environments such as surface water, groundwater [103].

To evaluate the potential for soil-based ECs to leach into groundwater and inform future monitoring efforts, a leaching assessment was conducted under average UK rainfall conditions (1386 mm/year), [133]. Following an approach similar to Nightingale et al., [104], leachability was estimated using predicted K_{OC} values based on the Franco and Trapp [41] regression equation (SI Figure. 5). The top 50 ECs predicted to leach into groundwater included a mix of herbicides (26.1 %), plasticisers (21.7 %), fungicides (8.7 %), plant growth regulators (4.35 %), and flame retardants (4.35 %), highlighting the diverse and persistent nature of contaminants likely to impact subsurface environments. The top 20 ECs predicted to reach groundwater were primarily PPPs, including herbicides such as difenzoquat, prosulfocarb, flufenacet, atraton, tiocarbazil, and atrazine (SI Table 34). In contrast, veterinary medicines and human pharmaceuticals were ranked lower, suggesting a comparatively reduced risk of leaching but potentially greater impacts on soil microbial communities and plant health [35,46]. Notable examples within the top 50 included tetranactin (a macrotetrolide used in animal feed, rank 18), crotamiton (a scabicidal antipruritic, rank 33), and alminoprofen (a Non-Steroidal Anti Inflammatory (NSAID) rank 42).

Taken together, this dataset highlights a significant gap in our understanding of chemical contamination in English soils, but also globally. The findings raise concerns about the adequacy of current risk assessment frameworks and the limited scope of monitoring in the context of modern agricultural practices. A substantial portion of ECs are being overlooked in both regulatory evaluations and environmental surveillance. Current estimates indicate that approximately 10.1 million tonnes of human waste-derived materials, including biosolids and green waste compost, are applied to UK soils annually, with this volume expected to grow significantly [146,31,5]. Meanwhile, the use of conventional agricultural chemicals such as pesticides and insecticides are declining due to a shift toward greener alternatives like biopesticides. Given that human waste products applied to soils are clearly a major source of EC pollution, it is essential that future regulatory strategies and monitoring efforts prioritise compounds known to be persistent and environmentally relevant.

The combined use of targeted and non-targeted screening as used in this study, offers a pragmatic and comprehensive approach to identifying ECs agricultural soils, supporting the need for a harmonised national monitoring framework. The detection of banned substances such as atrazine and atraton, prohibited in the UK since 1992 and 2009, respectively [37], highlights the persistence of certain ECs in the environment and the role of soils as long-term sinks of contaminants. Their continued presence underscores that regulatory removal from the market does not equate to environmental elimination. This finding reflects the incomplete nature of current chemical risk assessments, particularly regarding persistent residues, transformation products, and complex mixtures. It demonstrates the urgent need for improved monitoring, better alignment between agricultural practices and chemical regulations, and more robust pre-market risk evaluations to safeguard soil health. There is an increasing need for regulators and risk assessors to expand monitoring programs such as this one to ensure that environmental risk assessments better reflect real-world conditions. Although the data is currently limited, evidence suggests that pre-application treatment of waste materials (composting or anaerobic digestion), offers a suitable approach to reduce chemical contaminant loads. These practices could be adopted by farmers via various incentives to help mitigate potential environmental risks associated with organic fertiliser application. Together, these measures would provide a more realistic evaluation of chemical risks in agricultural systems and help safeguard soil and water quality.

4. Conclusions

Our results demonstrate the widespread occurrence of ECs in English agricultural soils receiving an array of organic amendments. Over 520 chemicals were identified to a high confidence level (Level 1–2), representing 194 unique entities. The contaminant classes most frequently detected were human pharmaceuticals (n = 52, Rank 1) > plasticisers (n = 51, Rank 2) > surfactants (n = 44, Rank 3) > fungicides (n = 40, Rank 4) > polymers (n = 35, Rank 5) > veterinary medicines (n = 23, Rank 6) > herbicides (n = 20, Rank 7) > human or veterinary use pharmaceuticals (n = 16, Rank 8) > disinfectant (n = 14, Rank 9) > pharmaceutical derivative/research chemical (n = 11, Rank 10). These dominant classes differ from those reported in previous studies, highlighting the importance of context-specific chemical classification when assessing risks to soil, plant and human health. Notably, 46.4 % of the identified pharmaceutical compounds—both veterinary and human—had no prior documentation in the scientific literature, regardless of environmental matrix. This underscores the limitations of current monitoring frameworks and the need for expanded EC surveillance. Quantified concentrations of pharmaceuticals in soils were found to range between 0.06 ± 0.07 (sulphamethoxazole) ng/g (dw) - 105.9 ± 86.7 (ivermectin) ng/g (dw), and positive correlations were identified between concentration and K_{OC}. This demonstrates that fate processes play a fundamental role in the presence and concentration of ECs in agricultural soils, and thus their risk towards terrestrial biota (i.e., microbial populations, earthworms). The potential risks associated with this exposure require further investigation. Although elevated concentrations of ivermectin, oxytetracycline, and ofloxacin raise immediate concerns for soil health, the primary contaminant classes posing leaching risks were plasticisers (35 %), herbicides (26.1 %), plasticisers (21.7 %), fungicides (8.7 %), plant growth regulators (4.4 %), and flame retardants (4.4 %). Overall, our understanding of the risks posed by ECs identified in this monitoring campaign are limited. This underscores the urgent need for a more harmonised approach to contaminant identification in agricultural soils and more comprehensive evaluations of the risks they pose to ecological receptors and connected environmental compartments. The engagement of farmers within this study through citizen science approach not only improved sampling reach but also fostered awareness and stewardship among key stakeholders. This will be critical in maintaining soil health as the UK moves towards a circular economy with increased organic waste reuse within agriculture. This study underscores the urgent need for a more harmonised approach to contaminant identification in agricultural soils and more comprehensive evaluations of the risks they pose to ecological receptors and connected environmental compartments.

Environmental implications

The presented study demonstrates that the reuse of contaminated organic fertilisers contributes to the widespread occurrence, and accumulation of emerging contaminants in agricultural soils. Over 520 contaminants were identified, some reported for the first time in England, with concentrations ranging from 0.06 ± 0.07–105.9 ± 86.7 ng/g (dw). Notably, 46.4 % of pharmaceuticals had not been detected in previous monitoring campaigns, underscoring the unknown risks these chemicals pose in arable soils. There is a clear need to accurately assess the risks associated with different organic fertiliser applications to protect both environmental and consumer health.

Author contributions

John Nightingale: Sampling, method development, method validation, sample processing, formal analyses, project administration, statistical analyses, modelling, reporting, writing-editing. Felicity Elder: Sampling campaign arrangement, sampling, and project administration, writing-editing. Andrea Garduño-Jiménez: Sampling campaign arrangement, sampling, project administration, sample processing, writing-editing. Laura Carter, funding acquisition, conceptualisation, project management, sampling campaign, sampling, processing, data processing, reporting, writing-editing.

CRediT authorship contribution statement

Elder Felicity: Writing – review & editing, Writing – original draft, Project administration, Investigation, Conceptualization. **John Henry Nightingale:** Writing – review & editing, Writing – original draft, Visualization, Validation, Software, Project administration, Methodology, Investigation, Formal analysis, Data curation. **Laura J. Carter:** Writing – review & editing, Writing – original draft, Supervision, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Conceptualization. **Andrea-Lorena Garduño-Jiménez:** Writing – review & editing, Writing – original draft, Project administration, Investigation, Formal analysis, Data curation, Conceptualization.

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Declaration of Competing Interest

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.jhazmat.2025.140433.

Data availability

Data will be made available on request.

References

[1] Adler, N., Bachmann, J., Blanckenhorn, W.U., Floate, K.D., Jensen, J., Römbke, J., 2016. Effects of ivermectin application on the diversity and function of dung and soil fauna: regulatory and scientific background information. Environ Toxicol Chem 35 (8), 1914–1923. https://doi.org/10.1002/etc.3308.

[2] Aggarwal V., Deng X., Tuli A., Goh K.S. 2012. Diazinon—chemistry and environmental fate: a California perspective. In: Reviews of Environmental Contamination and Toxicology, Volume 223, 107–140. https://doi.org/10.1007/978-1-4614-5577-6_5.

[3] Al-Rajab, A.J., Sabourin, L., Lapen, D.R., Topp, E., 2010. The non-steroidal anti-inflammatory drug diclofenac is readily biodegradable in agricultural soils. Sci Total Environ 409 (1), 78–82. https://doi.org/10.1016/j.scitotenv.2010.09.020.

[4] Alvarez-Ruiz, R., Choi, Y., Costello, M.C.S., Lee, L.S., 2025. Analysis of multi-class unregulated organic compounds in soil and biosolids using LC-MS/MS. Environ Pollut 368, 125727. https://doi.org/10.1016/j.envpol.2025.125727.

[5] Assured Biosolids Limited (2019). *Biosolids Agricultural Good Practice Guidance Leaflet*. 〈https://assuredbiosolids.co.uk/wp-content/uploads/2019/01/Biosolids-Agric-Good-Practice-Guidance-January-2019.pdf〉.

[6] Awad, Y.M., Kim, S.C., EI-Azeem, S.A.M.A., Kim, K.H., Kim, K.R., Kim, K.J., Jeon, C., Lee, S.S., Ok, Y.S., 2014. Veterinary antibiotics contamination in water, sediment, and soil near a swine manure composting facility. Environ Earth Sci 71, 1433–1440. https://doi.org/10.1007/s12665-013-2548-z.

[7] Banerjee, S., Van der Heijden, M.G., 2023. Soil microbiomes and One Health. Nat Rev Microbiol 21 (1), 6–20. https://doi.org/10.1038/s41579-022-00779-w.

[8] Billings, A., Carter, H., Cross, R.K., Jones, K.C., Pereira, M.G., Spurgeon, D.J., 2023. Co-occurrence of macroplastics, microplastics, and legacy and emerging plasticisers in UK soils. Sci Total Environ 880, 163258. https://doi.org/10.1016/j.scitotenv.2023.163258.

[9] Billings, A., Jones, K.C., Pereira, M.G., Spurgeon, D.J., 2021. Plasticisers in the terrestrial environment: sources, occurrence and fate. Environ Chem 18 (3), 111–130. https://doi.org/10.1071/EN21033.

[10] Blackwell, P.A., Boxall, A.B., Kay, P., Noble, H., 2005. Evaluation of a lower tier exposure assessment model for veterinary medicines. J Agric Food Chem 53 (6), 2192–2201. https://doi.org/10.1021/jf049527b.

[11] Blume, E., Bischoff, M., Moorman, T.B., Turco, R.F., 2004. Degradation and binding of atrazine in surface and subsurface soils. J Agric Food Chem 52 (24), 7382–7388. https://doi.org/10.1021/jf049830c.

[12] Borgman, O., Chefetz, B., 2013. Combined effects of biosolids application and irrigation with reclaimed wastewater on transport of pharmaceutical compounds in arable soils. Water Res 47 (10), 3431–3443. https://doi.org/10.1016/j.watres.2013.03.045.

[13] Braine, M.F., Kearnes, M., Khan, S.J., 2024. Quality and risk management frameworks for biosolids: An assessment of current international practice. Sci Total Environ 915, 169953. https://doi.org/10.1016/j.scitotenv.2024.169953.

[14] Brändli, R.C., Bucheli, T.D., Kupper, T., Mayer, J., Stadelmann, F.X., Tarradellas, J., 2007. Fate of PCBs, PAHs and their source characteristic ratios during composting and digestion of source-separated organic waste in full-scale plants. Environ Pollut 148 (2), 520–528. https://doi.org/10.1016/j.envpol.2006.11.021.

[15] Burns, E.E., Carter, L.J., Snape, J., Thomas-Oates, J., Boxall, A.B., 2018. Application of prioritization approaches to optimize environmental monitoring and testing of pharmaceuticals. J Toxicol Environ Health Part B 21 (3), 115–141. https://doi.org/10.1080/10937404.2018.1465873.

[16] Campo, J., Masiá, A., Blasco, C., Picó, Y., 2013. Occurrence and removal efficiency of pesticides in sewage treatment plants of four Mediterranean River Basins. J Hazard Mater 263, 146–157. https://doi.org/10.1016/j.jhazmat.2013.09.061.

[17] Carlson, J.C., Mabury, S.A., 2006. Dissipation kinetics and mobility of chlortetracycline, tylosin, and monensin in an agricultural soil in Northumberland County, Ontario, Canada. Environ Toxicol Chem 25 (1), 1–10. https://doi.org/10.1897/04-657R.1.

[18] Carrillo Heredero, A.M., Segato, G., Menotta, S., Faggionato, E., Vismarra, A., Genchi, M., Bertini, S., 2023. A new method for ivermectin detection and quantification through HPLC in organic matter (feed, soil, and water). J Anal Methods Chem 2023, 6924263. https://doi.org/10.1155/2023/6924263.

[19] Carter, L.J., Adams, B., Berman, T., Cohen, N., Cytryn, E., Elder, F.C.T., Garduño-Jiménez, A.L., Greenwald, D., Kasprzyk-Hordern, B., Korach-Rechtman, H., Lahive, E., 2025. Co-contaminant risks in water reuse and biosolids application for agriculture. Environ Pollut, 126219. https://doi.org/10.1016/j.envpol.2025.126219.

[20] Česen, M., Kosjek, T., Laimou-Geraniou, M., Kompare, B., Širok, B., Lambropoulou, D., Heath, E., 2015. Occurrence of cyclophosphamide and ifosfamide in aqueous environment and their removal by biological and abiotic wastewater treatment processes. Sci Total Environ 527, 465–473. https://doi.org/10.1016/j.scitotenv.2015.04.109.

[21] Chari, B.P., Halden, R.U., 2012. Predicting the concentration range of unmonitored chemicals in wastewater-dominated streams and in run-off from biosolids-amended soils. Sci Total Environ 440, 314–320. https://doi.org/10.1016/j.scitotenv.2012.05.042.

[22] Chen, Z.F., Ying, G.G., Ma, Y.B., Lai, H.J., Chen, F., Pan, C.G., 2013. Occurrence and dissipation of three azole biocides—climbazole, clotrimazole and miconazole—in biosolid-amended soils. Sci Total Environ 452, 377–383. https://doi.org/10.1016/j.scitotenv.2013.03.004.

[23] Clarke, R.M., Cummins, E., 2015. Evaluation of "classic" and emerging contaminants resulting from the application of biosolids to agricultural lands: a review. Hum Ecol Risk Assess Int J 21 (2), 492–513. https://doi.org/10.1080/10807039.2014.930295.

[24] Clarke, R., Healy, M.G., Fenton, O., Cummins, E., 2016. A quantitative risk ranking model to evaluate emerging organic contaminants in biosolid-amended land and potential transport to drinking water. Hum Ecol Risk Assess 22 (4), 958–990. https://doi.org/10.1080/10807039.2015.1121376.

[25] Davis, E.F., Klosterhaus, S.L., Stapleton, H.M., 2012. Measurement of flame retardants and triclosan in municipal sewage sludge and biosolids. Environ Int 40, 1–7. https://doi.org/10.1016/j.envint.2011.11.008.

[26] De Liguoro, M., Cibin, V., Capolongo, F., Halling-Sørensen, B., Montesissa, C., 2003. Use of oxytetracycline and tylosin in intensive calf farming: evaluation of transfer to manure and soil. Chemosphere 52 (1), 203–212. https://doi.org/10.1016/S0045-6535(03)00284-4.

[27] Delgado, N., Orozco, J., Zambrano, S., Casas-Zapata, J.C., Marino, D., 2023. Veterinary pharmaceutical as emerging contaminants in wastewater and surface water: an overview. J Hazard Mater 460, 132431. https://doi.org/10.1016/j.jhazmat.2023.132431.

[28] Department for Environment, Food & Rural Affairs & Pow, R (2020). Circular economy measures drive forward ambitious plans for waste. 〈https://www.gov.uk/government/news/circular-economy-measures-drive-forward-ambitious-plans-for-waste〉.

[29] Ding, Y., Wang, J., Zhang, Y., Zhang, Y., Xu, W., Zhang, X., Wang, Y., Li, D., 2023. Response characteristics of indigenous microbial community in PAHs-contaminated aquifers under polyethylene microplastics stress: A microcosmic experimental study. Sci Total Environ 894, 164900. https://doi.org/10.1016/j.scitotenv.2023.164900.

[30] Dolliver, H., Kumar, K., Gupta, S., 2007. Sulfamethazine uptake by plants from manure-amended soil. J Environ Qual 36 (4), 1224–1230. https://doi.org/10.2134/jeq2006.0266.

[31] Earl, K., Sleight, H., Ashfield, N., Boxall, A.B., 2024. Are pharmaceutical residues in crops a threat to human health? J Toxicol Environ Health Part A 87 (19), 773–791. https://doi.org/10.1080/15287394.2024.2371418.

[32] EFSA Panel on Additives and Products or Substances used in Animal Feed (FEEDAP), 2019. Safety and efficacy of Robenz® 66G (robenidine hydrochloride) for chickens for fattening and turkeys for fattening. EFSA J 17 (10), e05613. https://doi.org/10.2903/j.efsa.2019.5613.

[33] EFSA Panel on Plant Protection Products and their Residues (PPR) (Ockleford, C., et al. (2018). Scientific Opinion about the CRD (UK) guidance on aged sorption studies for pesticides. *EFSA Journal*, 16(8), e05382. https://doi.org/10.2903/j.efsa.2018.5382.

[34] EMA, 2016. Guideline on Environmental Impact Assessment for Veterinary Medicinal Products in Support of VICH Guidelines GL6 and GL38. European Medicines Agency. 〈https://www.ema.europa.eu/en/documents/scientific-guideline/guideline-environmental-impact-assessment-veterinary-medicinal-products-support-vich-guidelines-gl6_en.pdf〉. EMA/CVMP/ERA/418282/2005-Rev.1-Corr. [pdf] Available at.

[35] Esteban, S., Llamas, P.M., García-Cortés, H., Catalá, M., 2016. The endocrine disruptor nonylphenol induces sublethal toxicity in vascular plant development at environmental concentrations. Environ Pollut 216, 480–486. https://doi.org/10.1016/j.envpol.2016.05.086.

[36] European Commission (2023) Guidance Document on Pesticide Analytical Methods for Risk Assessment and Post-approval Control and Monitoring Purposes. SANTE/2020/12830 Rev. 2. Brussels: European Commission. Available at: 〈https://food.ec.europa.eu/document/download/d8f2d862-94ea-412d-a0dd-c037e41898b2_en〉.

[37] European Parliament & Council. (2009). Regulation (EC) No 1107/2009. *Official Journal of the European Union*, L 309, 1–50. (no DOI) 〈https://eur-lex.europa.eu/eli/reg/2009/1107/oj/eng〉.

[38] Fang, Q., Yan, Z., Zhang, C., Shi, Y., Zhang, Z., Gao, Q., Cao, H., 2023. Insights into the fungicide prothioconazole and its metabolite in wheat: Residue behavior and risk assessment. Agronomy 13 (12), 2906. https://doi.org/10.3390/agronomy13122906.

[39] Förster, M., Laabs, V., Lamshoft, M., Groeneweg, J., Zuhlke, S., Spiteller, M., Krauss, M., Kaupenjohann, M., Amelung, W., 2009. Sequestration of manure-applied sulfadiazine residues in soils. Environ Sci Technol 43 (6), 1824–1830. https://doi.org/10.1021/es8026538.

[40] Frade, V.M., Converti, A., Arni, S.A., Silva, M.F., Palma, M.S., 2017. Batch and Fed-Batch Degradation of Enrofloxacin by the Fenton Process. Chem Eng Technol 40 (4), 663–669. https://doi.org/10.1002/ceat.201600146.

[41] Franco, A., Trapp, S., 2008. Estimation of the soil–water partition coefficient normalized to organic carbon for ionizable organic chemicals. Environ. Toxicol. Chem.: Int. J. 27 (10), 1995–2004.

[42] Fučík, J., Amrichová, A., Brabcová, K., Karpíšková, R., Koláčková, I., Pokludová, L., Poláková, Š., Mravcová, L., 2024. Fate of fluoroquinolones in field soil environment after incorporation of poultry litter from a farm with enrofloxacin administration via drinking water. Environ Sci Pollut Res 31 (13), 20017–20032. https://doi.org/10.1007/s11356-024-32492-x.

[43] Gao, P., Ding, Y., Li, H., Xagoraraki, I., 2012. Occurrence of pharmaceuticals in a municipal wastewater treatment plant: mass balance and removal processes. Chemosphere 88 (1), 17–24. https://doi.org/10.1016/j.chemosphere.2012.02.017.

[44] Garduño-Jiménez, A.L., Carter, L.J., 2024. Insights into mode of action mediated responses following pharmaceutical uptake and accumulation in plants. Front Agron 5, 1293555. https://doi.org/10.3389/fagro.2023.1293555.

[45] Ghassempour, A., Mohammadkhah, A., Najafi, F., Rajabzadeh, M., 2002. Monitoring of the pesticide diazinon in soil, stem and surface water of rice fields. Anal Sci 18 (7), 779–783. https://doi.org/10.2116/analsci.18.779.

[46] Gomes, A.R., Justino, C., Rocha-Santos, T., Freitas, A.C., Duarte, A.C., Pereira, R., 2017. Review of the ecotoxicological effects of emerging contaminants to soil biota. J Environ Sci Health Part A 52 (10), 992–1007. https://doi.org/10.1080/10934529.2017.1328946.

[47] Gottschall, N., Topp, E., Metcalfe, C., Edwards, M., Payne, M., Kleywegt, S., Russell, P., Lapen, D.R., 2012. Pharmaceutical and personal care products in groundwater, subsurface drainage, soil, and wheat grain, following a high single application of municipal biosolids to a field. Chemosphere 87 (2), 194–203. https://doi.org/10.1016/j.chemosphere.2011.12.018.

[48] Gravert, T.K.O., Vuaille, J., Magid, J., Hansen, M., 2021. Non-target analysis of organic waste amended agricultural soils: characterization of added organic pollution. Chemosphere 280, 130582. https://doi.org/10.1016/j.chemosphere.2021.130582.

[49] Gros, M., Mas-Pla, J., Boy-Roura, M., Geli, I., Domingo, F., Petrović, M., 2019. Veterinary pharmaceuticals and antibiotics in manure and slurry and their fate in amended agricultural soils: Findings from an experimental field site (Baix Empordà, NE Catalonia). Sci Total Environ 654, 1337–1349. https://doi.org/10.1016/j.scitotenv.2018.11.061.

[50] Grote, M., Schwake-Anduschus, C., Michel, R., Stevens, H., Heyser, W., Langenkämper, G., 2007. … & Freitag, M. Inc Vet Antibiot into Crops Manure Soil Landbauforsch 57 (1), 25.

[51] Hansen, M., Krogh, K.A., Brandt, A., Christensen, J.H., Halling-Sørensen, B., 2009. Fate and antibacterial potency of anticoccidial drugs and their main abiotic degradation products. Environ Pollut 157 (2), 474–480. https://doi.org/10.1016/j.envpol.2008.09.022.

[52] Ho, Y.B., Zakaria, M.P., Latif, P.A., Saari, N., 2014. Occurrence of veterinary antibiotics and progesterone in broiler manure and agricultural soil in Malaysia. Sci Total Environ 488, 261–267. https://doi.org/10.1016/j.scitotenv.2014.04.109.

[53] Holling, C.S., Bailey, J.L., Vanden Heuvel, B., Kinney, C.A., 2012. Uptake of human pharmaceuticals and personal care products by cabbage (Brassica campestris) from fortified and biosolids-amended soils (D). J Environ Monit 14 (11), 3029–3036. https://doi.org/10.1039/C2EM30456B.

[54] Hou, J., Wan, W., Mao, D., Wang, C., Mu, Q., Qin, S., Luo, Y., 2015. Occurrence and distribution of sulfonamides, tetracyclines, quinolones, macrolides, and nitrofurans in livestock manure and amended soils of Northern China. Environ Sci Pollut Res 22 (6), 4545–4554. https://doi.org/10.1007/s11356-014-3632-y.

[55] Huang, D., Gao, L., Zheng, M., Qiao, L., Xu, C., Wang, K., Wang, S., 2022. Screening organic contaminants in soil by two-dimensional gas chromatography high-resolution time-of-flight mass spectrometry: A non-target analysis strategy and contaminated area case study. Environ Res 205, 112420. https://doi.org/10.1016/j.envres.2021.112420.

[56] Iglesias, L.E., Saumell, C., Sagüés, F., Sallovitz, J.M., Lifschitz, A.L., 2018. Ivermectin dissipation and movement from feces to soil under field conditions. J Environ Sci Health Part B 53 (1), 42–48. https://doi.org/10.1080/03601234.2017.1371554.

[57] Jones, A.D., Bruland, G.L., Agrawal, S.G., Vasudevan, D., 2005. Factors influencing the sorption of oxytetracycline to soils. Environ Toxicol Chem 24 (4), 761–770. https://doi.org/10.1897/04-037R.1.

[58] Kahle, M., Buerge, I.J., Hauser, A., Müller, M.D., Poiger, T., 2008. Azole fungicides: occurrence and fate in wastewater and surface waters. Environ Sci Technol 42 (19), 7193–7200. https://doi.org/10.1021/es8009309.

[59] Kang, D.H., Gupta, S., Rosen, C., Fritz, V., Singh, A., Chander, Y., Murray, H., Rohwer, C., 2013. Antibiotic uptake by vegetable crops from manure-applied soils. J Agric Food Chem 61 (42), 9992–10001. https://doi.org/10.1021/jf404045m.

[60] Karcı, A., Balcıoğlu, I.A., 2009. Investigation of the tetracycline, sulfonamide, and fluoroquinolone antimicrobial compounds in animal manure and agricultural soils in Turkey. Sci Total Environ 407 (16), 4652–4664. https://doi.org/10.1016/j.scitotenv.2009.04.047.

[61] Karnjanapiboonwong, A., Morse, A.N., Maul, J.D., Anderson, T.A., 2010. Sorption of estrogens, triclosan, and caffeine in a sandy loam and a silt loam soil. J Soils Sediment 10, 1300–1307. https://doi.org/10.1007/s11368-010-0223-5.

[62] Khan, B., Lee, L.S., Sassman, S.A., 2008. Degradation of synthetic androgens 17α- and 17β-trenbolone and trendione in agricultural soils. Environ Sci Technol 42 (10), 3570–3574. https://doi.org/10.1021/es702690p.

[63] Kim, S.C., Chung, D.Y., Kim, K.H., Lee, J.H., Kim, H.K., Yang, J.E., Ok, Y.S., Almarwei, Y.A., 2012. Concentration and environmental loading of veterinary antibiotics in agricultural irrigation ditches. Korean J Soil Sci Fertil 45 (6), 867–876. https://doi.org/10.7745/KJSSF.2012.45.6.867.

[64] Kim, K.R., Owens, G., Kwon, S.I., So, K.H., Lee, D.B., Ok, Y.S., 2011. Occurrence and environmental fate of veterinary antibiotics in the terrestrial environment. Water Air Soil Pollut 214, 163–174. https://doi.org/10.1007/s11270-010-0412-2.

[65] Kivits, T., Broers, H.P., Beeltje, H., van Vliet, M., Griffioen, J., 2018. Presence and fate of veterinary antibiotics in age-dated groundwater in areas with intensive livestock farming. Environ Pollut 241, 988–998. https://doi.org/10.1016/j.envpol.2018.05.085.

[66] Koestoer, R.H., Ligayanti, T., Kartohardjono, S., Susanto, H., 2024. Down-streaming small-scale green ammonia to nitrogen-phosphorus fertilizer tablets for rural communities. Emerg Sci J 8 (2), 625–643. https://doi.org/10.28991/ESJ-2024-08-02-016.

[67] Krogh, K.A., Jensen, G.G., Schneider, M.K., Fenner, K., Halling-Sørensen, B., 2009. Analysis of the dissipation kinetics of ivermectin at different temperatures and in four different soils. Chemosphere 75, 1097–1104. https://doi.org/10.1016/j.chemosphere.2009.01.015.

[68] Krogh, K.A., Søeborg, T., Brodin, B., Halling-Sørensen, B., 2008. Sorption and mobility of ivermectin in different soils. J Environ Qual 37 (6), 2202–2211. https://doi.org/10.2134/jeq2007.0592.

[69] Kuchta, S.L., Cessna, A.J., Elliott, J.A., Peru, K.M., Headley, J.V., 2009. Transport of lincomycin to surface and ground water from manure-amended cropland. J Environ Qual 38 (4), 1719–1727. https://doi.org/10.2134/jeq2008.0365.

[70] Kumar, K., Gupta, S.C., Baidoo, S.K., Chander, Y., Rosen, C.J., 2005. Antibiotic uptake by plants from soil fertilized with animal manure. J Environ Qual 34 (6), 2082–2085. https://doi.org/10.2134/jeq2005.0026.

[71] Kumar, R., Whelan, A., Cannon, P., Sheehan, M., Reeves, L., Antunes, E., 2023. Occurrence of emerging contaminants in biosolids in northern Queensland, Australia. Environ Pollut 330, 121786. https://doi.org/10.1016/j.envpol.2023.121786.

[72] Kümmerer, K., Al-Ahmad, A., Mersch-Sundermann, V., 2000. Biodegradability of some antibiotics, elimination of the genotoxicity and affection of wastewater bacteria in a simple test. Chemosphere 40 (7), 701–710. https://doi.org/10.1016/S0045-6535(99)00439-7.

[73] Lara-Martín, P.A., Schinkel, L., Eberhard, Y., Giger, W., Berg, M., Hollender, J., 2025. Suspect and Non-target Screening of Organic Micropollutants in Swiss Sewage Sludge: A Nationwide Survey. Environ Sci Technol. https://doi.org/10.1021/acs.est.4c13217.

[74] Lautz, L.S., Struijs, J., Nolte, T.M., Breure, A.M., Van der Grinten, E., Van de Meent, D., Van Zelm, R., 2017. Evaluation of SimpleTreat 4.0: Simulations of pharmaceutical removal in wastewater treatment plant facilities. Chemosphere 168, 870–876. https://doi.org/10.1016/j.chemosphere.2016.10.123.

[75] Lazewski, A., 2024. Feeding Our Food; Starving our environment: The impacts of synthetic fertilizers. Univ Colo Honors J. https://doi.org/10.33011/cuhj20242705.

[76] Leal, R.M.P., Figueira, R.F., Tornisielo, V.L., Regitano, J.B., 2012. Occurrence and sorption of fluoroquinolones in poultry litters and soils from São Paulo State,

Brazil. Sci Total Environ 432, 344–349. https://doi.org/10.1016/j.scitotenv.2012.06.002.
[77] Li, J., Carter, L.J., Boxall, A.B., 2020. Evaluation and development of models for estimating the sorption behaviour of pharmaceuticals in soils. J Hazard Mater 392, 122469. https://doi.org/10.1016/j.jhazmat.2020.122469.
[78] Li, Z.L., Cheng, R., Chen, F., Lin, X.Q., Yao, X.J., Liang, B., Huang, C., Sun, K., Wang, A.J., 2021. Selective stress of antibiotics on microbial denitrification: inhibitory effects, dynamics of microbial community structure and function. J Hazard Mater 405, 124366. https://doi.org/10.1016/j.jhazmat.2020.124366.
[79] Li, L., Sun, J., Liu, B., Zhao, D., Ma, J., Deng, H., Li, X., Hu, F., Liao, X., Liu, Y., 2013. Quantification of lincomycin resistance genes associated with lincomycin residues in waters and soils adjacent to representative swine farms in China. Front Microbiol 4, 364. https://doi.org/10.3389/fmicb.2013.00364.
[80] Li, Y., Wang, H., Liu, X., Zhao, G., Sun, Y., 2016. Dissipation kinetics of oxytetracycline, tetracycline, and chlortetracycline residues in soil. Environ Sci Pollut Res 23, 13822–13831. https://doi.org/10.1007/s11356-016-6513-8.
[81] Li, D., Yang, M., Hu, J., Ren, L., Zhang, Y., Li, K., 2008. Determination and fate of oxytetracycline and related compounds in oxytetracycline production wastewater and the receiving river. Environ Toxicol Chem 27 (1), 80–86. https://doi.org/10.1897/07-080.1.
[82] Liebig, M., Fernandez, Á.A., Blübaum-Gronau, E., Boxall, A., Brinke, M., Carbonell, G., Garric, J., 2010. Environmental risk assessment of ivermectin: a case study. Integr Environ Assess Manag 6 (S1), 567–587. https://doi.org/10.1002/ieam.96.
[83] Lin, H., Dong, B., Hu, J., 2017. Residue and intake risk assessment of prothioconazole and its metabolite prothioconazole-desthio in wheat field. Environ Monit Assess 189, 482. https://doi.org/10.1007/s10661-017-5943-1.
[84] Loftin, K.A., 2006. PhD thesis, University of Missouri-Rolla. Eff fate Sel Vet Antibiot two Mo Anaerob swine lagoons.
[85] Loke, M.L., Tjørnelund, J., Halling-Sørensen, B., 2002. Determination of the distribution coefficient (log Kd) of oxytetracycline, tylosin A, olaquindox and metronidazole in manure. Chemosphere 48 (3), 351–361. https://doi.org/10.1016/S0045-6535(02)00078-4.
[86] Lucci, P., Saurina, J., Núñez, O., 2017. Trends in LC-MS and LC-HRMS analysis and characterization of polyphenols in food. TrAC Trends Anal Chem 88, 1–24. https://doi.org/10.1016/j.trac.2016.12.006.
[87] LUFA Speyer (2024). *Chemical and physical characteristics of standard soils according to GLP. Version 8 D*12-18. (no DOI) ⟨https://www.lufa-speyer.de/sites/default/files/2025-06/Chemical%20and%20physical%20characteristics%20of%20standard%20soils%20according%20to%20GLP.pdf⟩.
[88] Malvar, J.L., Santos, J.L., Martín, J., Aparicio, I., Alonso, E., 2021. Occurrence of the main metabolites of the most recurrent pharmaceuticals and personal care products in Mediterranean soils. J Environ Manag 278, 111584.
[89] Mansouri, K., Grulke, C.M., Judson, R.S., Williams, A.J., 2018. OPERA models for predicting physicochemical properties and environmental fate endpoints. J Chemin 10 (1), 10. https://doi.org/10.1186/s13321-018-0263-1.
[90] Martin, I., Hart, A., 2023. Antifungal medicines in the terrestrial environment: levels in biosolids from England and Wales. Sci Total Environ 870, 161999. https://doi.org/10.1016/j.scitotenv.2023.161999.
[91] Martins, E.C., Bohone, J.B., Abate, G., 2018. Sorption and desorption of atrazine on soils: The effect of different soil fractions. Geoderma 322, 131–139. https://doi.org/10.1016/j.geoderma.2018.02.028.
[92] Meffe, R., De Santiago, A., Teijón, G., Martínez-Hernández, V., López-Heras, I., Nozal, L., De Bustamante, I., 2021. Should we be concerned about the presence of pharmaceuticals during unplanned water reuse for crop irrigation? Curr Opin Environ Sci Health 19, 100223.
[93] Menacherry, S.P.M., Kodešová, R., Fedorova, G., Sadchenko, A., Kočárek, M., Klement, A., Grabic, R., 2023. Dissipation of twelve organic micropollutants in three different soils: Effect of soil characteristics and microbial composition. J Hazard Mater 459, 132143. https://doi.org/10.1016/j.jhazmat.2023.132143.
[94] Mesa, L., Gutiérrez, M.F., Montalto, L., Perez, V., Lifschitz, A., 2020. Concentration and environmental fate of ivermectin in floodplain wetlands: an ecosystem approach. Sci Total Environ 706, 135692. https://doi.org/10.1016/j.scitotenv.2019.135692.
[95] Michael, C., Bayona, J.M., Lambropoulou, D., Agüera, A., Fatta-Kassinos, D., 2017. Two important limitations relating to spiking environmental samples with contaminants of emerging concern: How close to the real analyte concentrations are the reported recovered values? Environ Sci Pollut Res 24, 15202–15205. https://doi.org/10.1007/s00216-009-2669-0.
[96] Mohamed, A.H., Noorhisham, N.A., Yahaya, N., Mohamad, S., Kamaruzzaman, S., Osman, H., Aboul-Enein, H.Y., 2023. Sampling and sample preparation techniques for the analysis of organophosphorus pesticides in soil matrices. Crit Rev Anal Chem 53 (4), 906–927. https://doi.org/10.1080/10408347.2021.1992262.
[97] Mordechay, E.B., Mordehay, V., Tarchitzky, J., Chefetz, B., 2022. Fate of contaminants of emerging concern in the reclaimed wastewater-soil-plant continuum. Sci Total Environ 822, 153574. https://doi.org/10.1016/j.scitotenv.2022.153574.
[98] Mrozik, W., Stefańska, J., 2014. Adsorption and biodegradation of antidiabetic pharmaceuticals in soils. Chemosphere 95, 281–288. https://doi.org/10.1016/j.chemosphere.2013.09.012.
[99] Natal-da-Luz, T., Moreira-Santos, M., Ruepert, C., Castillo, L.E., Ribeiro, R., Sousa, J.P., 2012. Ecotoxicological characterization of a tropical soil after diazinon spraying. Ecotoxicology 21 (8), 2163–2176. https://doi.org/10.1007/s10646-012-0970-8.
[100] Nde, S.C., Felicite, O.M., Aruwajoye, G.S., Palamuleni, L.G., 2024. A meta-analysis and experimental survey of heavy metals pollution in agricultural soils. J Trace Elem Miner 9, 100180. https://doi.org/10.1016/j.jtemin.2024.100180.
[101] Nemeth-Konda, L., Füleky, G., Morovjan, G., Csokan, P., 2002. Sorption behaviour of acetochlor, atrazine, carbendazim, diazinon, imidacloprid and isoproturon on Hungarian agricultural soil. Chemosphere 48 (5), 545–552. https://doi.org/10.1016/S0045-6535(02)00106-6.
[102] Net, S., Delmont, A., Sempéré, R., Paluselli, A., Ouddane, B., 2015. Reliable quantification of phthalates in environmental matrices (air, water, sludge, sediment and soil): a review. Sci Total Environ 515, 162–180. https://doi.org/10.1016/j.scitotenv.2015.02.013.
[103] Nightingale, J., Carter, L., Sinclair, C.J., Rooney, P., Kay, P., 2023. Influence of manure application method on veterinary medicine losses to water. J Environ Manag 334, 117361. https://doi.org/10.1016/j.jenvman.2023.117361.
[104] Nightingale, J., Trapp, S., Garduño-Jiménez, A., Carter, L., 2025. A framework to assess pharmaceutical accumulation in crops: from wastewater irrigation to consumption. J Hazard Mater, 138297. https://doi.org/10.1016/j.jhazmat.2025.138297.
[105] Nordenholt, R.M., Goyne, K.W., Kremer, R.J., Lin, C.H., Lerch, R.N., Veum, K.S., 2016. Veterinary antibiotic effects on atrazine degradation and soil microorganisms. J Environ Qual 45 (2), 565–575. https://doi.org/10.2134/jeq2015.05.0235.
[106] OECD (2022). Pharmaceutical Market. OECD Statistics. ⟨https://stats.oecd.org/Index.aspx?DataSetCode=HEALTH_PHMC⟩.
[107] Otoo, B.A., Amoabeng, I.A., Darko, G., Borquaye, L.S., 2022. Antibiotic and analgesic residues in the environment – occurrence and ecological risk study from the Sunyani municipality, Ghana. Toxicol Rep 9, 1491–1500. https://doi.org/10.1016/j.toxrep.2022.07.003.
[108] Pannu, M.W., Toor, G.S., O'Connor, G.A., Wilson, P.C., 2012. Toxicity and bioaccumulation of biosolids-borne triclosan in food crops. Environ Toxicol Chem 31 (9), 2130–2137. https://doi.org/10.1002/etc.1930.
[109] Paz, A., Tadmor, G., Malchi, T., Blotevogel, J., Borch, T., Polubesova, T., Chefetz, B., 2016. Fate of carbamazepine, its metabolites, and lamotrigine in soils irrigated with reclaimed wastewater: Sorption, leaching and plant uptake. Chemosphere 160, 22–29. https://doi.org/10.1016/j.chemosphere.2016.06.048.
[110] Pourcher, A.M., Jadas-Hécart, A., Cotinet, P., Dabert, P., Ziebal, C., Le Roux, S., Moraru, R., Heddadj, D., Kempf, I., 2014. Effect of land application of manure from enrofloxacin-treated chickens on ciprofloxacin resistance of Enterobacteriaceae in soil. Sci Total Environ 482, 269–275. https://doi.org/10.1016/j.scitotenv.2014.02.136.
[111] Prado, N., Ochoa, J., Amrane, A., 2009. Biodegradation and biosorption of tetracycline and tylosin antibiotics in activated sludge system. Process Biochem 44 (11), 1302–1306. https://doi.org/10.1016/j.procbio.2009.08.006.
[112] Previšić, A., Vilenica, M., Vučković, N., Petrović, M., Rožman, M., 2021. Aquatic insects transfer pharmaceuticals and endocrine disruptors from aquatic to terrestrial ecosystems. Environ Sci Technol 55 (6), 3736–3746. https://doi.org/10.1021/acs.est.0c07609.
[113] Prosser, R.S., Lissemore, L., Topp, E., Sibley, P.K., 2014. Bioaccumulation of triclosan and triclocarban in plants grown in soils amended with municipal dewatered biosolids. Environ Toxicol Chem 33 (5), 975–984. https://doi.org/10.1002/etc.2505.
[114] Prosser, R.S., Sibley, P.K., 2015. Human health risk assessment of pharmaceuticals and personal care products in plant tissue due to biosolids and manure amendments, and wastewater irrigation. Environ Int 75, 223–233. https://doi.org/10.1016/j.envint.2014.11.020.
[115] Prosser, R.S., Trapp, S., Sibley, P.K., 2014. Modeling uptake of selected pharmaceuticals and personal care products into food crops from biosolids-amended soil. Environ Sci Technol 48 (19), 11397–11404. https://doi.org/10.1021/es503067v.
[116] PubChem, 2021. New data content and improved web interfaces. Nucleic Acids Res 49 (D1), D1388–D1395.
[117] Qiu, Y., Liu, L., Xu, C., Zhao, B., Lin, H., Liu, H., Xian, W., Yang, H., Wang, R., Yang, X., 2024. Farmland's silent threat: Comprehensive multimedia assessment of micropollutants through non-targeted screening and targeted analysis in agricultural systems. J Hazard Mater 476, 135064. https://doi.org/10.1016/j.jhazmat.2024.135064.
[118] Rabølle, M., Spliid, N.H., 2000. Sorption and mobility of metronidazole, olaquindox, oxytetracycline and tylosin in soil. Chemosphere 40 (7), 715–722. https://doi.org/10.1016/S0045-6535(99)00442-7.
[119] Ramakrishnan B., Megharaj M., Venkateswarlu K., Sethunathan N., Naidu R. 2011. Mixtures of environmental pollutants: effects on microorganisms and their activities in soils. In: Reviews of Environmental Contamination and Toxicology, Volume 211, pp.63-120. https://doi.org/10.1007/978-1-4419-8011-3_3.
[120] Romero, F., Labouyrie, M., Orgiazzi, A., Ballabio, C., Panagos, P., Jones, A., Tao, D., 2024. Soil health is associated with higher primary productivity across *Europe*. Nat Ecol Evol 8 (10), 1847–1855. https://doi.org/10.1038/s41559-024-02511-8.
[121] Sabourin, L., Al-Rajab, A.J., Chapman, R., Lapen, D.R., Topp, E., 2011. Fate of the antifungal drug clotrimazole in agricultural soil. Environ Toxicol Chem 30 (3), 582–587. https://doi.org/10.1002/etc.425.
[122] Salazar-Ledesma, M., Prado, B., Zamora, O., Siebe, C., 2018. Mobility of atrazine in soils of a wastewater-irrigated maize field. Agric Ecosyst Environ 255, 73–83. https://doi.org/10.1016/j.agee.2017.12.018.
[123] Sarmah, R., Bhagabati, S.K., Dutta, R., Nath, D., Pokhrel, H., Mudoi, L.P., Ingtipi, L., 2020. Toxicity of the synthetic phenolic antioxidant BHT in zebrafish embryo. Aquac Res 51 (9), 3839–3846. https://doi.org/10.1111/are.14732.

[124] Schulze, S., Zahn, D., Montes, R., Rodil, R., Quintana, J.B., Knepper, T.P., Reemtsma, T., Berger, U., 2019. Occurrence of emerging persistent and mobile organic contaminants in European water samples. Water Res 153, 80–90. https://doi.org/10.1016/j.watres.2019.01.008.

[125] Schwarz, S., Gildemeister, D., Hein, A., Schröder, P., Bachmann, J., 2021. Environmental fate and effects assessment of human pharmaceuticals: lessons learnt from regulatory data. Environ Sci Eur 33 (1), 68. https://doi.org/10.1186/s12302-021-00503-0.

[126] Schymanski, E.L., Jeon, J., Gulde, R., Fenner, K., Ruff, M., Singer, H.P., Hollender, J., 2014. Identifying small molecules via high-resolution mass spectrometry: communicating confidence. Environ Sci Technol 48 (4), 2097–2098. https://doi.org/10.1021/es5002105.

[127] Selim, H.M., 2003. Retention and runoff losses of atrazine and metribuzin in soil. J Environ Qual 32 (3), 1058–1071. https://doi.org/10.2134/jeq2003.1058.

[128] Shao, Y., Yang, K., Jia, R., Tian, C., Zhu, Y., 2018. Degradation of triclosan and carbamazepine in agricultural and garden soils amended with composted sewage sludge. Int J Environ Res Public Health 15 (11), 2557. https://doi.org/10.3390/ijerph15112557.

[129] Shenker, M., Harush, D., Ben-Ari, J., Chefetz, B., 2011. Uptake of carbamazepine by cucumber plants related to irrigation with reclaimed wastewater. Chemosphere 82 (6), 905–910. https://doi.org/10.1016/j.chemosphere.2010.10.052.

[130] Shija, G.E. (2023). *Environmental Fate of Ivermectin and its biological metabolites in Soils…* (Report/Thesis).

[131] Sikder, A., Pearce, A.K., Parkinson, S.J., Napier, R., O'Reilly, R.K., 2021. Recent trends in advanced polymer materials in agriculture-related applications. ACS Appl Polym Mater 3 (3), 1203–1217. https://doi.org/10.1021/acsapm.0c00982.

[132] Srinivasan, P., Sarmah, A.K., 2014. Dissipation of sulfamethoxazole in pasture soils as affected by soil and environmental factors. Sci Total Environ 479, 284–291. https://doi.org/10.1016/j.scitotenv.2014.02.014.

[133] Statista (2024). Average annual rainfall in the UK from 2001 to 2023. ⟨https://www.statista.com/statistics/322810/average-rainfall-in-the-united-kingdom-uk/⟩.

[134] Stoob, K., Singer, H.P., Mueller, S.R., Schwarzenbach, R.P., Stamm, C.H., 2007. Dissipation and transport of veterinary sulfonamide antibiotics after manure application to grassland in a small catchment. Environ Sci Technol 41, 7349–7355. https://doi.org/10.1021/es070840e.

[135] Straub, J.O., Caldwell, D.J., Davidson, T., D'Aco, V., Kappler, K., Robinson, P.F., Simon-Hettich, B., Tell, J., 2019. Environmental risk assessment of metformin and guanylurea. I. Environmental fate. Chemosphere 216, 844–854. https://doi.org/10.1016/j.chemosphere.2018.10.036.

[136] Thiele-Bruhn, S., Beck, I.C., 2005. Effects of sulfonamide and tetracycline antibiotics on soil microbial activity and biomass. Chemosphere 59 (4), 457–465. https://doi.org/10.1016/j.chemosphere.2005.01.023.

[137] Tóth, G., Jones, A., Montanarella, L., 2013. LUCAS Topsoil Survey. Methodology, data and results. Publications Office of the European Union, Luxembourg. https://doi.org/10.2788/97922.

[138] UK Parliament (2021). Written evidence submitted by DEFRA. ⟨https://committees.parliament.uk/writtenevidence/138802/html/⟩.

[139] Vonberg, D., Hofmann, D., Vanderborght, J., Lelickens, A., Köppchen, S., Pütz, T., Burauel, P., Vereecken, H., 2014. Atrazine soil core residue analysis from an agricultural field 21 years after its ban. J Environ Qual 43 (4), 1450–1459. https://doi.org/10.2134/jeq2013.12.0497.

[140] Wakim, L.M., Descat, A., Occelli, F., Deram, A., Goossens, J.F., 2024. Detection of 13 emerging soil pollutants using dual extraction (QuEChERS & SPE) and LC-MS/MS. MethodsX 12, 102771. https://doi.org/10.1016/j.mex.2024.102771.

[141] Walenna, M.A., Hanami, Z.A., Hidayat, R., Damayanti, A.D., Notodarmojo, S., Caroles, L., 2024. Examining soil microplastics: prevalence and consequences across varied land use contexts. Civ Eng J 10 (4), 1265–1291. https://doi.org/10.28991/CEJ-2024-010-04-017.

[142] Wang, H., Li, X., Sun, Z., Cao, X., Zhang, J., Chen, Q., Ma, R., 2024. The influence of pH on sulfamethoxazole in soil systems. Water Air Soil Pollut 235 (12), 1–13. https://doi.org/10.1007/s11270-024-06649-1.

[143] Wang, C., Teppen, B.J., Boyd, S.A., Li, H., 2012. Sorption of lincomycin at low concentrations from water by soils. Soil Sci Soc Am J 76 (4), 1222–1230. https://doi.org/10.2136/sssaj2011.0408.

[144] Wei, R., Ge, F., Zhang, L., Hou, X., Cao, Y., Gong, L., Chen, M., Wang, R., Bao, E., 2016. Occurrence of 13 veterinary drugs in animal manure-amended soils in Eastern China. Chemosphere 144, 2377–2383. https://doi.org/10.1016/j.chemosphere.2015.10.126.

[145] White, A., 2016. Atrazine—a case of discrediting science. Sci Educ N 65 (1), 22.

[146] WRAP (2023). *Anaerobic digestion and composting: Latest industry survey report (new summaries).* ⟨https://www.wrap.ngo/resources/report/anaerobic-digestion-and-composting-latest-industry-survey-report-new-summaries⟩.

[147] Wu, J., Jiang, Y., He, R., Liu, Z., Zhang, X., Wang, W., Kong, W., Wang, G., Wu, Y., 2024. Adsorption/desorption of enrofloxacin in farmland soil as a function of pH and coexisting ions. Environ Geochem Health 46 (9), 363. https://doi.org/10.1007/s10653-024-02143-8.

[148] Wu, C., Spongberg, A.L., Witter, J.D., Fang, M., Ames, A., Czajkowski, K.P., 2010. Detection of pharmaceuticals and personal care products in agricultural soils receiving biosolids application. CLEAN Soil Air Water 38 (3), 230–237. https://doi.org/10.1002/clen.200900263.

[149] Wu, J., Wang, J., Li, Z., Guo, S., Li, K., Xu, P., Ok, Y.S., Jones, D.L., Zou, J., 2023. Antibiotics and antibiotic resistance genes in agricultural soils: a systematic analysis. Crit Rev Environ Sci Technol 53 (7), 847–864. https://doi.org/10.1080/10643389.2022.2094693.

[150] Wu, Y., Williams, M., Smith, L., Chen, D., Kookana, R., 2012. Dissipation of sulfamethoxazole and trimethoprim from manure-amended soils. J Environ Sci Health Part B 47 (4), 240–249. https://doi.org/10.1080/03601234.2012.636580.

[151] Yan, Z., Feng, C., Leung, K.M., Luo, Y., Wang, J., Jin, X., Wu, F., 2023. Geographical distribution, bioaccumulation and ecological risks of organophosphate esters. J Hazard Mater 445, 130517. https://doi.org/10.1016/j.jhazmat.2022.130517.

[152] Yang, L., Wu, L., Liu, W., Huang, Y., Luo, Y., Christie, P., 2018. Dissipation of antibiotics in three agricultural soils after repeated biosolids application. Environ Sci Pollut Res 25, 104–114. https://doi.org/10.1007/s11356-016-8062-6.

[153] Yu, C., Bi, E., 2019. Adsorption site-dependent transport of diclofenac in water-saturated minerals and reference soils. Chemosphere 236, 124256. https://doi.org/10.1016/j.chemosphere.2019.06.226.

[154] Yu, Y., Mo, W., Zhu, X., Yu, X., Sun, J., Deng, F., Jin, L., Yin, H., Zhu, L., 2022. Biodegradation of tricresyl phosphate isomers by a novel microbial consortium and toxicity of its major products. Sci Total Environ 828, 154415. https://doi.org/10.1016/j.scitotenv.2022.154415.

[155] Zhang, Y.L., Lin, S.S., Dai, C.M., Shi, L., Zhou, X.F., 2014. Sorption–desorption and transport of trimethoprim and sulfonamides in agricultural soil: Effects of soil type, DOM, and pH. Environ Sci Pollut Res 21 (9), 5827–5835. https://doi.org/10.1007/s11356-014-2493-8.