Remediation potential of agricultural organic micropollutants in in-situ techniques
A review

Bai, Ying; Wang, Zongzhi ; van der Hoek, Jan Peter

DOI
10.1016/j.ecoinf.2021.101517

Publication date
2022

Document Version
Final published version

Published in
Ecological Informatics: an international journal on ecoinformatics and computational ecology

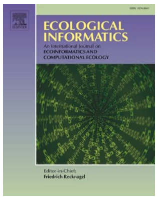
Citation (APA)
Bai, Y., Wang, Z., & van der Hoek, J. P. (2022). Remediation potential of agricultural organic micropollutants in in-situ techniques: A review. *Ecological Informatics: an international journal on ecoinformatics and computational ecology*, *68*, 1-11. Article 101517. https://doi.org/10.1016/j.ecoinf.2021.101517

Ecological Informatics 68 (2022) 101517

Contents lists available at ScienceDirect

Ecological Informatics

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



Remediation potential of agricultural organic micropollutants in *in-situ* techniques: A review

Ying Bai [a], Zongzhi Wang [a,*], Jan Peter van der Hoek [b,c]

[a] State Key Laboratory of Hydrology-Water Resources and Hydraulic Engineering, Nanjing Hydraulic Research Institute, Nanjing 210029, China
[b] Department of Water Management, Delft University of Technology, Stevinweg1, 2628CN Delft, the Netherlands
[c] Strategic Centre, Waternet, Korte Ouderkerkerdijk 7, 1096AC Amsterdam, the Netherlands

ARTICLE INFO

Keywords:
Agricultural organic micropollutants
Constructed wetland
In-situ remediation
Permeable reactive barriers
Riverbank filtration
Riparian buffer zone

ABSTRACT

Agricultural activities can be important source of organic pollution. In agricultural intensive areas, organic chemicals have a high possibility of entering the water cycle, which could pose a potential risk to human health. Low-cost and high efficiency *in-situ* techniques instead of energy and money consuming ones to control agricultural organic micropollutants (Agro-OMPs) in aquatic system are extremely needed. In this paper, emerging Agro-OMPs were discussed focusing on their occurrence, pathways and risks. The mechanisms, dominant parameters and effectiveness of riverbank filtration, riparian buffer zone, constructed wetland and permeable reactive barriers for removing these pollutants are presented and discussed. Ecological succession in Riverbank Filtration (RBF) system is worth noticing for its stability maintenance. Riparian buffer zone (RBZ) should be explored more focusing on flexibility improvement and construction standardization. Constructed wetland (CW) is quite efficient on antibiotic resistance genes (ARGs) attenuation, but the risk of ARGs propagation still exist. Besides, more innovations should be made on combination, field-scale application and long-term evaluation of *in-situ* remediation techniques, which will provide references for agricultural water management and water quality improvement.

1. Introduction

Agricultural activities were important sources of organic pollutants such as plant protection products (PPPs) and veterinary antibiotics (Tuncel et al., 2008). In agricultural intensive areas, organic chemicals applied in agriculture may be mobilized and enter the water cycle by surface runoff, leaching, groundwater recharge *etc.* With the improvement of detection methods, increasing organic micropollutants (OMPs) have been reported in aquatic environments (Calderon-Preciado et al., 2011; Grung et al., 2015; Heberer, 2002), which may also effect biogeochemical processes, such as nitrogen fixation, denitrification, amination and biodegradation (Aldén Demoling and Bååth, 2008; Hjorth et al., 2006; Pesce et al., 2010; Rotter et al., 2011; Schmitt-Jansen et al., 2008; Zabaloy et al., 2010). Agro-OMPs is defined as a diverse group of OMPs introduced to the environment by agricultural activities, which pose a potential risk on human health.

PPPs (including herbicides, insecticides, nematicides, fungicides, and soil fumigants (Imfeld and Vuilleumier, 2012)), have been applied throughout the world under the pressure of food production demand since 1960s. It is reported that the risk of reaching groundwater will increase considerably when PPPs' water solubility >30 mg L^{-1}, absorptivity (K_{oc}) $<$ 300–500, or half-life in soil $>$2–3 weeks. Among the PPPs, pesticides had been classified into four group based on the toxicity (Table S1). PPPs and metabolites residue have been observed worldwide in groundwater systems since the early 1980s (Table S2), which might pose cancer risk, neurotoxicity and reproductive abnormalities on human being.

Veterinary antibiotics enter the environment (Sarmah et al., 2006)

Abbreviations: RBF, Riverbank filtration; RBZ, Riparian buffer zone; CW, Constructed wetland; PRB, Permeable reactive barrier; PRBB, Permeable reactive biobarrier; EKSF, Electrokinetic soil flushing; GAC, Granular active carbon; OMPs, Organic micropollutants; Agro-OMPs, Agricultural organic micropollutants; MCPA, 2-methyl-4-chlorophenoxyacetic acid; 2,4,6-TCP, 2,4,6-trichlorophenol; 2.4-DCP, 2,4-dichlorophenol; HCB, Hexachlorobenzene; PCP, Pentachlorophenol; PAHs, Polycyclic aromatic hydrocarbons; VSSF, Vertical subsurface flow; VSDF, Vertical subsurface down flow; VSUF, Vertical subsurface up flow; HSF, Horizontal subsurface flow; MFC, Microbial fuel cell; BER, Biofilm electrode reactor.

* Corresponding author at: 225# Guangzhou Road, 210029 Nanjing, China.
E-mail address: wangzz77@163.com (Z. Wang).

https://doi.org/10.1016/j.ecoinf.2021.101517
Received 19 April 2021; Received in revised form 22 August 2021; Accepted 2 December 2021

through fecal releasing, runoff and leaching after the application of animal manure to farmland and pasture. Veterinary antibiotics have been detected in groundwater in concentration from ng L^{-1} to μg L^{-1} (Table S3). Antibiotics and metabolites may alter the microbial community composition and influence the elemental cycling in critical zone (Joergensen and Emmerling, 2006). Antibiotic resistance genes (ARGs) are introduced into water system by increased application of manure and sewage sludge on farmland (Castiglioni et al., 2008; Hu et al., 2008; Koike et al., 2007; Pei et al., 2006; Zhu et al., 2013). ARGs can be spread to human pathogens through mobile genetic elements (MGEs) by horizontal gene transfer (HGT) mechanism (Huddleston, 2014) and pose risks on human health (Heuer et al., 2011; Zhang et al., 2009).

The main characteristics of agricultural diffuse pollution are continuous occurrence, massive amount and extended distribution. Thus, a less-invasive, sustainable and cheap solution are ideal. Techniques complementary to the natural environment, which took advantage of nature's capacity to degrade the pollutants were thus continuously developed and implemented. This article comprehensively reviewed four potential *in-situ* remediation techniques for agro-OMPs remediation (riverbank filtration, riparian buffer zone, constructed wetland and permeable reactive barrier) on different transportation pathways focusing on mechanisms, effectiveness and influence factors, in order to provide reference and alternatives for agricultural non-point source pollution control, as well as water safety protection.

2. *In-situ* remediation technique for agro-OMPs

2.1. River bank filtration (RBF)

2.1.1. Technology description

RBF is a pretreatment process of drinking water production (van der Hoek et al., 2014). Pumping wells create a hydraulic gradient and continuous groundwater flow in RBF system. European cities along the major rivers (Rhine, Elbe, Danube, *etc.*) have utilized RBF for over a century (Sharma, 2012). Normally, RBF can effectively remove nitrate, natural organic matter, particles (turbidity), pathogenic organisms (viruses, bacteria and protozoa), as well as OMPs (Bertelkamp et al., 2014; Bertelkamp et al., 2016b; Heberer, 2002; Heberer et al., 2004; Heberer et al., 2008; Maeng et al., 2011; Ray, 2004). Residence time, redox condition and infiltrated water ratio were reported with high importance for OMPs attenuation in one RBF site (Kovacevic et al., 2017). Residence time is an important hydrogeological parameter by influencing interactions between water and aquifer matrix. Redox condition is another main parameter because of its significant influences on the mobility, biodegradability and toxicity of pollutants in solid-aquatic environment (Table 1).

2.1.2. Agro-OMPs removal efficiency

RBFs have been reported as effective, low cost and sustainable for OMPs removal compared with rapid sand filtration and activated carbon adsorption (Benner et al., 2013; Bertelkamp et al., 2015; Bertelkamp et al., 2016a; Regnery et al., 2015). At present, studies on RBF mainly focus on the occurrence and attenuation of OMPs in effluent of wastewater treatment plants. However, degradation of OMPs with different chemical structure and hydrophobicity can be quite distinct (Bertelkamp et al., 2014). Removal of OMPs during RBF is attributed to microbial degradation and adsorption. Thus compounds negatively charged or with low log K_{OW} values are more preferential in degradation. While compounds neutrally charged or with high log K_{OW} values are more preferential in adsorption (Benotti et al., 2012). It was also reported that the attenuation of OMPs during RBF happened both in hyporheic attenuation area and aquifer reduction area, which depended on not only the physical–chemical properties of the compounds and dissolved organic matter in the water, but also source load, hydrogeological conditions, and environmental parameters (Yang et al., 2017).

The removal rate of pesticides in RBF varies from sites and compounds. In RBF site at Lake Wannsee, Berlin, p,p'-DDA and o,p'-DDA were removed from 20 and 5 ng•L^{-1} to 10 and < 5 ng•L^{-1} respectively. Herbicides bentazone and mecoprop were eliminated from 25 and 15 ng•L^{-1} to 10 ng•L^{-1} (Heberer et al., 2004). Atrazine was persistent with a removal of only 20% in a pilot RBF system (Benotti et al., 2012). Carbofuran was efficiently removed by approximately 60% (339 ± 87 ng•L^{-1} to 130 ± 50 ng•L^{-1}) in RBF site of the Itajai do Sul River (Romero et al., 2010).

The behavior of veterinary antibiotics in RBF has been profoundly studied (Petrovic et al., 2009). Sulfamethoxazole was detected with median concentration of 151 ng•L^{-1} at RBF site of Lake Wannsee, Berlin and was removed by >98% under anoxic condition (Heberer et al., 2008). While sulfamethoxazole with ng•L^{-1} concentration was reported removed by <20% in RBF pilot setup (Benotti et al., 2012). Sulfamethoxazole and trimethoprim with concentration of 4–223 ng•L^{-1} was totally removed during RBF site in Serbia (Kovacevic et al., 2017). Chloramphenicol with a concentration of 8.9 ± 4.1 ng•L^{-1} was reduced to 1.6 ± 1.6 ng•L^{-1} by RBF in wet season and from 158.5 ± 58.3 ng•L^{-1} to 55.5 ± 27.5 ng•L^{-1} in dry season in Beiyun River, China, at the same time trimethoprim with a concentration of 71.3–302 ng•L^{-1} was almost totally removed (Yang et al., 2017). Other antibiotics like clarythromycin, roxythromycin and anhydroerythromycin could also be readily removed by RBF (Petrovic et al., 2009).

RBF is also effective for pathogenic microorganism removal. It has been noted that concentration reduction of at least 4-log for *Giardia-lamblia* and *Crytosporidiumparvum* (Gollnitz et al., 2003), and an average of 3-log are accomplished in RBF system (Ray et al., 2003). A removal of 1.3 to >5.2 log for total coliforms and 2.3 to >4.2 log for fecal coliforms were reported in RBF site in North India (Sandhu and Grischek, 2012). The average *E. coli* reduction were 2.15–2.97 log in RBF site of Santa Catarina, Brazil (Romero et al., 2010). Report about ARGs attenuation in RBF was not found, which still need more attention for risk control of antibiotic resistance dissemination.

Although RBF is an effective technique for the removal of OMPs and agro-OMPs, the reaction mechanisms have not been completely revealed. Both biodegradation and adsorption play an important role in OMPs removal in the RBFs, but their quantified contribution and potential refer to different OMPs groups are still topics of research

Table 1
Redox conditions and the indicators in RBF (Bertelkamp, 2015).

Redox clusters	Redox process	DO (mg L^{-1})	Indicator (mg L^{-1})	e^-acceptor	H_2 indicator (nM)	ΔG_0(W) (kcal mol^{-1})
oxic	aerobic respiration	>1	$Mn_2^+/Fe_2^+ < 0.1$, $SO_4^{2-} > 0.9$, $CH_4 < 0.1$	O_2	<0.1	−120
(sub)anoxic	denitrification	<1	$NO_3^- > 0.5$	NO_3^-	<0.1	−114
anoxic	Fe(III) reduction	<0.5	$NO_3^- < 0.5$, Mn_2^+/Fe_2^+ (increasing)	Mn_4^+/Fe_3^+	0.1–0.2/0.2–0.8	−28
deep anoxic	sulfate reduction	<0.5	$NO_3^- < 0.5$, $SO_4^{2-} < 5$ (decreasing), $HS^- > 0$	SO_4^{2-}	1–4	−25
	Mn(IV) reduction	<0.5	$CH_4 > 0.2$ (increasing)	CO_2	5	−81

(Bertelkamp et al., 2014). Microorganisms have been observed to employ two main catalytic processes when participating in biological-mediated reactions with OMPs: metabolic reactions and co-metabolic reactions (Benner et al., 2013). It is necessary to explore the mechanism of both within RBF, as well as to reveal the changes of microbial community structure and function during this biological process (Bai et al., 2019). Despite of all the advantages, RBF is still not sufficient for agro-OMPs remediation and requires additional purification process (Petrovic et al., 2009). It was proposed that several agro-OMPs, such as atrazine and selected antibiotics, may be used as tracers to evaluate pollution risk and the necessity of further multi-barrier treatment process after RBF (Verstraeten et al., 2003). RBF efficiency on agro-OMPs removal were shown in Table 2.

2.2. Riparian buffer zone (RBZ)

2.2.1. Technology description

Buffer zone is a linear band of vegetation adjacent to aquatic ecosystem trapping and removing various diffuse source pollutants from both overland and shallow subsurface flow. Width, length, vegetation structure, and placement are the most crucial parameters for RBZ. It was reported that pesticides (atrazine, chlorpyrifos, fluazifop-p-butyl, lambda-cyhalothrin and lactofen) removal ratio in RBZ followed the order of wood > shrubs > grass, and larger width will increase the efficiency (Aguiar et al., 2015). Besides, herbicides loaded to RBZ was related to atmospheric deposition and subsequent wash-off during rain events, which will also impact the overall removal (Rice et al., 2016).

2.2.2. Agro-OMPs removal efficiency

The most important mechanisms for pollutant removal and attenuation in RBZ are sedimentation, adsorption, oxidation, reduction, biodegradation and plant uptake (Vymazal and Březinová, 2015), among which biodegradation contribute greatly to agro-OMPs removal. From the current knowledge, vegetated buffer systems have the potential to reduce pesticides leaching and retain up to 90% in runoff. However, based on limited monitoring data, there is uncertainty to quantify the biodegradation effectiveness of RBZ in most cases (Entry et al., 1994; Vandermeeren et al., 2016).

Width of buffer zone was the most frequently reported parameter that influence the removal efficiency. A model $Y=K\times(1\text{-}e^{-bw})$, $(0 < K \le 100)$ was developed to describe the relationship between buffer width and pollutant removal, where K represents the maximum removal efficacy and b represents its probability to remove any single particle of pollutant in a unit distance (Zhang et al., 2010). RBZ in the Atlantic forest region can protect stream ecosystems from pesticides (Hunt et al., 2017). It was reported that a woody RBZ with 36 m width can remove 70–94% of the pesticides (Aguiar et al., 2015). 77% reduction of dissolved atrazine in surface runoff during non-tropical storm event was reported based on Riparian Ecosystem Management Model (Williams et al., 2016). Since sediments are easily removed by RBZ, hydrophobic pesticides with high K_{oc} ould be more effectively removed. The average retention (with ranges) of weakly (K_{oc} ⟨100), moderately (100 < K_{oc} ⟨1000), and strongly (K_{oc} > 1000) absorbed pesticides were 61% (0–100%), 63% (0–100%), and 76% (53–100%), respectively (Kapil Arora et al., 2010).

Hazel in RBZ can effectively reduce sulfadiazine in soil and rhizosphere, which indicates the potential removal of antibiotics by vegetated RBZ (Michelini et al., 2015). While, limited data about ARGs attenuation by RBZ was found in the reviewed articles.

Overall, RBZ is reported as flexible, economic and effective for agro-OMPs control, which can be applied by comprehensive arrangement at catchment scale (Reichenberger et al., 2007). More researches concerning substances exchange between different mediums in RBZ should be conducted at field scale, and its long-term efficiency should be comprehensively evaluated. Besides, because of the complexity of identifying the sufficient width for public health protection, more exploration on RBZ construction standard in agricultural area were needed (Birnbaum et al., 2017). Detailed efficiency of RBZ on agro-OMPs removal was summarized in Table 3.

2.3. Constructed wetland (CW)

2.3.1. Technology description

CW is firstly introduced to pesticides removal in the 1970s, and later is widely recommended as effective technique for reducing agriculture diffuse source pollution (Dordio and Carvalho, 2013; Elsayed et al.,

Table 2
Reported agro-OMPs removal in RBF systems.

Location	Compounds	Concentration ($ng \bullet L^{-1}$)	Removal efficiency	Parameters	References
Lake Wannsee and Tegel, Berlin	p,p'-DDA	20	50%	Shallow aquifer (average depth of 10-15 m)	Heberer et al. (2004)
	o,p'-DDA	5	50%		
	bentazone	25	60%		
	mecoprop	15	33%		
	sulfamethoxazole	151(100–326)	99% (anoxic), 52% (seasonally oxic)	preference for anoxic	Heberer et al. (2008)
	acetyl-sulfamethoxazole	7(4–14)	100%	slight preference for anoxic	
	clindamycin	7(15–48)	>98%(oxic)	preference for oxic	
	trimethoprim	12(7–49)	100%		
	roxithromycin	11(4.3–69)	100%(oxic)	preference for oxic	
	clarithromycin	8.9(1.8–43)	100%(oxic)	preference for oxic	
	clindamycin	31(15–48)	26% (anoxic), 93% (seasonally oxic)	preference for oxic	
	anhydroerythromycin	50(2.2–94)	98% (anoxic), 91% (seasonally oxic)	slight preference for anoxic	
Itajai do Sul River and Aquaculture lake, Brazil	carbofuran	339 ± 87	61%	hydraulic conductivities of 0.03(bottom)-46 (sub soil) m/d, travel time of 28–70 days, well depth of 2.8–4.7 m	Romero et al. (2010)
pilot	atrazine	–	<20%	36 days	Benotti et al. (2012)
	sulfamethoxazole	10			
Serbia	sulfamethoxazole	11–23	100%	100 days residence time and anoxic conditions	Kovacevic et al. (2017)
	azithromycin	20–56	n.c.		
	trimethoprim	4–223	100%		
Beiyun River, China	chloramphenicol	8.9 ± 4.1	82%	20 m depth	Yang et al. (2017)
	trimethoprim	71.3–302	100%		

Table 3
Reported agro-OMPs removal in RBZ systems.

Location	Compounds	Concentration (μg• L^{-1})	Removal efficiency	Parameters	Reference
Cará-Cará River Basin, Ponta Grossa, PR	atrazine	15.9	40%	woody, w = 12 m	Aguiar et al. (2015)
		7.0	70%	woody, w = 36 m	
		0.3	94%	woody, w = 60 m	
		10.4	50%	shrub, w = 60 m	
		13.3	40%	grass, w = 60 m	
Pirapó River, Paraguay	chlorpyrifos	0.17	n.c.	w = 89.3(3–500)m	Hunt et al. (2017)
	cypermethrin	0.02	n.c.	DO = 8.4(2.9–10) mg• L^{-1}	
	L-cyhalothrin	0.05	n.c.	TOC = 0.78(0.22–2.12)%	
San Francisco River, Brazil	chlorpyrifos	0.04	n.c.	w = 56.6(9–350)m	
	cCypermethrin	0.04	n.c.	DO = 8.8(7.3–14.6) mg• L^{-1}	
	L-cyhalothrin	0.02	n.c.	TOC = 2.32(1.37–3.24)%	
pilot	atrazine, metolachlor, glyphosate	–	58–72%	silt loam soil, w = 16 m, slope = 5%, (tall fescue, switchgrass and warm-season grasses)	Lin et al. (2011)
	sulfamethazine		>70%		
Coastal Zone, Southeastern Puerto Rico	atrazine	40	77%	w = 77 m, mixed forest	Williams et al. (2016)
			70%	w = 40 m, mixed forest	
			56%	w = 40 m, grasses	
modeling	total OMPs	–	>85%	w = 30 m, slope = 10%	Zhang et al. (2010)

W: width; DO: dissolved oxygen; TOC: total organic carbon.

2014; Robert Budd et al., 2009) and veterinary antibiotics (Berglund et al., 2014; Liu et al., 2014)).

The hydraulic conditions of CW include vertical flow, horizontal subsurface flow, free water surface flow and hybrid systems that incorporate surface and subsurface flow (Gorito et al., 2017; Shutes, 2001; USDA-Natural Resources Conservation Service, U, n.d.). Under different condition, various hydrological period, residence time and feeding water composition (Carluer et al., 2011; Lizotte Jr. et al., 2012; Romain et al., 2015) impact the capacity of CW. An intermittent flow in wetland under storm can enhance the mixing of anaerobic zone in sediment with the adjacent aerobic and anoxic zone in rhizosphere, which lead to temporal variation of hydro-chemical condition (Gregoire et al., 2009; Maillard et al., 2011; Passeport et al., 2011a, 2011b; Passeport et al., 2014; Tournebize et al., 2013a). One research revealed that 50–80% of pesticides were removed in CW at 10 times residence time forced by water recirculation and flow deceleration (Gregoire et al., 2009).

The configuration of CW also influent the performance. For example, an on-stream free water surface CW can achieve a mean pesticides reduction of 54% (32–100%), while the removal reduced to 45% for off-stream configuration (Tournebize et al., 2013b). Saturated vertical subsurface flow CW was reported efficient for trimethoprim and sulfamethoxazole removal (99% and 64% respectively) (Sgroi et al., 2018). In summer, the relatively low redox condition (nitrate reduction) in subsurface flow CW were favorable for anaerobic degradation of sulfamethoxazole (removal of 63%) (Rühmland et al., 2015).

2.3.2. *Agro-OMPs removal efficiency*

Plant metabolism and biodegradation in substrate matrix are the main removal mechanisms of agro-OMPs in CW (Lv et al., 2016). Sorption and biodegradation in rhizosphere account for 36.5–72.8% and 20.5–54.2% of the total macrolide antibiotics removal (Tai et al., 2017). Up to 88% of the suspended solid could be retained by CW, which increased the sedimentation capability for adsorbed pesticides (Maillard et al., 2011). It is notable that interactions between pesticides and substrates have a retarding effect on molecule transfer (Passeport et al., 2014), such as the remobilization of pesticides after heavy rainfall (Vallee et al., 2015). The effectiveness of CW is limited for pesticides or polar organic pollutants with low K_{oc} (Passeport et al., 2011b). Thus, CW can be either sink or source of agro-OMPs. The removal efficiency of CW on agro-OMPs was shown in Table 4.

Under EU water framework, CW is promoted as effective ecological bioengineering method for low pesticide-input agriculture and sustainable management. Based on reviewed literatures, average removal efficiency of pesticides in CW ranged from 20% to >90% (Tournebize et al., 2017), among which the highest ones are for organochlorine pesticides (97%), strobilurin/strobin pesticides (96%), organophosphate pesticides (94%) and pyrethroids (84%), while the lowest ones are for triazinone pesticides (24%), aryloxyalkanoic acid pesticides (35%) and urea-based pesticide (50%) (Vymazal and Březinová, 2015). Pesticides retention efficiency of field scale in-stream and off-stream CW is 54% and 45% respectively (Tournebize et al., 2013b). CW with light expanded clay aggregates as bed structure has removal efficiencies of 56%–97% for 2-methyl-4-chlorophenoxyacetic acid (MCPA) (Dordio and Carvalho, 2013).

Some frequently detected veterinary antibiotics are reported removable by CW. The removal efficiency of anhydroerythromycin A, roxithromycin, clarithromycin and ilmicosin in CW were 81.3–86.9%, 60.6–74.0%, 68.5–74.6% and 62.5–62.7% at 100 μg•L^{-1}, and 59.7–68.6%, 52.4–65.1%, 68.5–82.1%, and 65.1–91.8% at 300 μg•L^{-1} respectively (Tai et al., 2017). The removal efficiency of antibiotics in a CW microcosm ranged from 75.8% to 98.6% (Chen et al., 2016). An plant-clay aggregate CW achieved >97% removal for oxytetracycline with initial concentration of 1 μg•ml^{-1} in 3 days (Dordio and Carvalho, 2013).

CW is reported as effective barrier for ARGs in some condition. For example, sulfamethazine is excellently removed (>99.9%) in a full-scale hybrid CW, accompanying reduction of ***sul1, sul2***, and ***qnrA*** resistance genes (Yi et al., 2017). A horizontal subsurface flow CW reduces ***sul1*** resistance gene effectively (Nõlvak et al., 2013). CW contributed 42%–55% to almost complete removal of sulfamethoxazole (2 or 4 mg•L^{-1}) in a continuous flow microbial fuel cell CW coupled with biofilm electrode reactor, and ***sul*** gene abundance reduced because of bioelectricity in the CW (Li et al., 2018). A vertical subsurface flow CW and free water surface flow CW also showed a positive removal of ***sul1, sul2*** and ***ermB*** resistance genes (He et al., 2018). The removal of total ARGs is 63.9–84.0% by CW mesocosm (Chen et al., 2016), and the removal of ***tet*** genes and ***IntI*** range from 33.2% to 99.1% (Huang et al., 2017b).

In other circumstances, the removal of specific antibiotics in CW do not necessarily bring the reduction but a possible reservoir for ARGs. For example, in a biofilm electrode reactor and microbial fuel cell coupled CW with excellent sulfamethoxazole removal (>99.29%), the relative abundance of ***sul*** genes was enhanced (Zhang et al., 2018a). Similarly, in an effective vertical up-flow CW, the abundance of ***sul*** and ***tet*** resistance

Table 4
Reported agro-OMPs removal in CW systems.

Location	Compounds/ Genes	Concentration (μg•L^{-1})	Removal efficiency	Parameters	Reference
pilot(light expanded clay aggregates)	MCPA	1000	99.1 ± 3.6%	flow rate = 1.0 mL•min^{-1}, contact time = 6d	Dordio and Carvalho (2013)
			89.3 ± 3.1%	contact time = 3d	
		5000	77.0 ± 4.1%	contact time = 6d	
			66.2 ± 3.6%	contact time = 3d	
	oxytetracycline	1000	100 ± 1.8%	contact time = 9d	
			97 ± 2.3%	contact time = 3d	
Pilot(sandy substrate)	dicamba	1.5	0%	VSSF and reverse	Cheng et al. (2002)
	MCPA	2.8	36%		
	parathion	2	100%		
	omethoate	60	100%		
River Besos, Spain	mecoprop	7.80 ± 3.24	79–91%	FWS, HRT = 30d	Matamoros et al. (2008)
	MCPA	2.01 ± 1.50	79–93%		
	terbutylazine	2.30 ± 1.82	1–80%		
	flunixin	1.06 ± 1.36	0–64%		
pilot	imazalil	10	80–97%	Summer, HRT = 0.25d	Lv et al. (2016)
			78–88%	Winter, HRT = 2d	
			100%	Summer, HRT = 2d	
		100	92–100%	Summer, HRT = 2d	
	tebuconazole	10	80–100%	Summer, HRT = 2d	
		100	80–99%		
stormwater wetland, Rouffach, France	dimethomorph	3.4/1693 (mg load)	100%/76%	Spring/summer, HRT = 14d	Maillard et al. (2011)
	diuron	26.5/13.2 (mg load)	72%/57%	Spring/summer, HRT = 14d	
	glyphosate	585/3571 (mg load)	90%/77%	Spring/summer, HRT = 14d	
	simazine	13/2.1 (mg load)	36%/60%	Spring/summer, HRT = 14d	
	terbuthylazine	10.4/3.8 (mg load)	100%	Spring/summer, HRT = 14d	
Aulnoy and Bray catchment, France	Total pesticide	0.73	54%	subsurface drained, in-stream	Tournebize et al. (2013b)
		4.9	45%	subsurface drained, off-stream	
batch	Total macrolides	100	43.7–67.6%	–	Tai et al. (2017)
		300	44.3–82.2%		
pilot	Total antibiotics	–	75.8–98.6%	DO<0.5 mg•L^{-1},	Chen et al. (2016)
	erythromycin-H$_2$O	–	76.0–97.2%	FWS, VSDF, VSUF, HSF	
	leucomycin	–	100%		
	sulfamethazine	–	100%		
	monensin	–	79.4–86.3%		
	clarithromycin	–	67.2–87.3%		
	trimethoprim	–	84.3–84.6%		
	sul1	–	70.0–86.7%		
	sul2	–	47.2–79.1%		
	tetG	–	79.7–92.9%		
	floR	–	82.8–94.6%		
landfill site, Singapore	sulfamethazine	339(62–438) ng•L^{-1}	>99.9%	Hybrid CW,	Yi et al. (2017)
	sul1	5.4•10^{-2}(copies/16S)	96.9%	HRT = 22.75d	
	sul2	2.4•10^{-2}(copies/16S)	95.2%		
	aac6	5.4•10^{-5}(copies/16S)	55.9%		
	qnrA	2.0•10^{-6}(copies/16S)	91.5%		
Pilot(BER + MFC)	sulfamethoxazole	2000	99.48%	BER, HRT = 2.5d, contact time = 60d	Zhang et al. (2018a)
		4000	99.36%		
		2000	99.38%	HRT = 2.5d, contact time = 60d,	
		4000	99.51%	Direct voltage (1000 mV)	
pilot	sulfamethoxazole	200/500/800	98.4–99.9%	VSUF,	Song et al. (2018b)
	tetracycline	200/500/800	98.8–99.9%	relative humidity = 55–65%, HRT = 2.5d	
pilot	*tetA*	4.4•10^{-4}(copies/16S)	−5.21(fold)	HSF, HRT = 1.2d	Nõlvak et al. (2013)
	tetB	2.51•10^{-4}(copies/16S)	−12.09(fold)		
	qnrS	4.05•10^{-4}(copies/16S)	−181.76(fold)		
Pilot	sulfamethoxazole	2000/4000	>99%	MFC + BER, HRT = 16 h	Li et al. (2018)
			>97%	MFC + BER, HRT = 4 h	
	sul1/*sul2*	–	0.02(copies/16S)	MFC, HRT = 16 h	
			0.04(copies/16S)	BER, HRT = 16 h	
	sul3	–	5.0•10^{-5}(copies/16S)	MFC, HRT = 16 h	
			1.0•10^{-4}(copies/16S)	BER, HRT = 16 h	
Field work, Netherlands	erythromycin	19–21 ng•L^{-1}	>75%	VSF, HRT = 0.82-4d	He et al. (2018)
	sulfamethoxazole	55–74 ng•L^{-1}	>75%		
	lincomycin	4-5n ng•L^{-1}	30–60%		
	sul1/*sul2*/*ermB*	10^{-4}–10^{-3}	−10(fold)		

MCPA: 2-Methyl-4-chlorophenoxyacetic acid, VSSF: vertical subsurface flow; VSDF: vertical subsurface down flow; VSUF: vertical subsurface up flow; FWS: free water surface; HSF: horizontal subsurface flow; MFC: microbial fuel cell; BER: biofilm electrode reactor.

genes increased in the effluent (Song et al., 2018b). In an integrated surface flow CW, *sul3, tetA, tetC, tetE* and *qnrS* resistance genes abundance significantly increased ($p < 0.05$) in specific subsystems. The abundance of antibiotic resistant bacteria tended to increase in the effluents of a vertical flow CW (Huang et al., 2017c). Besides, ARGs exchange between water and soil, as well as potential HGT is observed (Fang et al., 2017).

2.4. *Permeable reactive barrier (PRB)*

2.4.1. *Technology description*

Since the invention of PRB in the early 1990s, a variety of materials has been employed to remove contaminants including heavy metals, chlorinated solvents, aromatic hydrocarbons, and pesticides (Henderson and Demond, 2007; Obiri-Nyarko et al., 2014). The main mechanism of PRB *in-situ* remediation is to trap the groundwater flow by creating a hydrological high permeable area, and create an active zone between the filling materials and targeting compounds. Common reactive materials include zero valent iron (ZVI) (Ludwig et al., 2009), activated carbons (AC) (Bortone et al., 2013), zeolites(Vignola et al., 2011), apatite(Oliva et al., 2011) and sodium dithionite (Arun Gavaskar et al., 2000; ITRC, 2011; Vodyanitskii, 2014).

The principal advantages of PRB are the sustainability, cost effectivity, long-term performance and low monitoring frequency requirement compared with *ex-situ* techniques (Higgins and Olson, 2009). Robertson et al. investigated the long-term removal rate of nitrate (50–100%) in a pilot scale Bio-PRB consisting of sand and sawdust mixtures during operation of 15 years (Robertson et al., 2008). The long-term performance of a ZVI PRB for trichloroethene degradation (81 to >99% in 22 years) was evaluated by geochemical and isotope approach (Wilkin et al., 2018).

As an alternative contaminants blocking technique in areas with considerable groundwater and surface water exchange, PRBs are promising also because of the innovation on the reactive materials, including green and brown waste, functional bacteria, modified activated carbon and nanoparticles (Karn et al., 2009). Woodchips and elemental sulfur were used as co-electron donors for heterotrophic and autotrophic denitrification process of *Thiobacillus* bacteria in a batch PRB, and the nitrate removal reached 7.57%, 12.98%, and 13.46% for insufficient, stoichiometric and excessive sulfur dosage respectively (Li et al., 2016). Another permeable reactive biobarrier (PRBB) using *Trichoderma longibrachiatum* over nylon sponge as bioreactive medium for phenanthrene removal was reported and the reduction efficiency reached 90% (14 d) in aqueous medium and 70% (28d) in soil (Cobas et al., 2013). A 135 mg•L^{-1} nitrate in groundwater was removed by over 60% during 130 h operation in a lab-scale EKPRBs filled with modified activated carbon (Ghaeminia and Mokhtarani, 2018). Bio-chemical redox barriers are also widely applied by stimulating natural anaerobic metal-reducing bacteria with available organic carbon sources (Vodyanitskii, 2014). Besides, new configuration such as funnel-and-gate has been used to solve the problems of blocking and loss of permeability during PRB operation (Hosseini et al., 2011). Microwave based regenerating technology was adopted in Cs-contaminated groundwater treatment PRB to resolve the efficiency reduction due to GAC saturation (Falciglia et al., 2020). All of the above innovations extended the flexibility and longevity of PRBs.

2.4.2. *Agro-OMPs removal efficiency*

Although PRB are originally introduced for industrial point source pollution remediation, lab experience suggests it as robust for agro-OMPs remediation (Kladivko, 1999). The reviewed cases were shown in Table 5.

An extensive coverage of common pesticides remediation by PRB is found. Atrazine and fenamiphos were removed <5% by a lab-scale PRB column with Polymer Mats (Patterson et al., 2002). Clay barrier modified with cationic surfactant was reported as effective for retention of linuron, atrazine and metalaxyl (86.2–96.7%, 0.47–56.4% and 1.36–34.5% respectively) in sandy soil (Rodríguez-Cruz et al., 2007). Redox PRB focusing on Cl-containing hydrocarbons and pesticides was effective by introducing electron donors. Chlorinated organics were removed *via* ZVI oxidation accompanied by reductive dechlorination (Henderson and Demond, 2007). A lab-scale sequential permeable reactive barrier filled with palladium coated iron and anaerobic microbes was effective for 2,4,6-trichlorophenol remediation in groundwater by a complete reductive dechlorination at the residence time of 30.24–21.16 h and a completely removal of phenol by biodegradation within 7–8 d (Choi et al., 2007). 2,4-dichlorophenol was removed by over 90% in a lab-scale PRB filled with surface hydrophobic granular

Table 5
Reported agro-OMPs removal in RBF systems.

Location	Target pollutants	Concentration	Removal efficiency	Parameters	References
pilot	atrazine and fenamiphos	–	<5% in 10d	Polymer Mats filled PRB	Patterson et al. (2002)
pilot	2,4,6-TCP	100 mg•L^{-1}	100% in 7-8d	Pd/Fe PRB & anaerobic microbes	Choi et al. (2007)
pilot	linuron, atrazine, metalaxyl	1000 mg•L^{-1}	86.2–96.7%, 0.47–56.4%, 1.36–34.5%	Clay PRB modified with cationic surfactant in sandy soil	Rodríguez-Cruz et al. (2007)
pilot	2.4-DCP	100 mg•kg^{-1}	54.92% in 10.5d, polarity-reversal intervals of 24 h;24.98% in 10.5d, polarity-reversal intervals of 12 h	EK (0-60 V) & activated bamboo charcoal PRB	Ma et al. (2010)
pilot	2.4-DCP	15 mg•L^{-1}	>90% in 5 h	ZVI & surface hydrophobic GAC PRB	Yang et al. (2010)
pilot	HCB	0.5 mg•L^{-1}	60% in 24 h	Pd/Fe PRB & Triton X-100	Wan et al. (2010)
pilot	PCP	15 mg•L^{-1}	49% removed and 22.9% to phenol	Pd/Fe PRB & EK (20 V)	Li et al. (2011)
batch	PAHs	100–400 μM	90% of aqueous in 14d, 70% of soil in 28d	PRBB using *Trichoderma longibrachiatum*	Cobas et al. (2013)
batch	bismerthiazol	0.01 M 0.1 M	>90% in 24 h	ZVI PRB	Shen et al. (2015)
pilot	tetracycline	20 mg•L^{-1}	50% in 110d, 40% in 110d, 10% in 110d	ZVI & M PRB ZVI PRB M PRB	Huang et al. (2017a)
bench	atrazine & oxyfluorfe	100 mg•dm^{-3}	90% in 15d (~15% adsorbed, 75% evaporated)	reversible EK (1 V•cm^{-1}*20 cm) & GAC PRB	Vieira Dos Santos et al. (2017)
bench	clopyralid	500 mg•dm^{-3}	16% in 30d (6% volatilized, 10% adsorbed), 31% in 30d (11% volatilized, 20% adsorbed)	reversible EK (1 V•cm^{-1}*38 cm) & GAC PRB EK (1 V•cm^{-1}*38 cm) & GAC PRB	Rodrigo et al. (2018)
bench	clopyralid	20 mg•kg^{-1}	35.9–57.9% in a month	EKSF (1 V•cm^{-1}*38 cm) & ZVI PRB	Vidal et al. (2018)

2,4,6-TCP: 2,4,6-trichlorophenol; 2.4-DCP: 2,4-dichlorophenol; HCB: hexachlorobenzene; PCP: pentachlorophenol; PAHs: polycyclic aromatic hydrocarbons; PRB: Permeable reactive barrier; EK: Electrokinetic; EKSF: Electrokinetic soil fushing; GAC: Granular active carbon; M: microorganism.

activated carbon and elemental iron (Yang et al., 2010). Layered double hydroxide PRB was also reported as effective for concentrated 2,4-dichlorophenol and dicamba removal by surface adsorption and interlayer ion exchange (Zolgharnein et al., 2011). The thiadiazole fungicides bismerthiazol was removed with the efficiency of >90% within 24 h by a batch-scale ZVI PRB resulting from surface complex interaction (Shen et al., 2015).

Among the reviewed publications, data about antibiotics removal in PRBs is few. Tetracycline were removed by 50%, 40% and 10% respectively in three lab-scale PRB columns packed with ZVI & microorganism, ZVI and microorganism (Huang et al., 2017a). Tetracycline was efficiently removed in MnO_2-ZVI based PRB system by 85%, higher than PRB with only ZVI (65%) and MnO_2 (50%) (Dong et al., 2018).

Some pilot and field-scale practices on other agriculture pollutants such as ammonium, nitrate and phosphorus have emerged. Permeable reactive interceptor (PRI) is introduced as a more holistic approach for blocking mixed agricultural diffuse pollutants (Fenton et al., 2014). A field-scale zeolite PRB reduced the ammonium concentrations in groundwater from 2 to 10 $mg \bullet L^{-1}$ to <0.5 $mg \bullet L^{-1}$ (Hou et al., 2014). Another pilot-scale research reported a two-layer heterotrophic-autotrophic denitrification PRB for groundwater nitrate remediation in an oxygen rich environment (Huang et al., 2015), with a removal of >91% and little accumulation of NO_2^- and NH_4^+. The nitrate reduction efficiency of a field ZVI-PRB was 15–20%, and the ZVI-induced abiotic nitrate reduction and denitrification was confirmed by chemical and isotopic evidence (Grau-Martínez et al., 2019). A field PRB filled with gravel and mulch was successful at removing nitrate in groundwater (>280 $mg \bullet L^{-1}$) with the efficiency of >97% by biological denitrification process (Gibert et al., 2019). A similar field PRB filled with processed agriculture straw materials for groundwater nitrate remediation was installed and operated continuously for over 15 months, and the nitrate removal was 60–90% with minimal secondary pollution (Zhang et al., 2020). A non-continuous PRB filled with mixture of ZVI, AC and sand effectively removed nitrate from shallow aquifer by >80% (120 to 20 $mg \bullet L^{-1}$ in a day) (Guan et al., 2019). A tidally influenced pilot PRB was installed behind a perforated marine bulkhead for nitrate removal from the groundwater, and the NO_x recover rate as N_2 reached 75% (Graffam et al., 2020). A field-scale horizontal woodchip PRB installed below managed aquifer recharge facility was reported with an average nitrate removal rate of 1.5 $g \bullet m^{-2} \bullet d^{-1}$ at the residence time of only 6 h (Beganskas et al., 2018). A PRB aiming at phosphate polluted agricultural runoff remediation was effective with P-PO_4 retention of 99%, 98%, 88% and 65% by reactive materials of Polonite®, AAC, zeolite and limestone respectively, which would reduce the agricultural P load significantly (Bus et al., 2019). These applications confirmed the possibility and potential of PRB for agricultural mixed diffuse pollutants remediation.

Electrokinetic (EK) coupled PRB technology (EKPRBs) is recently introduced for heavy metals and organic contaminants *in-situ* removal from soil and groundwater by applying a direct current electric field in the subsurface (Andrade and dos Santos, 2020; Rodrigo et al., 2014; Ye et al., 2019). A bench-scale EK and bamboo charcoal barrier was used for simultaneous removal of 2,4-DCP (100 $mg \bullet kg^{-1}$) and Cd (500 $mg \bullet kg^{-1}$) from a sandy loam (Ma et al., 2010). After 10.5 d, 75.97%, 40.13% Cd and 54.92%, 24.98% 2,4-DCP were removed at intervals of 24 h and 12 h respectively. A bench-scale EK was coupled with Pd/Fe PRB and enhanced by surfactant Triton X-100 for remediation of HCB contaminated soil with a much higher efficiency of 60% compared with EK alone (15%) (Wan et al., 2010). This coupled EKPRB also accomplished 49% removal of PCP and 22.9% phenol recovery at catholyte in remediation of PCP contaminated soil (Li et al., 2011). A reversible EK adsorption barrier filled with granular activated carbon (GAC) was enhanced with sodium dodecyl sulfate for removal of atrazine and oxyfluorfe (Vieira Dos Santos et al., 2017). After 15 days, about 90% of atrazine and oxyfluorfen were removed by carbon bed-adsorption mechanism, and volatilization of herbicides is obviously prevented compared with conventional EK soil flushing (EKSF). A reported EK adsorption barrier and reversible EK adsorption barrier allows organochlorine herbicide clopyralid removal from clay soils of 45 and 57% within 30 d (Rodrigo et al., 2018). Another bench-scale (175 L) ZVI barrier coupled with EKSF was reported as effective for clopyralid remediation in soils, with the formation of picolinic acid, partial dehalogenation to 6-Chloropyridine-2-carboxylic acid and 3-chloropyridine-2-carboxylic acid (Vidal et al., 2018). Besides, most of the clopyralid residue (60%) was concentrated near the anodes and could be easily extracted by a small amount (15%) of soil excavation. The main mechanisms in EK technique include electro-migration, electrophoresis, electro-osmosis, electrolysis and electrical heating (Rodrigo et al., 2014), which improved the applicability and efficiency but with the cost of energy consumption and maintenance. Therefore, more field applications along with the efficiency and economic effectiveness evaluation were needed.

2.5. *Summary and perspectives*

Efforts have been made on application and efficiency evaluation of these *in-situ* techniques (Heberer et al., 2004; Heberer et al., 2008; Lizotte Jr. et al., 2012; Massmann et al., 2008; Mueller et al., 2011; Obiri-Nyarko et al., 2014; Passeport et al., 2014; Vodyanitskii, 2014), while quite distinct behaviors are observed for different agro-OMPs brought by the variety of compounds and different environmental conditions. From this aspect, more profound exploration is needed to assist the schematic and technical design, including the hydrological and geological background, microorganism composition and activity, mass and energy exchange *etc.* Fundamental researches are also needed to comprehensively describe the pathways and removal mechanisms of agro-OMPs in these *in-situ* remediation systems (Nsenga Kumwimba et al., 2018). On the other hand, the installation of *in-situ* remediation system, especially PRBs, might introduce nanoparticles, exogenous organic compounds and by products in the environment, which could pose risk on the ecosystem during water cycling. Long-term monitoring data can provide better understanding of the longevity and potential risk, which will benefit the improvement of each technique. Thus, more field-scale application and exploration of effective combination of these techniques will be effective ways to make innovations on agro-OMPs removal and water-oriented risk control. Some perspectives are summarized in Table 6.

3. Conclusion

1) Agro-OMPs, including PPPs, veterinary antibiotics and concomitant ARGs, are emerging in the aquatic environment through pathways of spray drifting, wastewater drainage, surface/subsurface runoff, seepage and leaching. Therefore, *in-situ* remediation techniques are in demand urgently.
2) Efficient removal of pesticides, herbicides and antibiotics are widely reported in RBF, RBZ and CW, which has already been applied in agricultural diffuse pollution control and water treatment. RBF is effective for many agro-OMPs removal, especially pathogenic microorganism, while the subsequent ecological succession should be paid more attention for stability maintenance. For RBZ, comprehensive arrangement, flexible combination and construction standard should be explored in agriculture area at basin scale for agro-OMPs control. Removal of ARGs are particularly found in CWs, while the accumulation of specific ARGs and the potential risk on human health still exists. Based on batch and pilot experiment, EKPRBs and PRBBs are promising in removal of combined agricultural diffuse pollutants.
3) Sufficient reports exist for agro-OMPs emerging in the soil and the aquatic environment, while the information on field-scale application of *in-situ* remediation techniques and their effective combinations are few. A wide range of removal efficiency are observed for

Table 6
Comparison and perspective of each *in-situ* technique.

Technique	Removal potential for agro-OMPs			Application limits	Future perspectives
	pesticides	antibiotics	ARGs		
River bank filtration	• p,p'-DDA and o,p'-DDA, 50%; • bentazone and mecoprop, 33%–60%; • atrazine, 20%; • carbofuran, 60%.	• sulfamethoxazole, 20–100%; • trimethoprim, 100%; • chloramphenicol, 82%; • 4-formylaminoantipyrin, 35%; • phenazone, 33%; • 4-acetylaminoantipyrine, 10%.	–	• Specific hydrogeological condition. • Subsequent treatment processes when used as drinking water pretreatment technique.	• The capture ability for new emerging pollutants such as ARGs and advanced oxidation products (Wang et al., 2016). • Drinking water risk assessment and control (Farkas et al., 2015).
Riparian buffer zone	• general, 70–94%; • atrazine, 77%, • chlorpyrifos, 4–17%, • cypermethrin, 2–4%, • L-cyhalothrin, 2–5%, • metolachlor, glyphosate, 58–72%.	• sulfamethazine, >70%.	–	• Occupation of considerable broad bank area. • Benefits are 'scale-dependent' which can only be accomplished on large scale, such as tile-drainage practice. • Land use and hydro-morphological degradation in the sub-catchment can limit the benefit.	• The integration of flow and riparian vegetation dynamics in riparian buffer management (Feld et al., 2018). • The application of long-term BACI (*i.e.* before-after-control-impact) design in riparian buffer zone research (Hille et al., 2018). • The development of comprehensive evaluation strategy system at catchment scale based on the modeling of width, length, location, hydrological condition and removal efficiency of different compounds (Shan et al., 2014). • Novel form of edge-of-field RBZ such as integrated buffer zones (pond & filter bed) to reduce high nutrient loading in agricultural area (Zak et al., 2018).
Constructed wetland	• strobilurin/strobing, 96%; • organophosphate, 94%; • pyrethroids, 84%; • triazinone, 24%; • aryloxyalkanoic acid, 35%; • urea-based pesticide, 50%; • 2-methyl-4-chlorophenoxyacetic acid, 56–97%.	• general, 76–99%; • trimethoprim, 99%; • sulfamethoxazole, 63–64%; • anhydroerythromycin A, 81.3–86.9%; • roxithromycin, 60.6–74.0%; • clarithromycin, 68.5–74.6%; • ilmicosin, 62.5–62.7%; • oxytetracycline, 97%.	• total ARGs , reduced by 33–99%; • *sul,sul2,sul3*, reduced; • *qnrA, qnrS, ermB*, reduced; • *tetA, tetC, tetE*, reduced; • *sul* and *tet* genes, increased.	• Drainage water collection system for agricultural non-point source remediation. • Efficiency reduction caused by siltation and clogging.	• The performance of integrated CWs such as microbial fuel cell coupled CW (Song et al., 2018a) and hybrid CW combing horizontal subsurface flow CW with vertical subsurface flow CW (Zhang et al., 2018b). • The application and enhancement of CWs filling substrates such as biochar, zeolite, ion-exchange material and nano particles (Kasak et al., 2018; Yang et al., 2018).
Permeable reactive barrier	• 2,4-DCP & 2,4,6-TCP, 25–100%, • atrazine, 5–90%, • fenamiphos, <5%, • linuron, 86.2–96.7%, • metalaxyl, 1.36–34.5%, • HCB & PCP, 49–60%, • PAHs, 70–90%, • bismerthiazol & bismerthiazol, 90%, • clopyralid, 16–58%.	• tetracycline, 10–85%,	–	• Relatively time and money consuming during construction. • The uncertain long-term effectiveness.	• New form of PRB such as Horizontal Reactive Media Treatment Well to passively capture and treat proportionally large volumes of groundwater (Divine et al., 2018). • Development of coupled PRB and RBZ system to realize the treatment of pollution in both runoff and underground runoff. • Development of new active material such as comprehensive organic carbon material and engineered graphene oxide. • Application of PRBBs and EKPRBs on agro-OMPs removal.

each *in-situ* techniques regarding to different agro-OMPs, which further lucubration is required.

Declaration of Competing Interest

The authors have declared no conflict of interest.

Acknowledgements

This work was supported by National Key R & D Program of China (Grant No. 2018YFC0408005), Basic Research Foundation of National Public Research Institutes of China (Grant No. Y521008) and China Scholarship Council (201506190085).

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecoinf.2021.101517.

References

Aguiar, T.R., Bortolozo, F.R., Hansel, F.A., Rasera, K., Ferreira, M.T., 2015. Riparian buffer zones as pesticide filters of no-till crops. Environ. Sci. Pollut. Res. 22, 10618–10626.

Aldén Demoling, L., Bååth, E., 2008. The use of leucine incorporation to determine the toxicity of phenols to bacterial communities extracted from soil. Appl. Soil Ecol. 38, 34–41.

Andrade, D.C., dos Santos, E.V., 2020. Combination of electrokinetic remediation with permeable reactive barriers to remove organic compounds from soils. Current Opinion in Electrochemistry 22, 136–144.

Arora, Kapil, Mickelson, Steven K., Helmers, Matthew J., Baker, J.L., 2010. Review of pesticide retention processes occurring in buffer strips receiving agricultural runoff. J. Am. Water Resour. Assoc. 46, 618–647.

Arun Gavaskar, N.G., Sass, Bruce, Janosy, Robert, Hicks, James, 2000. Design Guidance for Application of Permeable Reactive Barriers for Groundwater Remediation. US Air Force Research Laboratory.

Bai, Y., Ruan, X., Wang, F., Antoine, G., van der Hoek, J.P., 2019. Sulfonamides removal under different redox conditions and microbial response to sulfonamides stress during riverbank filtration: a laboratory column study. Chemosphere 220, 668–677.

Beganskas, S., Gorski, G., Weathers, T., Fisher, A.T., Schmidt, C., Saltikov, C., Redford, K., Stoneburner, B., Harmon, R., Weir, W., 2018. A horizontal permeable reactive barrier stimulates nitrate removal and shifts microbial ecology during rapid infiltration for managed recharge. Water Res. 144, 274–284.

Benner, J., Helbling, D.E., Kohler, H.-P.E., Wittebol, J., Kaiser, E., Prasse, C., Ternes, T.A., Albers, C.N., Aamand, J., Horemans, B., Springael, D., Walravens, E., Boon, N., 2013. Is biological treatment a viable alternative for micropollutant removal in drinking water treatment processes? Water Res. 47, 5955–5976.

Benotti, M.J., Song, R., Wilson, D., Snyder, S.A., 2012. Removal of pharmaceuticals and endocrine disrupting compounds through pilot- and full-scale riverbank filtration. Water Science and Technology-Water Supply 12, 11–23.

Berglund, B., Khan, G.A., Weisner, S.E.B., Ehde, P.M., Fick, J., Lindgren, P.-E., 2014. Efficient removal of antibiotics in surface-flow constructed wetlands, with no observed impact on antibiotic resistance genes. Sci. Total Environ. 476–477, 29–37.

Bertelkamp, C., 2015. Organic Micropollutant Removal during River Bank Filtration. Technology University of Delft, The Netherlands, pp. 13–14.

Bertelkamp, C., Reungoat, J., Cornelissen, E.R., Singhal, N., Reynisson, J., Cabo, A.J., van der Hoek, J.P., Verliefde, A.R.D., 2014. Sorption and biodegradation of organic micropollutants during river bank filtration: a laboratory column study. Water Res. 52, 231–241.

Bertelkamp, C., Schoutteten, K., Vanhaecke, L., Vanden Bussche, J., Callewaert, C., Boon, N., Singhal, N., van der Hoek, J.P., Verliefde, A.R.D., 2015. A laboratory-scale column study comparing organic micropollutant removal and microbial diversity for two soil types. Sci. Total Environ. 536, 632–638.

Bertelkamp, C., van der Hoek, J.P., Schoutteten, K., Hulpiau, L., Vanhaecke, L., Vanden Bussche, J., Cabo, A.J., Callewaert, C., Boon, N., Löwenberg, J., Singhal, N., Verliefde, A.R.D., 2016a. The effect of feed water dissolved organic carbon concentration and composition on organic micropollutant removal and microbial diversity in soil columns simulating river bank filtration. Chemosphere 144, 932–939.

Bertelkamp, C., Verliefde, A.R.D., Schoutteten, K., Vanhaecke, L., Vanden Bussche, J., Singhal, N., van der Hoek, J.P., 2016b. The effect of redox conditions and adaptation time on organic micropollutant removal during river bank filtration: a laboratory-scale column study. Sci. Total Environ. 544, 309–318.

Birnbaum, L.S., Gunier, R.B., Bradman, A., Harley, K.G., Eskenazi, B., 2017. Will buffer zones around schools in agricultural areas be adequate to protect children from the potential adverse effects of pesticide exposure? PLoS Biol. 15, e2004741.

Bortone, I., Di Nardo, A., Di Natale, M., Erto, A., Musmarra, D., Santonastaso, G.F., 2013. Remediation of an aquifer polluted with dissolved tetrachloroethylene by an array of wells filled with activated carbon. J. Hazard. Mater. 260, 914–920.

Bus, A., Karczmarczyk, A., Baryła, A., 2019. Permeable reactive barriers for preventing water bodies from a phosphorus-polluted agricultural runoff-column experiment. Water 11, 432.

Calderon-Preciado, D., Jimenez-Cartagena, C., Matamoros, V., Bayona, J.M., 2011. Screening of 47 organic microcontaminants in agricultural irrigation waters and their soil loading. Water Res. 45, 221–231.

Carluer, N., Tournebize, J., Gouy, V., Margoum, C., Vincent, B., Gril, J.J., 2011. Role of buffer zones in controlling pesticides fluxes to surface waters. Procedia Environ. Sci. 9, 21–26.

Castiglioni, S., Pomati, F., Miller, K., Burns, B.P., Zuccato, E., Calamari, D., Neilan, B.A., 2008. Novel homologs of the multiple resistance regulator marA in antibiotic-contaminated environments. Water Res. 42, 4271–4280.

Chen, J., Ying, G.-G., Wei, X.-D., Liu, Y.-S., Liu, S.-S., Hu, L.-X., He, L.-Y., Chen, Z.-F., Chen, F.-R., Yang, Y.-Q., 2016. Removal of antibiotics and antibiotic resistance genes from domestic sewage by constructed wetlands: effect of flow configuration and plant species. Sci. Total Environ. 571, 974–982.

Cheng, S., Vidakovic-Cifrek, Z., Grosse, W., Karrenbrock, F., 2002. Xenobiotics removal from polluted water by a multifunctional constructed wetland. Chemosphere 48, 415–418.

Choi, J.H., Kim, Y.H., Choi, S.J., 2007. Reductive dechlorination and biodegradation of 2,4,6-trichlorophenol using sequential permeable reactive barriers: laboratory studies. Chemosphere 67, 1551–1557.

Cobas, M., Ferreira, L., Tavares, T., Sanroman, M.A., Pazos, M., 2013. Development of permeable reactive biobarrier for the removal of PAHs by Trichoderma longibrachiatum. Chemosphere 91, 711–716.

Divine, C.E., Roth, T., Crimi, M., DiMarco, A.C., Spurlin, M., Gillow, J., Leone, G., 2018. The horizontal reactive media treatment well (HRX well®) for passive in-situ remediation. Groundwater Monitoring & Remediation 38, 56–65.

Dong, G., Huang, L., Wu, X., Wang, C., Liu, Y., Liu, G., Wang, L., Liu, X., Xia, H., 2018. Effect and mechanism analysis of MnO 2 on permeable reactive barrier (PRB) system for the removal of tetracycline. Chemosphere 193, 702–710.

Dordio, A., Carvalho, A.J.P., 2013. Constructed wetlands with light expanded clay aggregates for agricultural wastewater treatment. Sci. Total Environ. 463, 454–461.

Elsayed, O.F., Maillard, E., Vuilleumier, S., Imfeld, G., 2014. Bacterial communities in batch and continuous-flow wetlands treating the herbicide S-metolachlor. Sci. Total Environ. 499, 327–335.

Entry, J.A., Donelly, P.K., Emmingham, W.H., 1994. Microbial mineralization of atrazine and 2,4-dichlorophenoxyacetic acid in riparian pasture and forest soils. Biol. Fertil. Soils 18, 89–94.

Falciglia, P.P., Gagliano, E., Brancato, V., Finocchiaro, G., Catalfo, A., De Guidi, G., Romano, S., Roccaro, P., Vagliasindi, F.G.A., 2020. Field technical applicability and cost analysis for microwave based regenerating permeable reactive barriers (MW-PRBs) operating in Cs-contaminated groundwater treatment. J. Environ. Manag. 260, 110064.

Fang, H., Zhang, Q., Nie, X., Chen, B., Xiao, Y., Zhou, Q., Liao, W., Liang, X., 2017. Occurrence and elimination of antibiotic resistance genes in a long-term operation integrated surface flow constructed wetland. Chemosphere 173, 99–106.

Farkas, K., Plutzer, J., Moltchanova, E., Török, A., Varró, M.J., Domokos, K., Frost, F., Hunter, P.R., 2015. Serological responses to Cryptosporidium antigens in inhabitants of Hungary using conventionally filtered surface water and riverbank filtered drinking water. Epidemiol. Infect. 143, 2743–2747.

Feld, C.K., Fernandes, M.R., Ferreira, M.T., Hering, D., Ormerod, S.J., Venohr, M., Gutiérrez-Cánovas, C., 2018. Evaluating riparian solutions to multiple stressor problems in river ecosystems — A conceptual study. Water Res. 139, 381–394.

Fenton, O., Healy, M.G., Brennan, F., Jahangir, M.M.R., Lanigan, G.J., Richards, K.G., Thornton, S.F., Ibrahim, T.G., 2014. Permeable reactive interceptors: blocking diffuse nutrient and greenhouse gases losses in key areas of the farming landscape. JAS 152, 71–81.

Ghaeminia, M., Mokhtarani, N., 2018. Remediation of nitrate-contaminated groundwater by PRB-Electrokinetic integrated process. J. Environ. Manag. 222, 234–241.

Gibert, O., Assal, A., Devlin, H., Elliot, T., Kalin, R.M., 2019. Performance of a field-scale biological permeable reactive barrier for in-situ remediation of nitrate-contaminated groundwater. Sci. Total Environ. 659, 211–220.

Gollnitz, W.D., Clancy, J.L., Whitteberry, B.L., Vogt, A.J.A., 2003. RBF as a microbial treatment process. Journal American Water Works Association. 95 (12), 56–66.

Gorito, A.M., Ribeiro, A.R., Almeida, C.M.R., Silva, A.M.T., 2017. A review on the application of constructed wetlands for the removal of priority substances and contaminants of emerging concern listed in recently launched EU legislation. Environ. Pollut. 227, 428–443.

Graffam, M., Paulsen, R., Volkenborn, N., 2020. Hydro-biogeochemical processes and nitrogen removal potential of a tidally influenced permeable reactive barrier behind a perforated marine bulkhead. Ecol. Eng. 155, 105933.

Grau-Martínez, A., Torrentó, C., Carrey, R., Soler, A., Otero, N., 2019. Isotopic evidence of nitrate degradation by a zero-valent iron permeable reactive barrier: batch experiments and a field scale study. J. Hydrol. 570, 69–79.

Gregoire, C., Elsaesser, D., Huguenot, D., Lange, J., Lebeau, T., Merli, A., Mose, R., Passeport, E., Payraudeau, S., Schuetz, T., Schulz, R., Tapia-Padilla, G., Tournebize, J., Trevisan, M., Wanko, A., 2009. Mitigation of agricultural nonpoint-source pesticide pollution in artificial wetland ecosystems. Environ. Chem. Lett. 7, 205–231.

Grung, M., Lin, Y., Zhang, H., Steen, A.O., Huang, J., Zhang, G., Larssen, T., 2015. Pesticide levels and environmental risk in aquatic environments in China - a review. Environ. Int. 81, 87–97.

Guan, Q., Li, F., Chen, X., Tian, C., Liu, C., Liu, D., 2019. Assessment of the use of a zero-valent iron permeable reactive barrier for nitrate removal from groundwater in the alluvial plain of the Dagu River, China. Environmental Earth Sciences 78.

He, Y., Nurul, S., Schmitt, H., Sutton, N.B., Murk, T.A.J., Blokland, M.H., Rijnaarts, H.H. M., Langenhoff, A.A.M., 2018. Evaluation of attenuation of pharmaceuticals, toxic potency, and antibiotic resistance genes in constructed wetlands treating wastewater effluents. Sci. Total Environ. 631-632, 1572–1581.

Heberer, T., 2002. Occurrence, fate, and removal of pharmaceutical residues in the aquatic environment: a review of recent research data. Toxicol. Lett. 131, 5–17.

Heberer, T., Mechlinski, A., Fanck, B., Knappe, A., Massmann, G., Pekdeger, A., Fritz, B., 2004. Field studies on the fate and transport of pharmaceutical residues in bank filtration. Ground Water Monitoring and Remediation 24, 70–77.

Heberer, T., Massmann, G., Fanck, B., Taute, T., Dünnbier, U., 2008. Behaviour and redox sensitivity of antimicrobial residues during bank filtration. Chemosphere 73, 451–460.

Henderson, A.D., Demond, A.H., 2007. Long-term performance of zero-valent iron permeable reactive barriers: a critical review. Environ. Eng. Sci. 24, 401–423.

Heuer, H., Schmitt, H., Smalla, K., 2011. Antibiotic resistance gene spread due to manure application on agricultural fields. Curr. Opin. Microbiol. 14, 236–243.

Higgins, M.R., Olson, T.M., 2009. Life-cycle case study comparison of permeable reactive barrier versus pump-and-treat remediation. Environ. Sci. Technol 43 (24), 9432–9438.

Hille, S., Andersen, D.K., Kronvang, B., Baattrup-Pedersen, A., 2018. Structural and functional characteristics of buffer strip vegetation in an agricultural landscape – high potential for nutrient removal but low potential for plant biodiversity. Sci. Total Environ. 628-629, 805–814.

Hjorth, M., Dahllöf, I., Forbes, V.E., 2006. Effects on the function of three trophic levels in marine plankton communities under stress from the antifouling compound zinc pyrithione. Aquat. Toxicol. 77, 105–115.

van der Hoek, J.P., Bertelkamp, C., Verliefde, A.R.D., Singhal, N., 2014. Drinking water treatment technologies in Europe: state of the art – challenges – research needs. J. Water Supply Res. Technol. AQUA 63, 124–130.

Hosseini, M., Ataie-Ashtiani, B., Kholghi, M., 2011. Bench-scaled Nano-Fe0 permeable reactive barrier for nitrate removal. Ground Water Monitoring & Remediation 31.

Hou, G., Liu, F., Liu, M., Kong, X., Li, S., Chen, L., Colberg, P.J.S., Jin, S., Chen, H., 2014. Performance of a permeable reactive barrier for in situ removal of ammonium in groundwater. Water Science and Technology-Water Supply 14, 585–592.

Hu, J., Shi, J., Chang, H., Li, D., Yang, M., Kamagata, Y., 2008. Phenotyping and genotyping of antihiotic-resistant Escherichia coli isolated from a natural river basin. Environ. Sci. Technol. 42, 3415–3420.

Huang, G., Huang, Y., Hu, H., Liu, F., Zhang, Y., Deng, R., 2015. Remediation of nitrate-nitrogen contaminated groundwater using a pilot-scale two-layer heterotrophic-autotrophic denitrification permeable reactive barrier with spongy iron/pine bark. Chemosphere 130, 8–16.

Huang, L., Liu, G., Dong, G., Wu, X., Wang, C., Liu, Y., 2017a. Reaction mechanism of zero-valent iron coupling with microbe to degrade tetracycline in permeable reactive barrier (PRB). Chem. Eng. J. 316, 525–533.

Huang, X., Zheng, J., Liu, C., Liu, L., Liu, Y., Fan, H., 2017b. Removal of antibiotics and resistance genes from swine wastewater using vertical flow constructed wetlands: effect of hydraulic flow direction and substrate type. Chem. Eng. J. 308, 692–699.

Huang, X., Zheng, J., Liu, C., Liu, L., Liu, Y., Fan, H., Zhang, T., 2017c. Performance and bacterial community dynamics of vertical flow constructed wetlands during the treatment of antibiotics-enriched swine wastewater. Chem. Eng. J. 316, 727–735.

Huddleston, J.R., 2014. Horizontal gene transfer in the human gastrointestinal tract: potential spread of antibiotic resistance genes. Infection and Drug Resistance 7, 167–176.

Hunt, L., Marrochi, N., Bonetto, C., Liess, M., Buss, D.F., Vieira da Silva, C., Chiu, M.C., Resh, V.H., 2017. Do riparian buffers protect stream invertebrate communities in south American Atlantic Forest agricultural areas? Environ. Manag. 60, 1155–1170.

Imfeld, G., Vuilleumier, S., 2012. Measuring the effects of pesticides on bacterial communities in soil: a critical review. Eur. J. Soil Biol. 49, 22–30.

ITRC, 2011. Permeable Reactive Barrier: Technology Update, Washington, DC.

Joergensen, R.G., Emmerling, C., 2006. Methods for evaluating human impact on soil microorganisms based on their activity, biomass, and diversity in agricultural soils. J. Plant Nutr. Soil Sci. 169, 295–309.

Karn, B., Kuiken, T., Otto, M., 2009. Nanotechnology and in situ remediation: a review of the benefits and potential risks. Environ. Health Perspect. 117, 1823–1831.

Kasak, K., Truu, J., Ostonen, I., Sarjas, J., Oopkaup, K., Paiste, P., Kõiv-Vainik, M., Mander, Ü., Truu, M., 2018. Biochar enhances plant growth and nutrient removal in horizontal subsurface flow constructed wetlands. Sci. Total Environ. 639, 67–74.

Kladivko, E.J., 1999. Pesticide and nitrate transport into subsurface tile drains of different spacings. J. Environ. Qual. 28, 997–1004.

Koike, S., Krapac, I.G., Oliver, H.D., Yannarell, A.C., Chee-Sanford, J.C., Aminov, R.I., Mackie, R.I., 2007. Monitoring and source tracking of tetracycline resistance genes in lagoons and groundwater adjacent to swine production facilities over a 3-year period. Appl. Environ. Microbiol. 73, 4813–4823.

Kovacevic, S., Radisic, M., Lausevic, M., Dimkic, M., 2017. Occurrence and behavior of selected pharmaceuticals during riverbank filtration in the republic of Serbia. Environ. Sci. Pollut. Res. 24, 2075–2088.

Li, H., Song, H.-L., Yang, X.-L., Zhang, S., Yang, Y.-L., Zhang, L.-M., Xu, H., Wang, Y.-W., 2018. A continuous flow MFC-CW coupled with a biofilm electrode reactor to simultaneously attenuate sulfamethoxazole and its corresponding resistance genes. Sci. Total Environ. 637-638, 295–305.

Li, R., Feng, C., Hu, W., Xi, B., Chen, N., Zhao, B., Liu, Y., Hao, C., Pu, J., 2016. Woodchip-sulfur based heterotrophic and autotrophic denitrification (WSHAD) process for nitrate contaminated water remediation. Water Res. 89, 171–179.

Li, Z., Yuan, S., Wan, J., Long, H., Tong, M., 2011. A combination of electrokinetics and Pd/Fe PRB for the remediation of pentachlorophenol-contaminated soil. J. Contam. Hydrol. 124, 99–107.

Lin, C.H., Lerch, R.N., Goyne, K.W., Garrett, H.E., 2011. Reducing herbicides and veterinary antibiotics losses from agroecosystems using vegetative buffers. J. Environ. Qual. 40, 791–799.

Liu, L., Liu, Y.-H., Wang, Z., Liu, C.-X., Huang, X., Zhu, G.-F., 2014. Behavior of tetracycline and sulfamethazine with corresponding resistance genes from swine wastewater in pilot-scale constructed wetlands. J. Hazard. Mater. 278, 304–310.

Lizotte Jr., R.E., Shields Jr., F.D., Murdock, J.N., Kroeger, R., Knight, S.S., 2012. Mitigating agrichemicals from an artificial runoff event using a managed riverine wetland. Sci. Total Environ. 427, 373–381.

Ludwig, R.D., Smyth, D.J.A., Blowes, D.W., Spink, L.E., Wilkin, R.T., Jewett, D.G., Weisener, C.J., 2009. Treatment of arsenic, heavy metals, and acidity using a mixed ZVI-compost PRB. Environ. Sci. Technol. 43, 1970–1976.

Lv, T., Zhang, Y., Zhang, L., Carvalho, P.N., Arias, C.A., Brix, H., 2016. Removal of the pesticides imazalil and tebuconazole in saturated constructed wetland mesocosms. Water Res. 91, 126–136.

Ma, J.W., Wang, F.Y., Huang, Z.H., Wang, H., 2010. Simultaneous removal of 2,4-dichlorophenol and Cd from soils by electrokinetic remediation combined with activated bamboo charcoal. J. Hazard. Mater. 176, 715–720.

Maeng, S.K., Sharma, S.K., Lekkerkerker-Teunissen, K., Amy, G.L., 2011. Occurrence and fate of bulk organic matter and pharmaceutically active compounds in managed aquifer recharge: a review. Water Res. 45, 3015–3033.

Maillard, E., Payraudeau, S., Faivre, E., Grégoire, C., Gangloff, S., Imfeld, G., 2011. Removal of pesticide mixtures in a stormwater wetland collecting runoff from a vineyard catchment. Sci. Total Environ. 409, 2317–2324.

Massmann, G., Duennboer, U., Heberer, T., Taute, T., 2008. Behaviour and redox sensitivity of pharmaceutical residues during bank filtration - investigation of residues of phenazone-type analgesics. Chemosphere 71, 1476–1485.

Matamoros, V., García, J., Bayona, J.M., 2008. Organic micropollutant removal in a full-scale surface flow constructed wetland fed with secondary effluent. Water Res. 42, 653–660.

Michelini, L., Meggio, F., Reichel, R., Thiele-Bruhn, S., Pitacco, A., Scattolin, L., Montecchio, L., Alberghini, S., Squartini, A., Ghisi, R., 2015. Sulfadiazine uptake and effects in common hazel (Corylus avellana L.). Environ. Sci. Pollut. Res. 22, 13362–13371.

Mueller, B., Scheytt, T., Zippel, M., Hannappel, S., Klein-Goedicke, J., Duscher, K., 2011. A new approach to calculate EMEA's predicted environmental concentration for human Pharmaceuticals in Groundwater at Bank filtration sites. Water Air and Soil Pollution 217, 67–82.

Nõlvak, H., Truu, M., Tiirik, K., Oopkaup, K., Sildvee, T., Kaasik, A., Mander, Ü., Truu, J., 2013. Dynamics of antibiotic resistance genes and their relationships with system treatment efficiency in a horizontal subsurface flow constructed wetland. Sci. Total Environ. 461–462, 636–644.

Nsenga Kumwimba, M., Meng, F., Iseyemi, O., Moore, M.T., Zhu, B., Tao, W., Liang, T.J., Ilunga, L., 2018. Removal of non-point source pollutants from domestic sewage and agricultural runoff by vegetated drainage ditches (VDDs): design, mechanism, management strategies, and future directions. Sci. Total Environ. 639, 742–759.

Obiri-Nyarko, F., Grajales-Mesa, S.J., Malina, G., 2014. An overview of permeable reactive barriers for in situ sustainable groundwater remediation. Chemosphere 111, 243–259.

Oliva, J., De Pablo, J., Cortina, J.L., Cama, J., Ayora, C., 2011. Removal of cadmium, copper, nickel, cobalt and mercury from water by apatite II: column experiments. J. Hazard. Mater. 194, 312–323.

Passeport, E., Benoit, P., Bergheaud, V., Coquet, Y., Tournebize, J., 2011a. Epoxiconazole degradation from artificial wetland and forest buffer substrates under flooded conditions. Chem. Eng. J. 173, 760–765.

Passeport, E., Benoit, P., Bergheaud, V., Coquet, Y., Tournebize, J., 2011b. Selected pesticides adsorption and desorption in substrates from artificial wetland and forest buffer. Environ. Toxicol. Chem. 30, 1669–1676.

Passeport, E., Richard, B., Chaumont, C., Margoum, C., Liger, L., Gril, J.-J., Tournebize, J., 2014. Dynamics and mitigation of six pesticides in a "Wet" forest buffer zone. Environ. Sci. Pollut. Res. 21, 4883–4894.

Patterson, B.M., Davis, G.B., McKinley, A.J., 2002. Laboratory column experiments using polymer mats to remove selected VOCs, PAHs and pesticides from groundwater. Groundwater Monitoring & Remediation 22, 99–106.

Pei, R., Kim, S.-C., Carlson, K.H., Pruden, A., 2006. Effect of river landscape on the sediment concentrations of antibiotics and corresponding antibiotic resistance genes (ARG). Water Res. 40, 2427–2435.

Pesce, S., Lissalde, S., Lavieille, D., Margoum, C., Mazzella, N., Roubeix, V., Montuelle, B., 2010. Evaluation of single and joint toxic effects of diuron and its main metabolites on natural phototrophic biofilms using a pollution-induced community tolerance (PICT) approach. Aquat. Toxicol. 99, 492–499.

Petrovic, M., Lopez De Alda, M.J., Diaz-Cruz, S., Postigo, C., Radjenovic, J., Gros, M., Barcelo, D., 2009. Fate and removal of pharmaceuticals and illicit drugs in conventional and membrane bioreactor wastewater treatment plants and by riverbank filtration. Philosophical Transactions of the Royal Society a-Mathematical Physical and Engineering Sciences 367, 3979–4003.

Ray, C., 2004. Modeling RBF efficacy FOR mitigating chemical shock loads. American Water Works Association. Journal 96, 114–128.

Ray, C., Melin, G., Linsky, R.B., 2003. RIVERBANK FILTRATION: Improving Source-Water Quality.

Regnery, J., Wing, A.D., Alidina, M., Drewes, J.E., 2015. Biotransformation of trace organic chemicals during groundwater recharge: how useful are first-order rate constants? J. Contam. Hydrol. 179, 65–75.

Reichenberger, S., Bach, M., Skitschak, A., Frede, H.-G., 2007. Mitigation strategies to reduce pesticide inputs into ground-and surface water and their effectiveness; a review. Sci. Total Environ. 384, 1–35.

Rice, C.P., Bialek, K., Hapeman, C.J., McCarty, G.W., 2016. Role of riparian areas in atmospheric pesticide deposition and its potential effect on water quality. JAWRA Journal of the American Water Resources Association 52, 1109–1120.

Robert Budd, A.O.G., Goh, Kean S., Bondarenko, Svetlana, Gan, Jay, 2009. Efficacy of constructed wetlands in pesticide removal from Tailwaters in the Central Valley, California. Environ. Sci. Technol. 43, 2925–2930.

Robertson, W.D., Vogan, J.L., Lombardo, P.S., 2008. Nitrate removal rates in a 15-year-old permeable reactive barrier treating septic system nitrate. Ground Water Monitoring & Remediation 28, 65–72.

Rodrigo, M.A., Oturan, N., Oturan, M.A., 2014. Electrochemically assisted remediation of pesticides in soils and water: a review. Chem. Rev. 114, 8720–8745.

Rodrigo, S., Saez, C., Canizares, P., Rodrigo, M.A., 2018. Reversible electrokinetic adsorption barriers for the removal of organochlorine herbicide from spiked soils. Sci. Total Environ. 640-641, 629–636.

Rodríguez-Cruz, M.S., Sánchez-Martín, M.J., Andrades, M.S., Sánchez-Camazano, M., 2007. Modification of clay barriers with a cationic surfactant to improve the retention of pesticides in soils. J. Hazard. Mater. 139, 363–372.

Romain, V., Sylvie, D., David, B., 2015. Water residence time and pesticide removal in pilot-scale wetlands. Ecol. Eng. 85, 76–84.

Romero, L.G., Pizzolatti, B.S., Soares, M.B.D., Michelan, D., Sens, M.L., Amer Soc, A., Biol, E., 2010. Bank filtration: Application in rural areas. Case studies in Santa Catarina, Brazil. Amer Soc Agricultural & Biological Engineers, St Joseph 701P0210cd, 21–24.

Rotter, S., Sans-Piché, F., Streck, G., Altenburger, R., Schmitt-Jansen, M., 2011. Active bio-monitoring of contamination in aquatic systems—an in situ translocation experiment applying the PICT concept. Aquat. Toxicol. 101, 228–236.

Rühmland, S., Wick, A., Ternes, T.A., Barjenbruch, M., 2015. Fate of pharmaceuticals in a subsurface flow constructed wetland and two ponds. Ecol. Eng. 80, 125–139.

Sandhu, C., Grischek, T., 2012. Riverbank filtration in India - using ecosystem services to safeguard human health. Water Science and Technology-Water Supply 12, 783–790.

Sarmah, A.K., Meyer, M.T., Boxall, A.B.A., 2006. A global perspective on the use, sales, exposure pathways, occurrence, fate and effects of veterinary antibiotics (VAs) in the environment. Chemosphere 65, 725–759.

Schmitt-Jansen, M., Veit, U., Dudel, G., Altenburger, R., 2008. An ecological perspective in aquatic ecotoxicology: approaches and challenges. Basic and Applied Ecology 9, 337–345.

Sgroi, M., Pelissari, C., Roccaro, P., Sezerino, P.H., García, J., Vagliasindi, F.G.A., Ávila, C., 2018. Removal of organic carbon, nitrogen, emerging contaminants and fluorescing organic matter in different constructed wetland configurations. Chem. Eng. J. 332, 619–627.

Shan, N., Ruan, X.-H., Xu, J., Pan, Z.-R., 2014. Estimating the optimal width of buffer strip for nonpoint source pollution control in the three gorges reservoir area, China. Ecol. Model. 276, 51–63.

Sharma, L., 2012. Riverbank Filtration: Modeling Fate of Dissolved Organic Carbon, Transport of Escherichia Coli and Coupling with Aquifer Storage to Addtress Temporal Water Scarcity. Civil and Environmental Engineering. The University of Hawai.

Shen, C., Lu, W., Huang, Y., Wu, J., Zhang, H., 2015. Removal of bismerthiazol from water using zerovalent iron: batch studies and mechanism interpretation. Chem. Eng. J. 260, 411–418.

Shutes, R.B.E., 2001. Artificial wetlands and water quality improvement. Environ. Int. 26, 441–447.

Song, H.-L., Li, H., Zhang, S., Yang, Y.-L., Zhang, L.-M., Xu, H., Yang, X.-L., 2018a. Fate of sulfadiazine and its corresponding resistance genes in up-flow microbial fuel cell coupled constructed wetlands: effects of circuit operation mode and hydraulic retention time. Chem. Eng. J. 350, 920–929.

Song, H.-L., Zhang, S., Guo, J., Yang, Y.-L., Zhang, L.-M., Li, H., Yang, X.-L., Liu, X., 2018b. Vertical up-flow constructed wetlands exhibited efficient antibiotic removal but induced antibiotic resistance genes in effluent. Chemosphere 203, 434–441.

Tai, Y., Tam, N.F.-Y., Dai, Y., Yang, Y., Lin, J., Tao, R., Yang, Y., Wang, J., Wang, R., Huang, W., Xu, X., 2017. Assessment of rhizosphere processes for removing water-borne macrolide antibiotics in constructed wetlands. Plant Soil 419, 489–502.

Tournebize, J., Passeport, E., Chaumont, C., Fesneau, C., Guenne, A., Vincent, B., 2013a. Pesticide de-contamination of surface waters as a wetland ecosystem service in agricultural landscapes. Ecol. Eng. 56, 52–60.

Tournebize, J., Passeport, E., Chaumont, C., Fesneau, C., Guenne, A., Vincent, B., 2013b. Pesticide de-contamination of surface waters as a wetland ecosystem service in agricultural landscapes. Ecol. Eng. 56, 51–59.

Tournebize, J., Chaumont, C., Mander, Ü., 2017. Implications for constructed wetlands to mitigate nitrate and pesticide pollution in agricultural drained watersheds. Ecol. Eng. 103, 415–425.

Tuncel, S.G., Oztas, N.B., Erduran, M.S., 2008. Air and groundwater pollution in an agricultural region of the Turkish Mediterranean coast. J. Air Waste Manage. Assoc. 58, 1240–1249.

USDA-Natural Resources Conservation Service, U., A Handbook of Constructed Wetlands, In: Davis, L.

Vallee, R., Dousset, S., Schott, F.-X., Pallez, C., Ortar, A., Cherrier, R., Munoz, J.-F., Benoit, M., 2015. Do constructed wetlands in grass strips reduce water contamination from drained fields? Environ. Pollut. 207, 365–373.

Vandermeeren, P., Baken, S., Vanderstukken, R., Diels, J., Springael, D., 2016. Impact of dry-wet and freeze-thaw events on pesticide mineralizing populations and their activity in wetland ecosystems: a microcosm study. Chemosphere 146, 85–93.

Verstraeten, I.M., Heberer, T., Scheytt, T., 2003. Occurrence, characteristics, transport, and fate of pesticides, pharmaceuticals, industrial products, and personal care products at riverbank filtration sites. In: Ray, C., Melin, G., Linsky, R.B. (Eds.), Riverbank Filtration: Improving Source-Water Quality. Springer Netherlands, Dordrecht, pp. 175–227.

Vidal, J., Saez, C., Cañizares, P., Navarro, V., Salazar, R., Rodrigo, M.A., 2018. ZVI – reactive barriers for the remediation of soils polluted with clopyralid: are they really worth? Chem. Eng. J. 350, 100–107.

Vieira Dos Santos, E., Saez, C., Canizares, P., Martinez-Huitle, C.A., Rodrigo, M.A., 2017. Reversible electrokinetic adsorption barriers for the removal of atrazine and oxyfluorfen from spiked soils. J. Hazard. Mater. 322, 413–420.

Vignola, R., Bagatin, R., D'Auris, A.D.F., Massara, E.P., Ghisletti, D., Millini, R., Sisto, R., 2011. Zeolites in a permeable reactive barrier (PRB): one-year of field experience in a refinery groundwater. Part 2: zeolite characterization. Chem. Eng. J. 178, 210–216.

Vodyanitskii, Y.N., 2014. Artificial permeable redox barriers for purification of soil and ground water: a review of publications. Eurasian Soil Science 47, 1058–1068.

Vymazal, J., Březinová, T., 2015. The use of constructed wetlands for removal of pesticides from agricultural runoff and drainage: a review. Environ. Int. 75, 11–20.

Wan, J., Li, Z., Lu, X., Yuan, S., 2010. Remediation of a hexachlorobenzene-contaminated soil by surfactant-enhanced electrokinetics coupled with microscale Pd/Fe PRB. J. Hazard. Mater. 184, 184–190.

Wang, F., van Halem, D., van der Hoek, J.P., 2016. The fate of H2O2 during managed aquifer recharge: a residual from advanced oxidation processes for drinking water production. Chemosphere 148, 263–269.

Wilkin, R.T., Lee, T.R., Sexton, M.R., Acree, S.D., Puls, R.W., Blowes, D.W., Kalinowski, C., Tilton, J.M., Woods, L.L., 2018. Geochemical and isotope study of Trichloroethene degradation in a zero-Valent Iron permeable reactive barrier: a twenty-two-year performance evaluation. Environ. Sci. Technol. 53, 296–306.

Williams, C.O., Lowrance, R., Potter, T., Bosch, D.D., Strickland, T., 2016. Atrazine transport within a coastal zone in southeastern Puerto Rico: a sensitivity analysis of an agricultural field model and riparian zone management model. Environmental Modeling & Assessment 21, 751–761.

Yang, J., Cao, L., Guo, R., Jia, J., 2010. Permeable reactive barrier of surface hydrophobic granular activated carbon coupled with elemental iron for the removal of 2,4-dichlorophenol in water. J. Hazard. Mater. 184, 782–787.

Yang, L., He, J.-T., Su, S.-H., Cui, Y.-F., Huang, D.-L., Wang, G.-C., 2017. Occurrence, distribution, and attenuation of pharmaceuticals and personal care products in the riverside groundwater of the Beiyun River of Beijing, China. Environ. Sci. Pollut. Res. 24, 15838–15851.

Yang, Y., Zhao, Y., Liu, R., Morgan, D., 2018. Global development of various emerged substrates utilized in constructed wetlands. Bioresour. Technol. 261, 441–452.

Ye, J., Chen, X., Chen, C., Bate, B., 2019. Emerging sustainable technologies for remediation of soils and groundwater in a municipal solid waste landfill site – A review. Chemosphere 227, 681–702.

Yi, X., Tran, N.H., Yin, T., He, Y., Gin, K.Y.-H., 2017. Removal of selected PPCPs, EDCs, and antibiotic resistance genes in landfill leachate by a full-scale constructed wetlands system. Water Res. 121, 46–60.

Zabaloy, M.C., Garland, J.L., Gomez, M.A., 2010. Assessment of the impact of 2,4-dichlorophenoxyacetic acid (2,4-D) on indigenous herbicide-degrading bacteria and microbial community function in an agricultural soil. Appl. Soil Ecol. 46, 240–246.

Zak, D., Kronvang, B., Carstensen, M.V., Hoffmann, C.C., Kjeldgaard, A., Larsen, S.E., Audet, J., Egemose, S., Jorgensen, C.A., Feuerbach, P., Gertz, F., Jensen, H.S., 2018. Nitrogen and phosphorus removal from agricultural runoff in integrated Buffer zones. Environ. Sci. Technol. 52, 6508–6517.

Zhang, S., Song, H.-L., Yang, X.-L., Li, H., Wang, Y.-W., 2018a. A system composed of a biofilm electrode reactor and a microbial fuel cell-constructed wetland exhibited efficient sulfamethoxazole removal but induced sul genes. Bioresour. Technol. 256, 224–231.

Zhang, W., Shan, N., Bai, Y., Yin, L., 2020. The innovative application of agriculture straw in in situ field permeable reactive barrier for remediating nitrate-contaminated groundwater in grain-production areas. Biochem. Eng. J. 164, 107755.

Zhang, X., Liu, X., Zhang, M., Dahlgren, R.A., Eitzel, M., 2010. A review of vegetated buffers and a Meta-analysis of their mitigation efficacy in reducing nonpoint source pollution. J. Environ. Qual. 39, 76–84.

Zhang, X., Jing, R., Feng, X., Dai, Y., Tao, R., Vymazal, J., Cai, N., Yang, Y., 2018b. Removal of acidic pharmaceuticals by small-scale constructed wetlands using different design configurations. Sci. Total Environ. 639, 640–647.

Zhang, X.-X., Zhang, T., Fang, H.H.P., 2009. Antibiotic resistance genes in water environment. Appl. Microbiol. Biotechnol. 82, 397–414.

Zhu, Y.-G., Johnson, T.A., Su, J.-Q., Qiao, M., Guo, G.-X., Stedtfeld, R.D., Hashsham, S.A., Tiedje, J.M., 2013. Diverse and abundant antibiotic resistance genes in Chinese swine farms. Proc. Natl. Acad. Sci. 110, 3435–3440.

Zolgharnein, J., Shahmoradi, A., Ghasemi, J., 2011. Pesticides removal using conventional and low-cost adsorbents: a review. Clean-Soil Air Water 39, 1105–1119.