Environmental Pollution 315 (2022) 120367

Contents lists available at ScienceDirect

Environmental Pollution

journal homepage: www.elsevier.com/locate/envpol

Modeling the fate of ionizable pharmaceutical and personal care products (iPPCPs) in soil-plant systems: pH and speciation

Abrar Shahriar[a], David Hanigan[a], Paul Verburg[b], Krishna Pagilla[a], Yu Yang[a],*

[a] Department of Civil and Environmental Engineering, University of Nevada, Reno, 1664 N Virginia St, Reno, NV, 89557, USA
[b] Department of Natural Resources and Environmental Science, University of Nevada, Reno, 1664 N Virginia St, Reno, NV, 89557, USA

ARTICLE INFO

Keywords:
Reclaimed wastewater
iPPCPs
Plant uptake model
Speciation

ABSTRACT

A model was developed to simulate the pH-dependent speciation and fate of ionizable pharmaceutical and personal care products (iPPCPs) in soils and their plant uptake during thedt application of reclaimed wastewater to agricultural soils. The simulation showed that pH plays an important role in regulating the plant uptake of iPPCPs, i.e., ibuprofen (IBU; with a carboxylic group), triclosan (TCS; phenolic group), and fluoxetine (FXT; amine group) as model compounds. It took 89–487 days for various iPPCPs to reach the steady-state concentrations in soil and plant tissues. The simulated steady-state concentrations of iPPCPs in plant tissues at pH 9 is 2.2–2.3, 2.5–2.6, and 1.07–1.08 times that at pH 5 for IBU, TCS, and FXT, respectively. Assuming sorption only for neutral compounds led to miscalculation of iPPCPs concentrations in plant tissues by up to one and half orders magnitude. Efflux of compounds in soil, lettuce leaf, and soybean pods was primarily contributed by their degradation in soil and dilution due to plant tissue growth. Overall, the results demonstrated the importance of considering pH and speciation of iPPCPs when simulating their fate in the soil-plant system and plant uptake.

1. Introduction

The fate of pharmaceuticals and personal care products (PPCPs) in soil-plant systems irrigated with reclaimed wastewater has attracted growing interests because human beings can be exposed to PPCPs directly through consumption of food crops and indirectly through consumption of products of animals fed with forage crops that have taken up wastewater-derived PPCPs (Archer et al., 2017; Prosser & Sibley, 2015; Qin et al., 2015). These compounds can cause a range of adverse effects to humans and plants, including phytotoxicity and endocrine disruption (Fu et al., 2019; Li et al., 2015; Poustie et al., 2020). Reliable evaluation of the environmental and human health effects of PPCPs in water reused for agriculture requires comprehensive understanding for the fate of PPCPs in soil-plant systems.

Many PPCPs have ionizable functional groups, i.e., carboxylic, phenolic, and amine groups, and can deprotonate/protonate in soils depending on soil pH. Normalized octanol-water partitioning coefficients (D_{OW}) based on pK_a and pH have been used by regulatory organizations and scientific studies to evaluate the fate and plant uptake of soil-bound PPCPs, which assumes that the charged form of ionizable compounds is not absorbed by soil particles (Government of Canada, 2013; Tanoue et al., 2012; Wu et al., 2013). However, such sorption has been shown to occur for most ionizable PPCPs (iPPCPs) (Hyland et al., 2015; Li et al., 2020; Wu et al., 2013). iPPCPs can interact with soil organic matter or minerals through electrostatic interactions and/or surface complexation with substantial sorption coefficients (MacKay & Vasudevan, 2012; Miller et al., 2016). For instance, the sorption coefficient of triclosan (pK_a = 7.9) was measured to be 210.3 L/kg at pH 4, and 126 L/kg at pH 8, but was estimated to be 93 L/kg at pH 8 assuming no sorption for charged species (Wu et al., 2009).

Without appropriately accounting for the fate of iPPCPs in soil-plant systems, their potential risk to human health can be substantially miscalculated. Plant uptake factor (Concentration in plant/Concentration in soil) of gemfibrozil and triclosan by lettuce (*Lactuca sativa*) was found to be 5.6×10^{-2} and 4.53×10^{-1}, respectively at pH 6.5, even though their log K_{ow} was similar with the values of 4.77 and 4.76 and plant uptake factor would be close if K_{ow} is the dominant factor (Wu et al., 2013). Malchi et al. (2014) studied the plant uptake of ten iPPCPs including carbamazepine, lamotrigine, and others, and the plant uptake factor varied 0.32–3.9, which is not well explained by their log K_{ow}. In a recent study, Shariq et al. (2021) examined the maize uptake of a range of chemical additives in hydraulic fracturing produced water, including

---

* Corresponding author.
E-mail address: yuy@unr.edu (Y. Yang).

https://doi.org/10.1016/j.envpol.2022.120367
Received 26 April 2022; Received in revised form 29 September 2022; Accepted 2 October 2022

many organic amines such as didecyldimethylammonium chloride (DDAC) and tetramethylammonium chloride (TMAC) and indicated the importance of ionization in regualting their plant uptake: the LogK_{ow} of DDAC (2.59) was much higher than TMAC (−4.18) (referred to neutral amine), but the uptake of TMAC was greater than DDAC. Such difference indicates that other factors beyond log K_{ow} regulate their plant uptake. Instead of K_{ow} alone, dissimilar sorption of ionized/neutral forms of iPPCPs may play an important role in regulating their plant uptake as well as other processes in soil. Most of the previous modeling studies focused on the neutral species of the iPPCPs and didn't address the impact of pH on the speciation and fate of iPPCPs in soil-plant system during the wastewater reuse for agriculture (Chen et al., 2013; Clarke et al., 2018; García-Santiago et al., 2017; Hurtado et al., 2016). Few recent modeling studies considered ionized species, but the roles of sorption of ionic and neutral part of the iPPCPs on both soil organic matter and minerals in the partitioning of iPPCPs and their plant uptake as well as the impact of soil pH were not extensively studied (Brunetti et al., 2022; Delli Compagni et al., 2020a, 2020b; García et al., 2019; Polesel et al., 2015; Prosser et al., 2014b; Trapp, 2009).

We previously developed a model for calculating the fate of only neutral species of PPCPs during irrigation of agricultural soils with reclaimed wastewater (Shahriar et al., 2021). For the current study, the partitioning of iPPCPs between soil and water was calculated considering the dissimilar sorption of neutral and ionic species of iPPCPs. Modeling was conducted using four scenarios: only K_{ow} was considered, and speciation was ignored in scenario 1; no sorption for charged species and fraction of neutral species were calculated in scenario 2; sorption of both neutral and charged species by only organic matter were counted in scenario 3; and sorption of both neutral and charged species by both soil organic matter and minerals were calculated in scenario 4 (Table 1). Results calculated from different scenarios were compared to analyze the impact of different species and processes (sorption by organic matter/mineral) on the plant uptake of iPPCPs. Impact of pH on the fate and plant uptake of iPPCPs was systematically evaluated; importance of consideration for speciation in modeling the fate of iPPCPs was uncovered by comparing different scenarios. With major focus on forward simulation and no calibration based on experimental data, the calculated plant uptake factors of iPPCPs were comprehensively compared and validated with the available literature data in the matrix of pH-soil organic carbon content for the screening evaluation.

Table 1
Summary of the key features for the scenarios used in the modeling development in this study.

Scenario	Considerations and Approaches			
	Speciation	Sorption sites	Sorption calculation	Plant partition coefficient
Scenario 1	No speciation was considered	Soil organic carbon only	Based on soil properties and log K_{ow} of iPPCPs	Based on plant physiological parameters, and log K_{ow} of iPPCPs
Scenario 2	Only neutral species of iPPCPs were considered.	Soil organic carbon only	Based on soil properties and log D_{ow} of iPPCPs	Based on plant physiological parameters, and log D_{ow} of iPPCPs
Scenario 3	Both ionic and neutral species were considered	Soil organic carbon only	Based on soil properties, pH, log K_{ow} and pK_as of iPPCPs	Based on plant and soil physiological parameters, pH, and log K_{ow} and pK_as of iPPCPs
Scenario 4	Both ionic and neutral species were considered	Both soil organic carbon and soil minerals	Based on soil properties, pH and log K_{ow} and pK_as of iPPCPs	Based on plant and soil physiological parameters, pH, and log K_{ow} and pK_as of iPPCPs

2. Methods

2.1. Model framework

Based on similar principles developed and used in previous studies (Legind et al., 2011; Trapp, 2007), cascading-boxes (soil-plant root-stem-leaf/pods) modeling was used to calculate the fate of iPPCPs in the soil-plant system (Supplementary Information (SI), Fig. S1). The soil-plant modeling approach accounted for the critical processes of iPPCP, including degradation, leaching, and volatilization of organic compounds in soils, as well as uptake, translocation, and growth dilution by agricultural plants (Legind et al., 2011; Shahriar et al., 2021; Trapp, 2007). Calculation for the plant uptake and translocation of iPPCPs mainly accounted for their advection with water and partitioning with plant tissues (i.e., passive transport), but the detailed processes, including cross-membrane uptake and subcellular distribution, were not included. This box model did not address the reactive transport processes within the soil matrix, and therefore the roles of iPPCPs speciation in their advection-diffusion-transport require further investigation.

We selected three model iPPCPs: ibuprofen (IBU), fluoxetine (FXT), and triclosan (TCS), which represent compounds with a range of physicochemical properties and ionizable functional group, e.g., carboxylic (IBU), phenolic (TCS), and amine (FXT) groups (SI, Table S1). These compounds also represent three different groups of pharmaceuticals and personal care products: nonsteroidal anti-inflammatory (IBU), antidepressant (FXT), and anti-microbial (TCS) respectively. Finally, these three iPPCPs are commonly found in reclaimed wastewater even after conventional treatment (Sharma et al., 2020; Verlicchi et al., 2012; Yang et al., 2011). Calculation for these model compounds can represent the roles of different species in soil-plant systems and impact of pH for a larger range of iPPCPs. We selected two relevant agricultural plants i.e., lettuce (*Lactuca sativa* L.) and soybean (*Glycine max*) as model plants for this study, because they have different physiological parameters including water and lipid content and growth rate (Legind et al., 2011; Prosser et al., 2014a).

Four scenarios were examined (Table 1). For the first scenario, sorption was calculated based on K_{ow}. In the second scenario, sorption calculated based on D_{ow} considering only neutral species from ionization. For the third scenario, dissimilar sorption of both neutral and charged species was calculated only accounting for sorption by soil organic matter. In the fourth scenario, which is most representative for environmental conditions, sorption of ionic and neutral species of iPPCPs by both soil organic matter and minerals was calculated. Comparison of the four scenarios can uncover the importance of considering the speciation of iPPCPs in simulating their fate in soil-plant system and plant uptake.

2.2. Time-dependent concentrations calculation

Time-dependent concentrations of iPPCPs in different compartments (soil, plant root, stem, leaf, pods) were calculated based on differential equations, with the primary equations for scenario 4 presented here. Simulation time range for all the scenarios was set to be 1000 days with different harvest periods for two agricultural plants (55 and 81 days for lettuce and soybean, respectively), so that the steady-state concentrations of iPPCPs after long-term (multiple harvest cycles, common for realistic water reuse) reclaimed water reuse can be calculated. Other equations for scenarios 1, 2, and 3 are provided in the SI. The soil concentrations were calculated accounting for the input by wastewater irrigation, and output by plant root uptake, leaching, volatilization, and degradation:

$$\frac{dC_s}{dt} = \frac{V_w C_w}{M_s} - \left(\frac{Q_R}{K_d M_s} + \frac{Q_{inf}}{K_d M_s} + \frac{A_s F_{sa} \alpha}{M_s} + K_{deg} \right) \times C_s \qquad (1)$$

V_w (L/d) is the volume of the reclaimed wastewater applied per day,

C_w (mg/kg) is the concentration of iPPCPs in the applied reclaimed wastewater, M_S (kg) is the mass of soil available for soil uptake, $Q_{\inf}$ (L/d) is the leaching rate, K_d (L/kg) is the soil water partition coefficient, Q_R (L/d) is the transfer rate of water from soil to root, A_s (m^2) is the soil surface area, F_{sa} (kg/m^2/d) is soil to air transport flux, α is the fraction of neutral species of the iPPCPs, K_{deg} (1/d) is the degradation rate constant of iPPCPs in soil, and C_S (mg/kg) is the concentration of iPPCPs in soil. Average leaching rate ($Q_{\inf}$) of typical agricultural soils was used (Legind et al., 2011). Transfer rate (Q_R) was calculated based on growth and transpiration coefficient (Legind et al., 2011) (SI, Text S1).

The root concentrations were calculated accounting for uptake from soil, and output by translocation to stem and growth dilution:

$$\frac{dC_R}{dt} = \frac{Q_R}{K_d M_R} C_s - \left(\frac{Q_R}{K_{RW} M_R} + K_{G,R}\right) \times C_R \tag{2}$$

K_{RW} (L/kg) is the root-water partition coefficient, M_R (kg) is the root mass, $K_{G,R}$ (1/d) is the growth rate of the root, and C_R (mg/kg) is the concentration of iPPCPs in root of the plant.

The concentrations in other plant tissues, i.e., stem, leaf, and pods, were calculated based on similar principles. Details of the equations of each soil-plant compartment for all scenarios are compiled in SI.

2.3. Key processes

2.3.1. Soil-water partitioning

For scenario 1 and 2, K_d was calculated using equations (3) and (4), respectively.

$$K_d = f_{OC} \times K_{OC} \tag{3}$$

$$K_d = \alpha \times f_{OC} \times K_{OC} \tag{4}$$

For scenario 3 and 4, K_d was constructed assuming contribution from both neutral and ionic species of the compounds (Equations. 5 and 6) (Table S2). However, for scenario 3, adsorption by soil minerals was not considered (Equation (4)).

$$K_d = f_{neutral} \times K_{d,Neutral} + f_{charged} \times K_{d,charged} \tag{5}$$

$$K_d = f_{neutral} \times \left(f_{oc} K_{oc,Neutral} + f_{mineral} K_{mineral,Neutral}\right) + f_{charged} \times \left(f_{oc} K_{oc,charged} + f_{mineral} K_{mineral,charged}\right) \tag{6}$$

For TCS and IBU, for the neutral form, sorption was governed by hydrophobic interactions, and for the negatively charged ion, electrostatic repulsive interactions and surface complexation between carboxylic/phenolic group and the charged mineral surface dominate (SI, Fig. S2). The calculation was based on K_{OC} and $K_{mineral}$ (for both neutral and charged species), where K_{OC} was calculated based on linear regression considering pH-dependent speciation and species-specific partition coefficients (Equation (7)) (Franco et al., 2009). $K_{mineral}$ was calculated based on the experimental data of the sorption on soil mineral by Behera et al. (2012) and Behera et al. (2010) for IBU and TCS, respectively.

$$K_{OC} = \frac{10^{0.54 \log K_{ow} + 0.11}}{1 + 10^{(pH - pK_a - 0.6)}} + \frac{10^{0.11 \log K_{ow} + 1.54}}{1 + 10^{(pK_a - pH + 0.6)}} \tag{7}$$

For FXT, sorption was governed by the hydrophobic interactions (neutral form) and electrostatic attraction (positive charge) with soil organic matter and minerals. The calculation was based $K_{d,Neutral}$ and $K_{d,charged}$ (Equations (8)–(10)) (Droge & Goss, 2013a). For $K_{d,Neutral}$, the interaction was dominated by sorption to organic carbon and was calculated using Equations (8) and (9) (Droge & Goss, 2013a; Droge & Goss, 2013b). $K_{d,charged}$ was modeled considering sorption of cation on soil organic matter was normalized to f_{oc}, whereas sorption on minerals was normalized to cation exchange capacity (Equation (10)). Fragment based approach was applied to determine $K_{CEC,Clays}$ and $D_{OC,IE}$ of FXT (Table S2).

$$K_{d,Neutral} = K_{oc,Neutral} \times f_{oc} \tag{8}$$

$$\log K_{OC,Neutral} = 0.37 \times \log K_{ow} + 1.7 \tag{9}$$

$$K_{d,charged} = K_{CEC,Clays} CEC_{Clay} + f_{OC} D_{OC,IE} = K_{CEC,Clays}\left(CEC_{Soil} - 3.4 \times f_{OC}\right) + f_{OC} D_{OC,IE} \tag{10}$$

2.3.2. Partition coefficient between plant compartments and water

Partition coefficients between plant compartments and the pore water was estimated based on the lipid-water partition coefficient (K_{lipid}) (Equation (11)). K_{lipid} for neutral species was described using the polyparameter linear free energy relationships (pp-LFERs) model (Equation (12)) (Geisler et al., 2012).

$$K_{PW} = f_{lipid} \times K_{lipid} \tag{11}$$

K_{PW} refers to the partition coefficient between plant compartments and pore water, e.g., K_{RW} for root-water partition coefficient. f_{lipid} is the fraction of lipid content is the plant part.

$$\log K_{lipid} = 0.70 \times E - 1.08 \times S - 1.72 \times A - 4.14 \times B + 4.11 \times V - 0.07 \tag{12}$$

where, E, S, A, B, and V denotes the excess molar refraction, dipolarity/polarizability, solute hydrogen-bond acidity, hydrogen-bond, and McGowan volume of solute, respectively. The value of the descriptor for each iPPCPs are listed in the SI (Table S3). In case of calculating K_{PW}, the ionic species were ignored as their contribution was negligible.

2.3.3. Volatilization

The soil-to-air transport flux density was determined by the equations developed by D. Mackay (Equation (13)) (Mackay, 1979).

$$F_{sa} = \frac{f_s}{\left(\frac{RT}{K_{13}} + \frac{L_3}{\left(\frac{RT}{D_{air}} + \frac{H}{D_{water}}\right)}\right)} \tag{13}$$

where K_{13}, L_3, D_{air} and D_{water} are air-soil mass transfer coefficient (m/h), diffusion path length in soil (m), molecular diffusivities in air and water (m^2/h), respectively. H is Henry's constant (Pa-m^3/mol). R and T represent the gas constant in J/mol-K and absolute temperature in K. f_s is the fugacity coefficient in soil. Further details of the model can be found in the SI.

2.4. Parameterization

All the parameters used in the calculations are provided with associated references in Table S4 to S6 (SI). Parameters selected for our model are dependent on the specific compound, plant, or properties of the soil. The degradation rate constant of iPPCPs in soil was based on the data found in literatures. The logK_{ow} for each iPPCP was obtained from EPI suite, and the pK_a values were collected from literatures (US EPA, 2012). Input concentrations of compounds in wastewater were based on the average concentration of iPPCPs found in treated wastewater (Shahriar et al., 2021). Specific environment-dependent parameters, e.g., soil density, leaching rate, fraction of organic carbon/minerals, were determined based on data from literature (Table S4).

Calculations were performed with Matlab 2019b (The MathWorks Inc., Natick, MA, USA), and statistical analyses were conducted with IBM SPSS Statistics version 25.0 (IBM Corp., Armonk, NY, USA) and R base library (R Core Team, 2021).

3. Results

3.1. Time to reach steady state iPPCPs concentrations in soil-plant systems

Concentrations of three iPPCPs in soils and four compartments of lettuce (*Lactuca sativa*) and soybean (*Glycine max*) reached steady state after 89–487 days based on model output (Figs. S3–S9). Compared to scenario 1, consideration of speciation and their sorption by both organic matter and minerals in soils under scenario 4 did not substantially change the period for iPPCPs to reach steady-state concentrations. Under scenario 4, with pH from 5 to 9, it took 218–221 days for TCS in lettuce leaf to reach steady state, near the end of fourth harvest cycle (Fig. S6). The duration to reach steady-state concentrations varied among the target model compounds but did not differ much between two plants. For example, the concentration of IBU in soils reached steady state after 297 and 319 days for system with lettuce and soybean, respectively, under scenario 1 while for compound TCS it took 143 and 133 days, respectively, to reach the steady state. The time for the soil concentration to stabilize depended primarily on the degradation rate constant of target compounds, which ranged from $5.26 \times 10^{-3} \pm 2.54 \times 10^{-3}$ d^{-1} for IBU and $4.44 \times 10^{-2} \pm 8.93 \times 10^{-2}$ d^{-1} for TCS.

3.2. Steady-state concentrations

Steady-state concentrations of iPPCPs in soils in the scenarios 3 and 4 did not change significantly over the pH range of 5–9 (*t*-test, $p > 0.05$) (Fig. 1, SI, Fig. S10). Conversely, under scenario 2, the concentration of IBU decreased from 5.81×10^{-4} mg/kg to 2.33×10^{-7} mg/kg in lettuce-grown soil when pH increased from 5 to 9. Under scenario 2, concentrations of FXT in lettuce-grown soils increased by four orders of magnitude, when pH increased from 5 to 9. Compared to scenario 4, calculations assuming sorption only for neutral compounds in scenario 2 led to much lower soil concentrations (pH = 9) for IBU by up to 3.6 orders of magnitude.

Steady-state concentrations of iPPCPs in the plant tissues under scenario 3 and 4 mostly increased with pH for all iPPCPs (Fig. 1, Figs. S11 and S12). For instance, steady-state concentrations of TCS in soybean root, stem, leaves, and pods increased from 1.29×10^{-6}, 2.02×10^{-8}, 5.31×10^{-10}, and 5.28×10^{-10} mg/kg at pH 5 to 3.29×10^{-6}, 5.15×10^{-8}, 1.36×10^{-9}, and 1.35×10^{-9} mg/kg at pH 9, respectively, representing approximately 2.5 times increase in concentration for all plant parts. iPPCPs concentration in plant tissues correlated with pH values (Pearson correlation coefficient of 0.92 and 0.95 for TCS and IBU, respectively for plant tissues, $p < 0.05$). When pH increased from 5 to 9, the steady-state concentrations in lettuce leaves under scenario 4 changed from 2.13×10^{-3} to 4.72×10^{-3}, 2.9×10^{-8} to 7.4×10^{-8}, 2×10^{-8} to 2.14×10^{-8} mg/kg, for IBU, TCS, and FXT, respectively. Consideration of harvest period substantially decreased the steady-state concentrations of iPPCPs in lettuce leaves and soybean pods than the calculation assuming plant continuously growing (SI, Fig. S13). For instance, the steady-state concentration changed by around 19% for lettuce leaves.

Compared to scenario 4, calculation without considering speciation in scenario 1 led to the largest difference in calculation of plant tissue concentration for FXT, which was more than two orders of magnitude higher compared to scenario 4. For FXT in soybean pods, the scenario assuming only sorption for neutral compounds (scenario 2) also overestimated concentrations by approximately seven orders of magnitude. Considering uncertainties for selected parameters (K_{deg}, $K_{\mathrm{G,R}}$, $K_{\mathrm{G,St}}$, $K_{\mathrm{G,L}}$, and f_{oc}), coefficient of variations for steady-state concentrations ranged 4.3–5.4%, 114.9–149.3%, and 12–46.6%, for IBU, TCS, and FXT, respectively, under scenario 4 at pH 7 (Fig. S14). Details about Monte-Carlo uncertainty analysis and values can be found in SI, Text S2 and

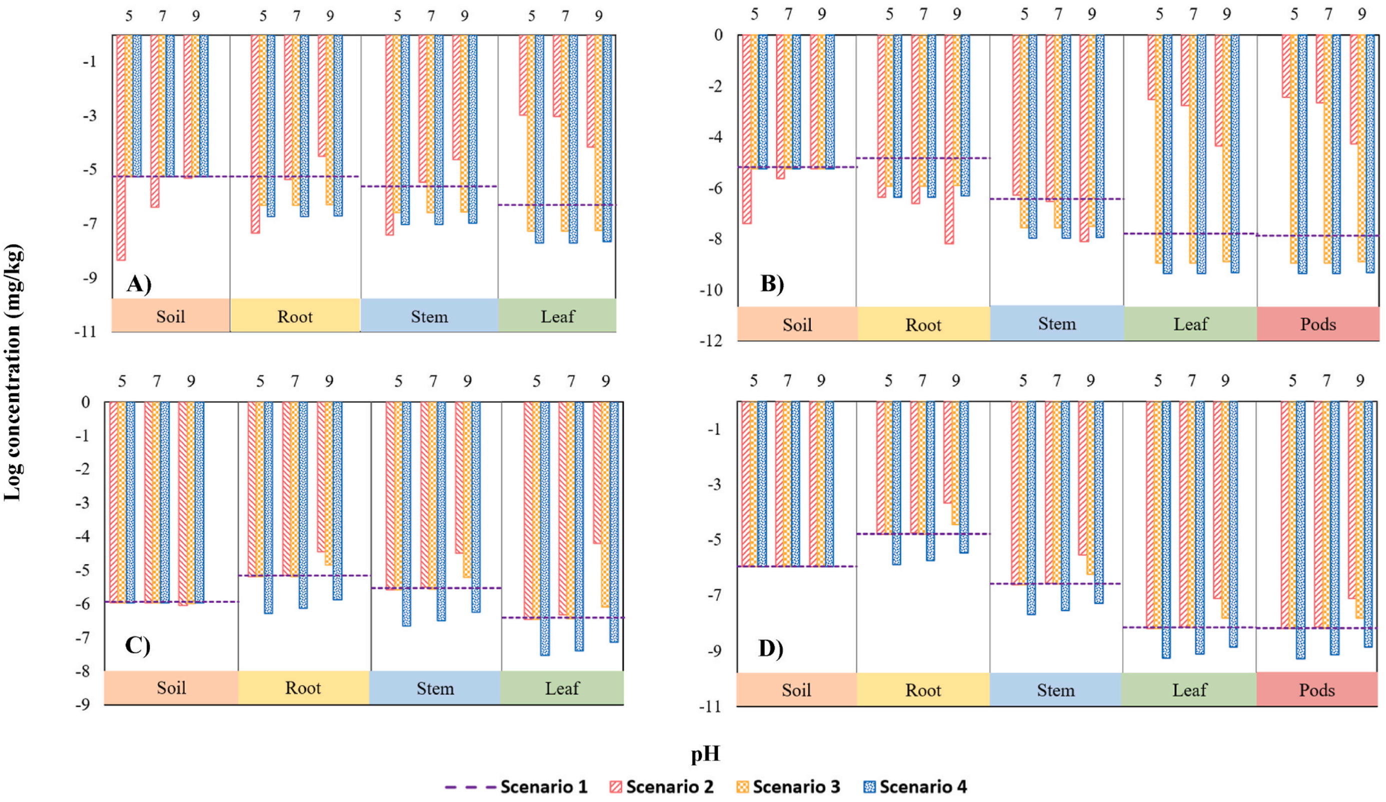

Fig. 1. Steady-state concentrations of A) FXT in lettuce-soil; B) FXT in soybean-soil; C) TCS in lettuce-soil; and D) TCS in soybean-soil systems at varying soil pH. The dashed purple line refers to the pH independent concentration for scenario 1. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

Table S7. Local sensitivity analysis (LSA) showed the importance of the degradation rate constant for determining the concentration of iPPCPs in soils: TCS in lettuce-grown soils increased by 3.8 times, when the degradation rate constant of TCS in soil decreased by factor of 10 (Fig. S15). Root concentration was less sensitive to growth rate of root with TCS in lettuce root increasing by 1.2 times, when the growth rate decreased by an order of magnitude. Furthermore, global sensitivity analysis (GSA) showed that the importance of K_{deg} to soil concentration (standardized regression coefficient of 0.35–0.78 for K_{deg} vs < 0.1 for f_{oc}, $K_{\mathrm{G,R}}$, $K_{\mathrm{G,St}}$, and $K_{\mathrm{G,L}}$), and $K_{\mathrm{G,R}}$ to plant root concentration (standardized regression coefficient of 0.42–0.69 vs < 0.3 for f_{oc}, $K_{\mathrm{G,St}}$, and $K_{\mathrm{G,L}}$), when the results need to be taken with cautions as the R^2 ranged 0.2–0.6 (Fig. S16).

3.3. Fluxes

iPPCPs concentrations in soils were predominantly regulated by their degradation. In scenario 4, the degradation of IBU in soil contributed more than 96% of its total loss, whereas plant uptake by roots contributed only 4% (Fig. 2). When pH changed from 5 to 9, the contribution from degradation only changed slightly (98%–96%). Contribution from leaching and volatilization to the total efflux from the soil was very small (<2%) in all scenarios. For scenario 2, root uptake of iPPCPs was dominant over other processes, for example contributing more than 87% of total FXT outflux at pH 9.

Overall root efflux was dominated by both root-stem transfer and growth dilution. For IBU in lettuce root at pH 7 in scenario 4, root-stem transfer contributed more to the total efflux than root growth dilution (90% compared to 10%). In scenario 1, the two effluxes contributed almost equally (52 and 48%, respectively). Similar results were observed in stems. As a comparison, transpiration of iPPCPs from the stem was very low (at least 2 orders of magnitude less than stem-leaves/pods transfer or growth dilution efflux) for all scenarios. Moreover, transpiration of iPPCPs from leaves was also negligible in all scenarios. For example, transpiration efflux was three orders of magnitude lower than the growth-dilution for TCS in scenario 4 at pH 7.

4. Discussion

4.1. Plant uptake factor and validation

Based on the calculation of soil/plant concentrations, plant uptake factors from soil to lettuce roots were calculated as a function of soil pH and f_{OC} (Fig. 3A). Higher uptake of iPPCPs from soil to plant tissue occurred at lower f_{OC} and higher pH. For instance, at pH 7, when f_{OC} was increased from 0.01 to 0.1, the uptake factor changed from 1.48 to 0.84, 0.63 to 0.31, and 0.036 to 0.02, for IBU, TCS, and FXT, respectively under scenario 4. Overall, the uptake factor was higher for IBU and TCS, primarily because of their lower sorption to soil organic carbon and minerals.

Our calculated plant uptake factors were comparable with reported values. Wu et al. (2010) observed that the plant uptake factor of FXT for soybean root is 0.15 for soil with f_{OC} of 0.016 and pH of 5.1, compared to the corresponding value of 0.036 based on our calculation under scenario 4. Calculated uptake factors of FXT under scenario 1 and 2 were much higher (1 and 4.4 compared to 0.036) than scenario 4 and the reported values. The uptake factor calculated based on scenario 3 was 0.09, similar to scenario 4, due to the minor contribution of minerals to sorption of FXT. For TCS, Prosser et al. (2014a, 2014b) reported an uptake factor of 0.68 for radish root, and 1.9 for soybean root when the plants were grown in spiked soil and biosolid-amended soil at pH 7.8, with soil organic carbon ranging from 1.76 to 2.41%. Bearing the difference in plants, this study found an uptake factor of 0.72 for lettuce root with pH 7.8 and organic carbon of 2%. Fig. 3B depicted the comparison between reported plant uptake ratio and our model prediction based on the soil pH and f_{OC} provided in those particular literature. These results were also comparable. For example, based on the data stated in Fu et al. (2016), our model predicted the uptake factor of TCS in radish skin to be 0.67 at soil pH 6.5 and f_{OC} of 2.5%, whereas the experimental result was 0.27. As shown in Fig. 3B, most of the comparison were within same order of magnitude.

Due to limited data for plant uptake of iPPCPs, especially FXT, in soils irrigated with reclaimed water, data for plant uptake from agricultural soils amended with biosolids were also used. Although biosolids

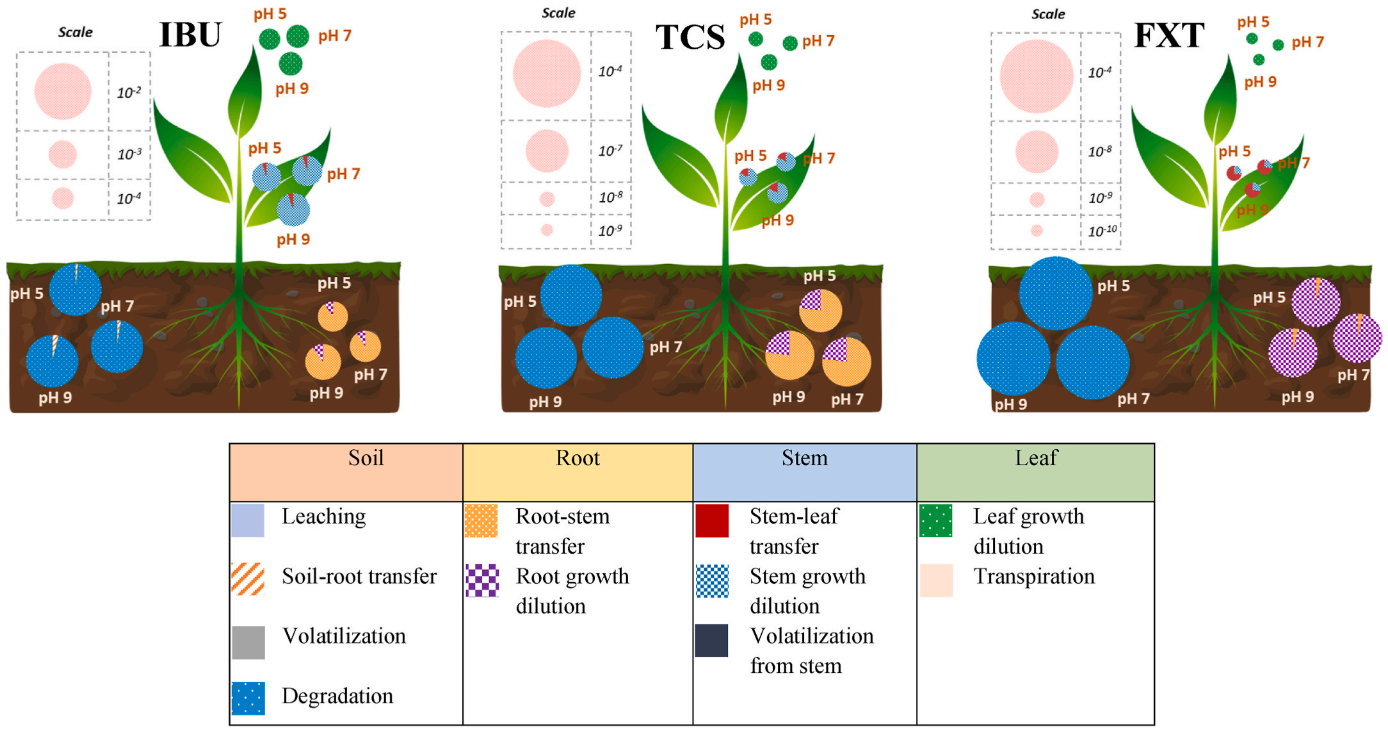

Fig. 2. Effluxes of different compartments of the soil-plant system for three iPPCPs at scenario 4. The size of each pie chart corresponds to total efflux in mg/d.

addition may alter soil physicochemical properties more compared to reclaimed water irrigation, in general the addition of biosolid is moderate ($\leq$10% dw/dw), and there was good match between our prediction and reported values of plant uptake ratios for soils with reclaimed water irrigation and biosolid application (Fig. 3) (Fu et al., 2016; Holling et al., 2012; Pannu et al., 2012). More accurate simulation for the fate of iPPCPs in biosolid-amended soils requires consideration of impact of biosolid amendment on soil physicochemical properties.

Few studies accounted for the contribution of plant proteins to the sorption of organic compounds, by estimating their K_{pw} using human serum albumin partition coefficient (K_{HSA}) (Brunetti et al., 2022; García et al., 2019). Considering typical fraction of proteins in plant tissues (1.3%) vs lipid (2–2.5%), the simulated plant uptake of iPPCPs was not changed substantially when proteins were considered (SI, Text S3). For example, the concentration of IBU (K_{HSA} value of 65.66 (Wanat et al., 2021)) in root, stem, and leaves at pH 7 under scenario 4 changed from 7.27×10^{-4}, 6.96×10^{-4}, and 3.02×10^{-3} mg/kg to from 7.67×10^{-4}, 7.31×10^{-4}, and 2.95×10^{-3} mg/kg, respectively. Furthermore, K_{HSA} values of the same compound was found to differ by orders of magnitude from literature to literature, and therefore investigations were warranted to constrain the contribution of proteins and other components in plant tissues to uptake of iPPCPs.

4.2. *Impact of pH on steady-state concentration in soils and plant tissues*

The impact of pH on the steady-state concentrations of iPPCPs in soils and plant tissues was caused by their speciation and associated sorption of their different species, best represented in scenario 4. For IBU, (carboxylic acid with pK_a of 4.4) in scenario 4, increasing soil pH from 4 to 10 reduced the calculated K_d from 18.75 to 4.9 L/kg (SI, Figs. S17 and S18). The sorption of neutral IBU was higher than that for the dissociated IBU^- likely because of similar charges between IBU^- and the mineral surface, and charge repulsion (Miller et al., 2016). When pH increased from 4 to 10, the K_{OC} decreased from 176.85 to 94.78 L/kg, while the $K_{mineral}$ decreased from 16.9 to 3.9 L/kg, based on the empirical fitting to the reported sorption on kaolinite (Behera et al., 2012). Calculations based on sorption model and empirical fittings agree with experimental observation of sorption to soils. For instance, the sorption of IBU on a clay soil at pH 4 (where the neutral component is dominant and with $K_d = 1.22$ L/kg) was higher than its sorption at pH 8 (with $K_d = 0.42$ L/kg) (Hiller & Šebesta, 2017). In the pH range of 4–10, the calculated K_d, accounting for the sorption of deprotonated and protonated forms by both soil organic matter and minerals, changed from 65.8 to 14.3 L/kg, and 4797.4 to 2658.2 L/kg, for TCS ($pK_a = 7.9$), and FXT ($pK_a = 10.09$), respectively. These calculated patterns were supported by the measurement of sorption coefficient for soils with different pH, showing that the sorption of TCS, and FXT reduced with increasing pH (Behera et al., 2010, 2012; Figueroa et al., 2004). Compared to scenario 4, the calculation in scenario 2 showed the largest difference between estimated and measured soil concentrations. For instance, scenario 2 underestimated the concentration by 3.6 orders of magnitude for IBU in lettuce-grown soils at pH 9, while the IBU efflux was overestimated mainly for the leachate and lettuce root uptake due to the underestimation of sorption coefficient ignoring the sorption of charged IBU^-.

Despite the strong response of sorption of iPPCPs to pH, the steady-

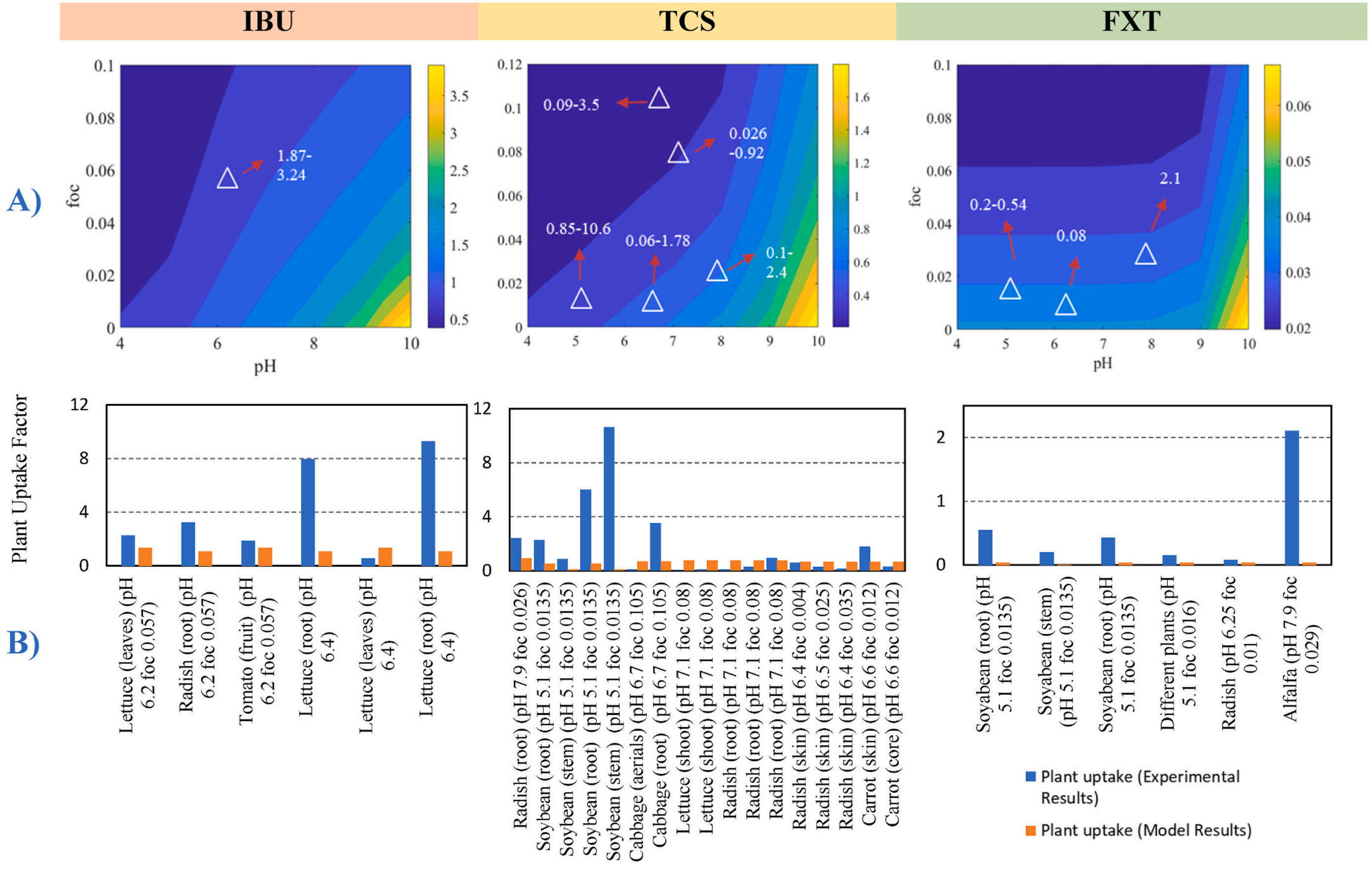

Fig. 3. A) Contour plot of plant root uptake factor of IBU, TCS, and FXT, for scenario 4 for lettuce plant. Z-axis represents the unitless plant uptake factor. The markers refer to the literature reported uptake factors. **B)** Comparison between uptake factor of soil and different plant tissue of iPPCPs of experimental results collected from literature and the calculated model values. The plant tissue parts, soil pH and the fraction of organic carbon of the soil are cited in the parentheses. (References of the literature (Carter et al., 2014; Cortés et al., 2013; Fu et al., 2016; Holling et al., 2012; Hurtado et al., 2016; Kovacs et al., 2021; Pannu et al., 2012; Prosser et al., 2014a; Sharma et al., 2020; Wu et al., 2010)).

state soil concentration was marginally affected by the pH in this dynamic soil-plant system, mainly because the steady-state concentration was primarily determined by the balance of influx through wastewater irrigation and efflux primarily due to degradation, both of which are independent of soil pH. It is however possible that sorption can affect the bioavailability for microbes and subsequent degradation. In current modeling, soil degradation was assumed to be independent of compound speciation, sorption, and soil pH, but in general the freely dissolved species are the major available component for microbial degradation and desorption to the solution phase was required for the microbial degradation of sorbed compounds (Chang et al., 2021; Z. Chen et al., 2015; Reichenberg & Mayer, 2006). If bioavailability was accounted for, with only the freely dissolved iPPCPs being subject to degradation, steady-state soil concentrations of iPPCPs were greater and increased substantially with soil pH. At pH 7, the steady-state concentrations of IBU, TCS, and FXT in lettuce-grown soils increased from 9.96×10^{-4} to 4.58×10^{-3}, 1.1×10^{-6} to 4.2×10^{-5}, and 5.7×10^{-6} to 7.1×10^{-5}, mg/kg, respectively when the bioavailability was taken into account (Fig. 4a). For IBU, when considering bioavailability, the steady-state concentration changed from 4.81×10^{-3} to 4.14×10^{-3}, when pH increased from 5 to 9 (Fig. 4b). There is still a limited amount of data regarding the degradation of iPPCPs under different soil chemical conditions (pH), and further studies are needed to fully incorporate bioavailability in the model presented here.

Unlike soil concentrations, generally steady-state plant tissue iPPCPs concentrations increased substantially with soil pH. Most remarkably, this was primarily due to the response of plant root uptake to the soil pH, as a result of the change in the K_d. Compared to the calculation of K_d for soils, there are more uncertainties for calculation of K_d for plant tissue, which were modeled analogous to soil organic matter to account for the interactions with both neutral and charged species. Compared to scenario 4, scenario 1 and 2 overestimated iPPCPs concentrations, especially in pods of soybean, which were overestimated by more than seven orders of magnitude. Typically, agricultural soil pH ranges from 5 to 9 (strongly acidic to strongly alkaline), which can be important for the response of iPPCPs (USDA, 1998). In desert/semi-desert regions, where the reuse of reclaimed water is particularly important and attractive, soil pH tends to be high (up to ~9.6), which can facilitate plant uptake of iPPCPs (United States Department of Agriculture, Natural Resources Conservation Service, 2009). The results emphasize the importance of consideration of pH and speciation for steady-state concentrations of iPPCPs in soil and plant tissues. The pH inside the plant doesn't vary as much as the soil pH. The largest fraction of plant cell is vacuoles with pH range of 4–5.5 (Trapp, 2009). As a result, the effect of changes in the pH of plant tissues was not accounted in this study, when its potential influences were explored (SI, Text S4).

4.3. Other important factors

There is still limited data regarding the metabolism of iPPCPs inside agricultural plants. Few experimental studies with plant (carrot and radish) cell culture showed that IBU and TCS were metabolized rapidly with reaction rate constants of 3.97 ± 5.15 and 28.53 ± 47.34 d^{-1}, respectively (SI, Table S8), based on limited experiments (He et al., 2017; Li et al., 2018; Macherius et al., 2012; Wu et al., 2016). Accounting for such rapid metabolism in our model, the steady-state concentration of iPPCPs in plant and plant uptake factor decreased by several orders of magnitude. For example, the plant uptake factor for lettuce leaves at pH 7 under scenario 4 decreased from 1.52 to 8.96×10^{-3} for IBU and from 1.4×10^{-2} to 1.32×10^{-7} for TCS. Such low plant uptake factor of IBU and TCS deviates from the reported experimental data. For instance, Pannu et al. (2012) reported a TCS uptake factor of $1.6 \times 10^{-1} \pm 1.6 \times 10^{-1}$ in soybean grain and $7 \times 10^{-2} \pm 5 \times 10^{-2}$ in corn leaf, which is closer to our results obtained without consideration of plant metabolism. Wu et al. (2013) found an IBU uptake factor of 2.4×10^{-1} for pepper leaves, which was also much higher than the calculated values when accounting for plant metabolism. These deviations emphasize the need for future research in this area. The calculations can be more challenging for compounds with multiple ionizable functional groups (zwitterions). As an example, the plant uptake of tetracycline (TCL) was calculated under Scenario 1, 3 and 4, using the regression of measured sorption coefficient against the factions of species (Text S5, Fig. S19). The simulated plant uptake of TCL increased with soil pH, and plant uptake was significantly miscalculated, when sorption by minerals was neglected. To predict the fate of zwitterions in soil-plant systems requires further investigations.

5. Conclusions

Our modeling approach demonstrated that the pH and speciation of compounds may strongly impact the fate of iPPCPs in the soil-plant system, especially for plant uptake and accumulation of iPPCPs in food crops. Overall, the accumulation of iPPCPs in plant tissues increased with pH, with the most remarkable example of TCS in lettuce leaves increasing by more than one order of magnitude when pH increased from 5 to 9. As a result, appropriate representation of iPPCPs speciation and transport processes in fate modeling are critical. Comparison of calculations under different scenarios indicates that estimating iPPCPs fate based only on K_{ow} or its normalization (D_{ow}), can drastically misrepresent plant uptake and it is important to consider the

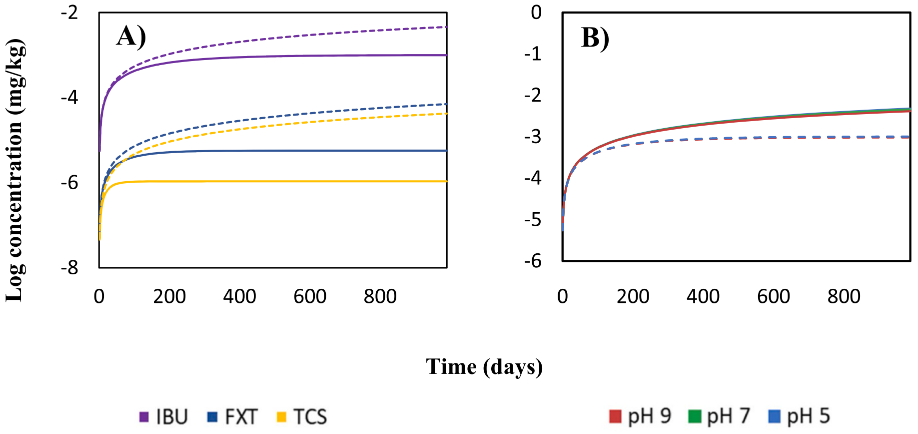

Fig. 4. Comparison between a) concentration of four iPPCPs in soil at pH 7 under scenario 4 based on bioavailability b) concentration of IBU at different pH under scenario 4. The dashed lines represent the concentration considering the bioavailability for degradation rate constant calculation while the solid lines represent concentration without considering it.

sorption of neutral and ionized species by both soil organic matter and minerals. Validation with reported plant uptake factors in literatures concurred the importance of soil pH and speciation of iPPCPs for their plant uptake. Compared to the soil-water partitioning, data is even more scarce for the plant tissue-internal solution partitioning. Our model showed a minimal response of soil concentration to the pH and speciation, because the steady-state concentrations were primarily determined by the balance of influx and efflux, dominated by the pH/speciation-independent degradation. However, if the bioavailability was taken into account, soil concentrations were highly dependent on the pH. More data is needed for the bioavailability and degradation of iPPCPs in soils as a function of pH, especially upon the modeling expanded to a larger range of iPPCPs and other ionizable chemicals occurring in reclaimed wastewater (including other water bodies such as hydrofracking produced water), e.g., wood preservatives, disinfectants, pesticides, herbicides, and others. Also, the model can be expanded to predict the plant uptake of these wide range of chemicals by further considering the processes like foliar uptake.

Author statement

Abrar Shahriar: Methodology; Validation; Data Formal analysis; Investigation; Writing**. David Hanigan:** Resources; Writing – review & editing. **Paul Verburg:** Resources; Writing – review & editing. **Krishna Pagilla:** Resources; Writing – review & editing. **Yu Yang:** Conceptualization; Methodology; Investigation; Resources; Data Formal analysis; Writing; Supervision; Project administration; Funding acquisition.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

Acknowledgement

This project was mainly supported by a USDA grant (No. 2017-69007-26309). It was also partially supported by NSF grants (No. 1808372 and No. 2108270).

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envpol.2022.120367.

References

Archer, E., Petrie, B., Kasprzyk-Hordern, B., Wolfaardt, G.M., 2017. The fate of pharmaceuticals and personal care products (PPCPs), endocrine disrupting contaminants (EDCs), metabolites and illicit drugs in a WWTW and environmental waters. Chemosphere 174, 437–446. https://doi.org/10.1016/j.chemosphere.2017.01.101.

Behera, S.K., Oh, S.Y., Park, H.S., 2012. Sorptive removal of ibuprofen from water using selected soil minerals and activated carbon. Int. J. Environ. Sci. Technol. 9 (1), 85–94.

Behera, S.K., Oh, S.-Y., Park, H.-S., 2010. Sorption of triclosan onto activated carbon, kaolinite and montmorillonite: effects of pH, ionic strength, and humic acid. J. Hazard Mater. 179 (1–3), 684–691.

Brunetti, G., Kodešová, R., Švecová, H., Fér, M., Nikodem, A., Klement, A., Grabic, R., Sim\uunek, J., 2022. A novel multiscale biophysical model to predict the fate of ionizable compounds in the soil-plant continuum. J. Hazard Mater. 423, 127008.

Carter, L.J., Harris, E., Williams, M., Ryan, J.J., Kookana, R.S., Boxall, A.B., 2014. Fate and uptake of pharmaceuticals in soil–plant systems. J. Agric. Food Chem. 62 (4), 816–825.

Chang, Q., Ji, W., Lu, Q., Xue, J., Hua, R., Wu, X., 2021. Bioavailability and toxicity of imazethapyr in maize plant estimated by four chemical extraction techniques in different soils. Sci. Total Environ. 801, 149594 https://doi.org/10.1016/j.scitotenv.2021.149594.

Chen, W., Xu, J., Lu, S., Jiao, W., Wu, L., Chang, A.C., 2013. Fates and transport of PPCPs in soil receiving reclaimed water irrigation. Chemosphere 93 (10), 2621–2630. https://doi.org/10.1016/j.chemosphere.2013.09.088.

Chen, Z., Zhang, Y., Gao, Y., Boyd, S.A., Zhu, D., Li, H., 2015. Influence of dissolved organic matter on tetracycline bioavailability to an antibiotic-resistant bacterium. Environ. Sci. Technol. 49 (18), 10903–10910. https://doi.org/10.1021/acs.est.5b02158.

Clarke, R., Healy, M.G., Fenton, O., Cummins, E., 2018. Quantitative risk assessment of antimicrobials in biosolids applied on agricultural land and potential translocation into food. Food Res. Int. 106, 1049–1060.

Cortés, J.M., Larsson, E., Jönsson, J.Å., 2013. Study of the uptake of non-steroid anti-inflammatory drugs in wheat and soybean after application of sewage sludge as a fertilizer. Sci. Total Environ. 449, 385–389. https://doi.org/10.1016/j.scitotenv.2013.01.061.

Delli Compagni, R., Gabrielli, M., Polesel, F., Turolla, A., Trapp, S., Vezzaro, L., Antonelli, M., 2020a. Modeling tools for risk management in reclaimed wastewater reuse systems: focus on contaminants of emerging concern (CECs). In: Verlicchi, P. (Ed.), Advances in Chemical Pollution, Environmental Management and Protection. Elsevier, pp. 181–220. https://doi.org/10.1016/bs.apmp.2020.07.010.

Delli Compagni, R., Gabrielli, M., Polesel, F., Turolla, A., Trapp, S., Vezzaro, L., Antonelli, M., 2020b. Risk assessment of contaminants of emerging concern in the context of wastewater reuse for irrigation: an integrated modelling approach. Chemosphere 242, 125185. https://doi.org/10.1016/j.chemosphere.2019.125185.

Droge, S.T., Goss, K.-U., 2013a. Development and evaluation of a new sorption model for organic cations in soil: contributions from organic matter and clay minerals. Environ. Sci. Technol. 47 (24), 14233–14241.

Droge, S.T.J., Goss, K.-U., 2013b. Sorption of organic cations to phyllosilicate clay minerals: CEC-normalization, salt dependency, and the role of electrostatic and hydrophobic effects. Environ. Sci. Technol. 47 (24), 14224–14232. https://doi.org/10.1021/es403187w.

Figueroa, R.A., Leonard, A., MacKay, A.A., 2004. Modeling tetracycline antibiotic sorption to clays. Environ. Sci. Technol. 38 (2), 476–483.

Franco, A., Fu, W., Trapp, S., 2009. Influence of soil pH on the sorption of ionizable chemicals: modeling advances. Environ. Toxicol. Chem.: Int. J. 28 (3), 458–464.

Fu, Q., Malchi, T., Carter, L.J., Li, H., Gan, J., Chefetz, B., 2019. Pharmaceutical and Personal Care Products: from Wastewater Treatment into Agro-Food Systems. Environmental Science & Technology. https://doi.org/10.1021/acs.est.9b06206.

Fu, Q., Wu, X., Ye, Q., Ernst, F., Gan, J., 2016. Biosolids inhibit bioavailability and plant uptake of triclosan and triclocarban. Water Res. 102, 117–124. https://doi.org/10.1016/j.watres.2016.06.026.

García, M.G., Fernández-López, C., Polesel, F., Trapp, S., 2019. Predicting the uptake of emerging organic contaminants in vegetables irrigated with treated wastewater–implications for food safety assessment. Environ. Res. 172, 175–181.

García-Santiago, X., Garrido, J.M., Lema, J.M., Franco-Uría, A., 2017. Fate of pharmaceuticals in soil after application of STPs products: influence of physicochemical properties and modelling approach. Chemosphere 182, 406–415. https://doi.org/10.1016/j.chemosphere.2017.05.049.

Geisler, A., Endo, S., Goss, K.-U., 2012. Partitioning of organic chemicals to storage lipids: elucidating the dependence on fatty acid composition and temperature. Environ. Sci. Technol. 46 (17), 9519–9524. https://doi.org/10.1021/es301921w.

Government of Canada, E. and C. C. C., 2013. Evaluating Existing Substances—Draft Screening Assessment for the Challenge for Chlorhexidine Acetate. Environment Canada. https://www.ec.gc.ca/ese-ees/default.asp?lang=En&n=8CCF6AE7-1.

He, Y., Langenhoff, A.A., Sutton, N.B., Rijnaarts, H.H., Blokland, M.H., Chen, F., Huber, C., Schröder, P., 2017. Metabolism of ibuprofen by Phragmites australis: uptake and phytodegradation. Environ. Sci. Technol. 51 (8), 4576–4584.

Hiller, E., Šebesta, M., 2017. Effect of temperature and soil pH on the sorption of ibuprofen in agricultural soil. Soil Water Res. 12 (2), 78–85.

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

Hurtado, C., Trapp, S., Bayona, J.M., 2016. Inverse modeling of the biodegradation of emerging organic contaminants in the soil-plant system. Chemosphere 156, 236–244. https://doi.org/10.1016/j.chemosphere.2016.04.134.

Hyland, K.C., Blaine, A.C., Higgins, C.P., 2015. Accumulation of contaminants of emerging concern in food crops—part 2: plant distribution. Environ. Toxicol. Chem. 34 (10), 2222–2230. https://doi.org/10.1002/etc.3068.

Kovacs, E.D., Silaghi-Dumitrescu, L., Kovacs, M.H., Roman, C., 2021. Determination of the uptake of ibuprofen, ketoprofen, and diclofenac by tomatoes, radishes, and lettuce by gas chromatography–mass spectrometry (GC–MS). Anal. Lett. 54 (1–2), 314–330.

Legind, C.N., Kennedy, C.M., Rein, A., Snyder, N., Trapp, S., 2011. Dynamic plant uptake model applied for drip irrigation of an insecticide to pepper fruit plants. Pest Manag. Sci. 67 (5), 521–527. https://doi.org/10.1002/ps.2087.

Li, J., Carter, L.J., Boxall, A.B.A., 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.

Li, Y., Chuang, Y.-H., Sallach, J.B., Zhang, W., Boyd, S.A., Li, H., 2018. Potential metabolism of pharmaceuticals in radish: comparison of in vivo and in vitro exposure. Environ. Pollut. 242, 962–969.

Li, Z., Xiang, X., Li, M., Ma, Y., Wang, J., Liu, X., 2015. Occurrence and risk assessment of pharmaceuticals and personal care products and endocrine disrupting chemicals in reclaimed water and receiving groundwater in China. Ecotoxicol. Environ. Saf. 119, 74–80. https://doi.org/10.1016/j.ecoenv.2015.04.031.

Macherius, A., Eggen, T., Lorenz, W., Moeder, M., Ondruschka, J., Reemtsma, T., 2012. Metabolization of the bacteriostatic agent triclosan in edible plants and its consequences for plant uptake assessment. Environ. Sci. Technol. 46 (19), 10797–10804. https://doi.org/10.1021/es3028378.

MacKay, A.A., Vasudevan, D., 2012. Polyfunctional ionogenic compound sorption: challenges and new approaches to advance predictive models. Environ. Sci. Technol. 46 (17), 9209–9223. https://doi.org/10.1021/es301036t.

Mackay, D., 1979. Finding fugacity feasible. Environ. Sci. Technol. 13 (10), 1218–1223.

Malchi, T., Maor, Y., Tadmor, G., Shenker, M., Chefetz, B., 2014. Irrigation of root vegetables with treated wastewater: evaluating uptake of pharmaceuticals and the associated human health risks. Environ. Sci. Technol. 48 (16), 9325–9333. https://doi.org/10.1021/es5017894.

Miller, E.L., Nason, S.L., Karthikeyan, K.G., Pedersen, J.A., 2016. Root uptake of pharmaceuticals and personal care product ingredients. Environ. Sci. Technol. 50 (2), 525–541. https://doi.org/10.1021/acs.est.5b01546.

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.

Polesel, F., Plósz, B.G., Trapp, S., 2015. From consumption to harvest: environmental fate prediction of excreted ionizable trace organic chemicals. Water Res. 84, 85–98.

Poustie, A., Yang, Y., Verburg, P., Pagilla, K., Hanigan, D., 2020. Reclaimed Wastewater as a Viable Water Source for Agricultural Irrigation: A Review of Food Crop Growth Inhibition and Promotion in the Context of Environmental Change. Science of The Total Environment. https://doi.org/10.1016/j.scitotenv.2020.139756.

Prosser, R.S., Lissemore, L., Topp, E., Sibley, P.K., 2014a. 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.

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.

Prosser, R.S., Trapp, S., Sibley, P.K., 2014b. 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.

Qin, Q., Chen, X., Zhuang, J., 2015. The fate and impact of pharmaceuticals and personal care products in agricultural soils irrigated with reclaimed water. Crit. Rev. Environ. Sci. Technol. 45 (13), 1379–1408. https://doi.org/10.1080/10643389.2014.955628.

R Core Team, 2021. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing. https://www.R-project.org/.

Reichenberg, F., Mayer, P., 2006. Two complementary sides of bioavailability: accessibility and chemical activity of organic contaminants in sediments and soils. https://doi.org/10.1016/j.envpol.2021.116532 Environ. Toxicol. Chem. 25 (5), 1239–1245. https://doi.org/10.1897/05-458R.1.

Shahriar, A., Tan, J., Sharma, P., Hanigan, D., Verburg, P., Pagilla, K., Yang, Y., 2021. Modeling the fate and human health impacts of pharmaceuticals and personal care products in reclaimed wastewater irrigation for agriculture. Environ. Pollut. 276, 116532.

Shariq, L., McLaughlin, M.C., Rehberg, R.A., Miller, H., Blotevogel, J., Borch, T., 2021. Irrigation of wheat with select hydraulic fracturing chemicals: evaluating plant uptake and growth impacts. Environ. Pollut. 273, 116402 https://doi.org/10.1016/j.envpol.2020.116402.

Sharma, P., Poustie, A., Verburg, P., Pagilla, K., Yang, Y., Hanigan, D., 2020. Trace organic contaminants in field-scale cultivated alfalfa, soil, and pore water after 10 years of irrigation with reclaimed wastewater. Sci. Total Environ. 744, 140698 https://doi.org/10.1016/j.scitotenv.2020.140698.

Tanoue, R., Sato, Y., Motoyama, M., Nakagawa, S., Shinohara, R., Nomiyama, K., 2012. Plant uptake of pharmaceutical chemicals detected in recycled organic manure and reclaimed wastewater. J. Agric. Food Chem. 60 (41), 10203–10211. https://doi.org/10.1021/jf303142t.

Trapp, S., 2007. Fruit Tree model for uptake of organic compounds from soil and air. SAR QSAR Environ. Res. 18 (3–4), 367–387.

Trapp, S., 2009. Bioaccumulation of polar and ionizable compounds in plants. In: Ecotoxicology Modeling. Springer, pp. 299–353.

United States Department of Agriculture, Natural Resources Conservation Service, 2009. Soil of Great Basin National Park, Nevada. United States Department of Agriculture. https://www.nrcs.usda.gov/wps/portal/nrcs/soilsurvey/soils/survey/state/.

US EPA, 2012. Estimation Programs Interface SuiteTM for Microsoft® Windows (4.11. United States Environmental Protection Agency.

USDA, 1998. Soil Quality Indicators: PH. https://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/nrcs142p2_052208.pdf.

Verlicchi, P., Al Aukidy, M., Zambello, E., 2012. Occurrence of pharmaceutical compounds in urban wastewater: removal, mass load and environmental risk after a secondary treatment—a review. Sci. Total Environ. 429, 123–155. https://doi.org/10.1016/j.scitotenv.2012.04.028.

Wanat, K., Żydek, G., Hekner, A., Brzezińska, E., 2021. In silico plasma protein binding studies of selected group of drugs using TLC and HPLC retention data. Pharmaceuticals 14 (3), 202.

Wu, C., Spongberg, A.L., Witter, J.D., 2009. Adsorption and degradation of triclosan and triclocarban in soils and biosolids-amended soils. J. Agric. Food Chem. 57 (11), 4900–4905.

Wu, C., Spongberg, A.L., Witter, J.D., Fang, M., Czajkowski, K.P., 2010. Uptake of pharmaceutical and personal care products by soybean plants from soils applied with biosolids and irrigated with contaminated water. Environ. Sci. Technol. 44 (16), 6157–6161. https://doi.org/10.1021/es1011115.

Wu, X., Ernst, F., Conkle, J.L., Gan, J., 2013. Comparative uptake and translocation of pharmaceutical and personal care products (PPCPs) by common vegetables. Environ. Int. 60, 15–22. https://doi.org/10.1016/j.envint.2013.07.015.

Wu, X., Fu, Q., Gan, J., 2016. Metabolism of pharmaceutical and personal care products by carrot cell cultures. Environ. Pollut. 211, 141–147. https://doi.org/10.1016/j.envpol.2015.12.050.

Yang, X., Flowers, R.C., Weinberg, H.S., Singer, P.C., 2011. Occurrence and removal of pharmaceuticals and personal care products (PPCPs) in an advanced wastewater reclamation plant. Water Res. 45 (16), 5218–5228. https://doi.org/10.1016/j.watres.2011.07.026.